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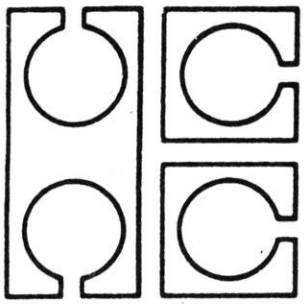

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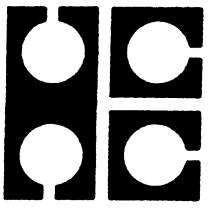

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PROCEEDINGS OF

THE TWENTY THIRD ANNUAL CONFERENCE ON ECOSYSTEMS RESTORATION AND CREATION

May, 1996

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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 Third 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 three years ago. We are grateful for his help and participation. Appreciation is also extended to Charles Duesner for providing administrative support for the conference.

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Thanks are extended to the staff of **Two Rivers Ranch** for arranging and conducting very successful field trip to wetland restoration sites.

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

To all these people, thank you.

THE ESTABLISHMENT OF RECOVERY THRESHOLDS IN A PETROLEUM IMPACTED TIDAL MARSH

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ABSTRACT

On January 2, 1990, a pipeline connecting two refineries on the Arthur Kill and Kill Van Kull waterways between New York and New Jersey ruptured, spilling 1,750,000 liters of No. 2 fuel oil which negatively affected approximately 90 hectares of intertidal marsh but severely damaged some 10 hectares. B.Laing Associates, Inc. put forward the thesis that the PRP should seek not only to clean up the affected wetlands but also to restore them to their pre-spill condition. A literature search turned up no examples of either laboratory or field attempts to accomplish such restoration and remediation. Samples of heavily oiled sediments were collected from two locations on the marsh surface. A sample was also collected from marshlands which were known not to have been affected by the spill. The heavily oiled samples were then serially cut with reference material to 50, 25, 12.5, 6.25 and 3.125 percent of their original strength. The heavily oiled samples were found to have 1,700 ppm saturated hydrocarbons (SHC). Seed from S. alterniflora (obtained from Maryland) were planted in shallow polyethylene containers at a rate of 100 per plot. Each plot was watered and seedling counts/heights were recorded daily.

The results in germination rates among the seeds varied significantly as a function of oil concentrations in the sediments of each plot. A test of statistical confidence showed that the number of germinating plants in the heavily oiled materials was significantly lower than those germinating in the serial cuts at a 99 percent level of statistical significance. The results were also subjected to a Student's t test and f test. The test found that all

three groups were in fact separate from each other. Thus, a threshold of oiling occurs at which point one can attempt to re-establish a Spartina alterniflora tidal wetland with a reasonable expectation of success. That threshold occurs at approximately 1,000 ppm SCH.

Field scale plots of 0.8 hectares were seeded the following summer (late June 1991) in heavily oiled and denuded areas. Approximately 50 percent of the area was successfully revegetated (Engineering News Record, October 1991) and returned the following spring (1992).

INTRODUCTION

On January 2, 1990, a pipeline connecting two refineries on the Arthur Kill and Kill Van Kull waterways between New York and New Jersey ruptured, spilling 1,750,000 liters of No. 2 fuel oil into these tidal channels and adjacent wetlands. The spill negatively affected approximately 90 hectares of intertidal marsh but severely damaged some 10 hectares. Under contract to Exxon Co. USA, Inc., B.Laing Associates, Inc. put forward the thesis that the potentially responsible party (PRP) should seek not only to clean up the affected wetlands but also to restore them to their approximate pre-spill condition. Aside from being the correct approach in any case, the restoration of the marsh was also expected to enhance biodegration of petroleum hydrocarbons. A literature search turned up no examples of either laboratory or field attempts to accomplish such restoration and remediation through active management of the resource. Therefore, B.Laing Associates, Inc. undertook a methodical approach with the aim of demonstrating whether or not active manipulation of an oil-damaged wetland could be undertaken with a reasonable expectation of success.

LITERATURE

As indicated above, the literature in 1990 was devoid of any active attempts at resource management to restore (i.e., doing more than cleaning) oil-damaged environments. However, one small piece of information indicated that such an approach may be worthwhile to attempt. In 1978, Kathryn A. Burns and John M. Teal analyzed the effects of a 1969 No. 2 fuel oil spill from the barge Florida on a Spartina spp. intertidal marsh on the coast of West Falmounth, Massachusetts. The barge spilled 700,000 liters of oil and had a mortality rate of up to 95 percent for macro biota in the marsh. However, eight years later, "plant regrowth was accelerated in 1973 and 1975, paralleling a decrease in aeromatic hydrocarbons in surface sediments." and in 1977, "dead areas remained, where sediments contained greater than about 1 mg/g mud." That is, when recovery was left to occur "naturally", a threshold of oil contamination appeared to exist below which active marsh re-establishment might be attempted.

METHODS

The initial location to test the active management hypothesis was in the laboratory where the many extraneous variables affecting plant growth could be controlled. The experiment had to be designed to reflect the initial conditions of heavy oiling that had resulted in high plant mortality and the expected amelioration of oiling levels as both gross saturated petroleum hydrocarbons (SHC) and its constituents declined. The macrophytic species which had suffered most heavily from the spill was salt marsh cordgrass (Spartina alterniflora). This plant species populates mean high tide to spring high tide flats in marshlands of the middle Atlantic states.

The experimental design was laid out as follows:

1. Surface sediments were collected from areas denuded of Spartina spp. as a clear result of the January 1990 spill. Such resource damage had been determined by Natural Resource Damage Assessments (NRDA). The sediments were collected from dominantly organic areas with a silty mix, as these locations were most likely to have absorbed and retained large fractions of the spilled product. Surface sediments were defined as 0 to 5 centimeters in depth.

2. Surface sediments were collected from areas of the southern Arthur Kill which were known to be unaffected by the spill through the same NRDA. Such materials (i.e., reference materials) were to be collected amongst, if not within, a thriving stand of Spartina alterniflora. This material was a coarse sand.

3. All sediments were collected using separate, new, unoiled spades and stored in separate, new plastic cans lined with plastic bags, sealed with tight lids and little head space. All materials were to be "homogenized" within the two groups as best as was possible in the field. The cans were to be left for one to two days (sealed) to allow free moisture to settle to the bottom. Both the heavily oiled and reference materials were then sampled and tested for SHC using gas chromatographic procedures specific to the NRDA process but equivalent to New York State Department of Health Methods 310.13 and 310.14.

4. The materials were transported to a college campus on Long Island, New York. The materials were weighed and serially "cut" with kitty litter (moistened to provide equivalent weight/volume ratios). The result was that six "batches" were created at 100, 50, 25, 12.5, 6.25 and 3.125 percent of the original heavily-oiled SHC concentrations. Each batch was then divided into three separate, new plastic tubs. A second set of three tubs each was also established with 100 percent oiled materials. Three, 100 percent reference material and three, 100 percent

kitty litter tubs were established. The material was observed carefully for native Spartina alterniflora seed and none were found.

5. Each tub also received 100 seed of Spartina alterniflora. The seed originated in the Delmarva Peninsula from a reputable dealer used previously by B.Laing Associates, Inc. Such seed had prior germination rates of 50 percent or higher. The seeds were placed 0.5 to 1.0 centimeters below the material surface.

6. The tub area was underlain with two sheets of plastic liner and received full sunlight from dawn until dusk. The tubs were watered and checked every day for new growth. Fresh, potable water was allowed to sit for at least 24 hours to dechlorinate prior to use. The tubs were rotated randomly on a weekly basis. The whole area was fenced in with chicken wire (four feet tall) was covered nightly with a plastic tarpaulin and was uncovered each morning. No animals (birds, squirrels, etc.) were ever observed in the enclosure.

7. Ten to fourteen days following planting of the Spartina alterniflora seed, germination of those seeds began to become apparent. The seeds are approximately 0.5 to 1.0 centimeters in length, have serrated edges and are oblong in shape; that is, they are readily recognized. At that time, the number of seed germinated and their height was recorded daily in each tub. This paper presents the results of the germination data.

8 After two months (June and July 1990), the data were compiled and subjected to mathematical analyses. The first analysis was a test of statistical significance of the means at 99 percent confidence intervals for number of seeds germinated. The grouping of heavily oiled material with and without serial cuts and reference materials were then subjected to an F statistic test and a Student's t analysis to determine if the groupings would also be mathematically non-random.

9. A Finding of statistically significant differences in germination rates between the heavily oiled and any of the serially cut materials plus a finding that germination rate varied in a non-random fashion among the heavily oiled or "cut" groups of tubs would indicate that below a certain threshold of SCH, tidal macrophytes would have a significantly improved chance of survival. That is, active management of an oil-impacted tidal marsh would likely succeed. With no mathematical separation of the results (i.e., the germination patterns observed were random), active management of an oil-impacted tidal marsh would have little chance of success.

RESULTS

The experiment in establishing thresholds for recovery of tidal marshes yielded positive and interesting results. The first mathematical analysis was to apply an averaging of the results with a test for statistical significance between the means. The sets of three tubs each were first placed in four separate groups. The first group was the two 100 percent heavily oiled sets (six tubs). The second group is the first three serial cuts (50, 25 and 12.5 percent heavily oiled materials) totaling nine tubs. The third group was the last two serial cuts (6.25 and 3.125 percent heavily oiled material) and the 100 percent reference material (all coarse sand), totaling nine tubs. The fourth "group" was the three tubs of 100 percent cat litter. The second statistical analysis was an "F" statistic test to determine if the four groups are differing populations. Finally, the Student's t test determines which groups actually result in these differences. Table 1 presents these statistical analyses of germination rates.

The sediment materials were also tested for pH and nitrogen, phosphorus and potassium using an Hach field test kit. The only item being significantly variant between the groupings were pH and potassium. In the reference materials (coarse sands) pH was distinctly

Table 1

Germination and Statistical Results

tub #	1st grp	2nd grp	Treatment	Germ. # Seeds	Avg.	C.I.*	1st/2nd grp F Stat. (1)	1st grp t-test	2nd grp t-test
25	1	B	100%	4					
26	1	B	oiled	4				1vs.2-3.48**	Bvs.C6.00**
27	1	B		4					
1	1	B		13				1vs.3-1.53	
2	1	B		2					
3	1	B		6	5.33	2.91**	11/21.1**	1vs.4-3.36**	
4	2	A	50%	14					
5	2	A	oiled	22				2vs.3-2.19(3)	Avs.B4.20**
6	2	A		29					
7	2	A	25%	18				2vs.4.2-90**	Avs.C3.30**
8	2	A	oiled	18					
9	2	A		30					
10	2	A	12.5%	25					
11	2	A	oiled	20					
12	2	A		21	21.89	3.25**	11/21.2**		
13	3	A	6.25%	12					
14	3	A	oiled	39				3vs.4-4.46**	
15	3	A		10					
16	3	B	3.125%	3					
17	3	B	oiled	29					
18	3	B		2					
19	3	B	100%	4					
20	3	B	Reference	0					
21	3	B	(control)	15	12.67	8.19	11/21.2**		
22	4	C	Cat	30					
23	4	C	Litter	44					
24	4	C	(control)	43	39.00	7.22**	11/21.2**		

- number; grp - group; Avg - Mathematical average; % - Percent.

* All statistical significance levels at 99 percent.
** Groups showing statistically significant results per *.

(1) F test at 4.98 and 5.57 for groupings, respectively.
(2) Student t significant at 2.72 and 2.50 for groupings, respectively.
(3) Very close, passes at 95% confidence.

acidic (4.5 units) and declined steadily from the heavily oiled materials (silts and clays - pH 7.0) and the cat litter (pH 7.0) which were of neutral pH. Further, potassium levels were "very low" in the reference materials versus "medium" levels in the other groups.

Finally, SCH levels in the materials were recorded at 1,700 parts per million (ppm) in the heavily oiled samples and 184 ppm in the reference materials. Thus, the concentration of SCH in the cuts was cut 1 -, 942 ppm; cut 2 - 563 ppm; cut 3 - 373 ppm; cut 4 - 278 ppm and cut 5 - 231 ppm.

DISCUSSION

The most immediately striking result is the number of statistically significant results obtained. With regard to the simple averages, the heavily oiled material (group 1) showed very low germination averages (5.33) with a very low confidence interval variance (2.91) versus the much higher 50, 25 and 12.5 percent heavily oiled group's (no. 2) average (21.89) and low confidence interval variance (3.25). The two groupings (no. 1 and 2) also separated themselves from the cat litter control (group 4) which had the highest average germination (39.00) and a moderate confidence interval variation (7.22). The F statistic test showed that, as a population, these groups all varied significantly from each other. The student's t test of these four groups all showed a strong separation between groups 1 (100 percent heavily oiled), 2 (50, 25 and 12.5 percent heavily oiled) and 4 (cat litter). Groups 1, 2 and 4 also separated themselves significantly from group 3(6.25 percent and 3.125 percent heavily oiled materials and 100 percent reference materials) but the reverse was not true. These results clearly demonstrate that the heavy oiling of sediments was retarding the germination of Spartina alterniflora seeds.

The mechanism of such an inhibition is not clear but almost certainly it must relate to the uptake of petroleum constituents, perhaps "aeromatic" constituents as suggested by Burns and Teal (1978). This hypothesis is bolstered by the fact that seed germination includes very important moisture and nutrient uptake mechanisms. If this mechanism is either interrupted or substituted by a "toxic" constituent, then germination will not occur. This result is especially supported by the fact that only the oil free material (the cat litter - group 4) significantly exceeded all other test plots in numbers germinated. The threshold at which such "toxic" effects may occur is shown in the statistically significant separation between the 100 percent heavily oiled group and the second group (no. 2) consisting of the 50 percent, 25 percent and 12.5 percent heavily oiled materials. The SCH levels in these two plots was 1,700 and 942 ppm, respectively. Therefore, the results of the experiment clearly showed a "threshold" of oiling above which active intervention using seeding would fail and below which it would likely succeed. This threshold was and is 1,500 to 1,000 ppm and it bears a striking resemblance to the only previously existing literature's observation, "dead areas remained, where sediments contained greater than about 1/mg/g mud." (i.e., 1,000 ppm oil).

The only "fly in the ointment" of these results are the poor germination of those tubs where the reference materials became dominant and the silty, oiled materials became insignificant (in soils, content of one soil fraction less than 10 percent usually decreases the significance of its influence to virtually mill). In this case, germination in group 3 (6.25 and 3.125 percent heavily oiled material and 100 percent reference material) falls off from the first serial cut group 2 (50, 25 and 12.5 percent heavily oiled materials) and the group 4 control (100 percent cat litter) averaging 12.67 versus 21.89 and 39.00, respectively. Also, the Student's "t" test (but not the "F" statistic) would "group" the 93.5 to 100 percent reference material tubs with the 100 percent heavily oiled tubs (see 2nd grouping A, B & C in Table 1). Significantly, however, variability was greatest in group 3. While the other groups separated from each other and group 3, the results viewed in reverse (group 3 versus the others) was not significantly different

due to the variability of group 3 alone. This result suggests that some other factor is at play in group 3 versus the heavily oiled group 1. The key is in the soil chemistry. Spartina alterniflora has a difficult time germinating and surviving in high or low pH substrates. The soil chemistry results showed an acidic pH of 4.5 units in the reference material (sands) verses a neutral pH of 7.0 in the heavily oiled (silts) materials. Acidic pH's drop nutrient availability (low to medium in all tests) exponentially as the metallic forms precipitate or chelate out of solution. Thus, low pH most likely interfered with the absorption of water and nutrients by the seeds in those tubs, resulting in reduced germination (but still substantially higher on average than in the 100 percent heavily oiled group 1). The highly variable nature of this germination indicates that a different factor or set of factors is affecting germination in group 3 versus group 1. The most likely explanation is that group 3 lacks some vital material or condition (i.e., a pH-induced water or nutrient imbalance), hence the wide variability, and that group 1 is directly impacted by some material or condition (toxicity due to oiling), hence the low variability.

CONCLUSION

In summary, the statistically tested results show a clear separation in germination rates between 100 percent heavily oiled and less contaminated materials. These groups all separate themselves out from each other at a 99 percent level of statistical confidence except that the variability of group 3 greatly exceeds the variability of the other groups. This effect occurred to group 3 only when sands became the dominant substrate and pH dropped to acidic levels. Further, group 4 with no petroleum contamination, had the highest germination rate of all groups and was significantly different from all other groups. The most significant separation, for the purposes of this study, occurred between group 1 (100 percent heavily oiled material) and group 2 (50, 25 and 12.5 percent heavily oiled material). The SCH levels of these groups was 1,700 ppm

and 942 ppm, respectively. It can be concluded that tidal marshes with SCH levels exceeding 1,000 to 1,500 ppm are still relatively toxic to seeded Spartina alterniflora. However, if one actively intercedes to seed the oiled tidal marsh with this plant species (with otherwise proper tidal ranges, minimal wave impacts, the physically proper substrate, etc.) at levels less than 1,000 ppm to 1,500 ppm SCH, the chances for successful re-establishment of this macrophyte are very good. Therefore, following petroleum spill clean-up operations, it is possible to actively manage a tidal wetland and accelerate "repairs" to damage resulting from the toxic effects of petroleum products.

POST SCRIPT

In the year following both the spill and this experiment, 1991, B.Laing Associates, Inc. undertook a field test of the prior results. Field scale plots of 0.8 hectares were seeded in late June 1991 on heavily oiled and denuded areas. Approximately 50 percent of the area was successfully revegetated (Kosawatz, Engineering News Record, November 1991) and sprouted again the following spring (1992). No funds were available to statistically record these efforts and the program was then taken over by the New York City Department of Parks. By the fall of 1995, the New York City Department of Parks has successfully revegetated the 23 most heavily impacted and denuded acres. Further, they reported a drop in SHC levels at those revegetated locations from a high of 50,000 ppm to non-detect in two growing seasons (A. Bergen, Contaminated Soils Conference, Amherst, MA, October 1995). Their use of plugs rather than seed and a deeper sampling methodology most probably explains their success at initially higher SHC levels. In any event, their results demonstrated at full scale that active management of tidal marshes is an effective, worthwhile technique to restore and so, further the clean-up of these oil damaged environments.

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SPOIL ISLAND ENHANCEMENT FOR SHOREBIRD NESTING HABITAT IN THE SOUTHERN INDIAN RIVER LAGOON, FLORIDA.

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ABSTRACT

Several species of Larids use South Florida's coastal regions each year as nesting, wintering and roosting sites. Habitat, destruction, degradation and alteration, anthropogenic disturbances at breeding colonies, as well as other factors, have caused drastic declines in several tern species. In particular the least tern (*Sterna antillarum*), is now a threatened species in the state of Florida. Least terns require unvegetated, open, sandy areas free from disturbance and mammalian predators. Forced from their natural nesting habitat, least terns currently nest upon rooftops or other man-made structures in the region.

The goal of this project is to offer an alternative to rooftop nesting by enhancing three relatively ecologically unproductive spoil islands in the Indian River Lagoon to create suitable tern nesting habitat. This is the first year of a multi-year study which will monitor shorebird use on enhanced spoil islands in St. Lucie County, Florida. Three spoil islands were selected, devoided of exotic vegetation, and prepared for shorebird nesting. Weekly observation by boat of the sites occurred through the nesting season (April through August) to record the number of individuals per species; courtship, mating, and incubating behaviors; nesting; and human disturbance. Although one island was significantly disturbed by vandals in April, 1996, successful nesting was observed during the months of June and July on the island.

INTRODUCTION

Decrease in shorebird populations began in the late 1880's through the early 1900's as plume hunters for the millinery trade decimated wading bird and shorebird populations. Specifically, least terns were hunted for their plumage which was used to adorn women's hats of that period. Because of public outcry, these practices ceased and populations began to rebound in the 1930's and 1940's (Sprunt, 1954). However, as Florida's population began to grow along the coasts, shorebird populations again faced decline.

As this pressure increased, displaced least terns have been forced to locate alternative nesting sites. These areas include large, flat, gravel covered roofs of commercial buildings (Fisk, 1978). In many areas of the state, most least tern nesting occurs on rooftops (Hovis and Robson 1989, Gore, 1991).

Though rooftops are not ideal, they provide some advantages to nesting terns. Rooftop colonies, experience less human disturbance, very little mammalian predation, and roofs are not affected by high tides. However, roofs tend to retain rainwater for extended periods inundating nest. In addition, the eggs may roll in high winds because of a lack of sufficient substrate for making a scrape; and chick mortality may increase as a result of falls from the roof before or during fledging. Ironically, these habitats too are in jeopardy with the introduction of alternative roofing materials and building techniques (Gore and Kinnison, 1991).

Least terns prefer to nest in open sandy coastal areas; however, they will nest in substrates of coarse pea gravel and even asphalt (Lowe et al. 1990, Sprunt 1954). Early in the season high altitude courtship flights are common. Following this activity, fish feeding and courtship dances occur in the vicinity of, if not on, the selected colony site (Lowe et al. 1990, J. Tutton pers. comm.). Copulation occurs after this stage with the female taking a fish from the male during or shortly following the mating event (Lowe et al. 1990, J. Tutton, pers. comm.). Once a pair bond is

established, they will make one or more scrapes at the selected site (Fisk, 1978, J. Tutton pers. comm.). At this point, territorial behavior may be displayed when other least terns encroach into the area. This territorial display is a subtle tail raise accompanied by pointing the bill toward the ground, it is not uncommon for both individuals in a pair to exhibit this behavior (Tutton, pers. comm.).

SITE DESCRIPTION

The Indian River Lagoon (IRL) (Fig. 1) is a shallow bar built estuary extending 156 miles from Volusia through Palm Beach counties, making up 40 percent of Florida's east coast. Dredging of the Indian River Lagoon for construction of the Intracoastal Waterway began in the 1950's by the U.S. Army Corp of Engineers. The spoil was deposited adjacent to the channel creating a series of islands throughout the lagoon. In 1961, 34 spoil islands were created in St. Lucie County (Brown-Peterson and Eames, 1989). Upon the initial creation of the spoil islands, it was documented that least terns nested on the new spoil until vegetation, typically exotics, overtook islands (Schreiber, and Schreiber, 1978).

Today, in most cases, exotic species such as Australian pine (*Casuarina* sp.) and Brazilian pepper (*Schinus terebinthifolius*) dominate the spoil islands. A typical spoil island is composed of a monotypical stand of Australian pine in the interior which is flanked by Brazilian pepper and some native tropical hardwoods, and a mangrove fringe. Though dominated by exotics, some islands support large wading bird breeding colonies (pers. obs.). In 1988 the former Florida Department of Natural Resources, now Florida Department of Environmental Protection (FDEP), developed a management plan for the spoil islands of the Indian River Lagoon (Brown-Peterson and Eames, 1989). For each island's, management recommendations, an extensive biological inventory, and a designation of use (conservation, education or recreation) was included in the plan. Of the 34 spoil islands in St. Lucie County, there are 18 recreation, 14 conservation, and 2 education islands.

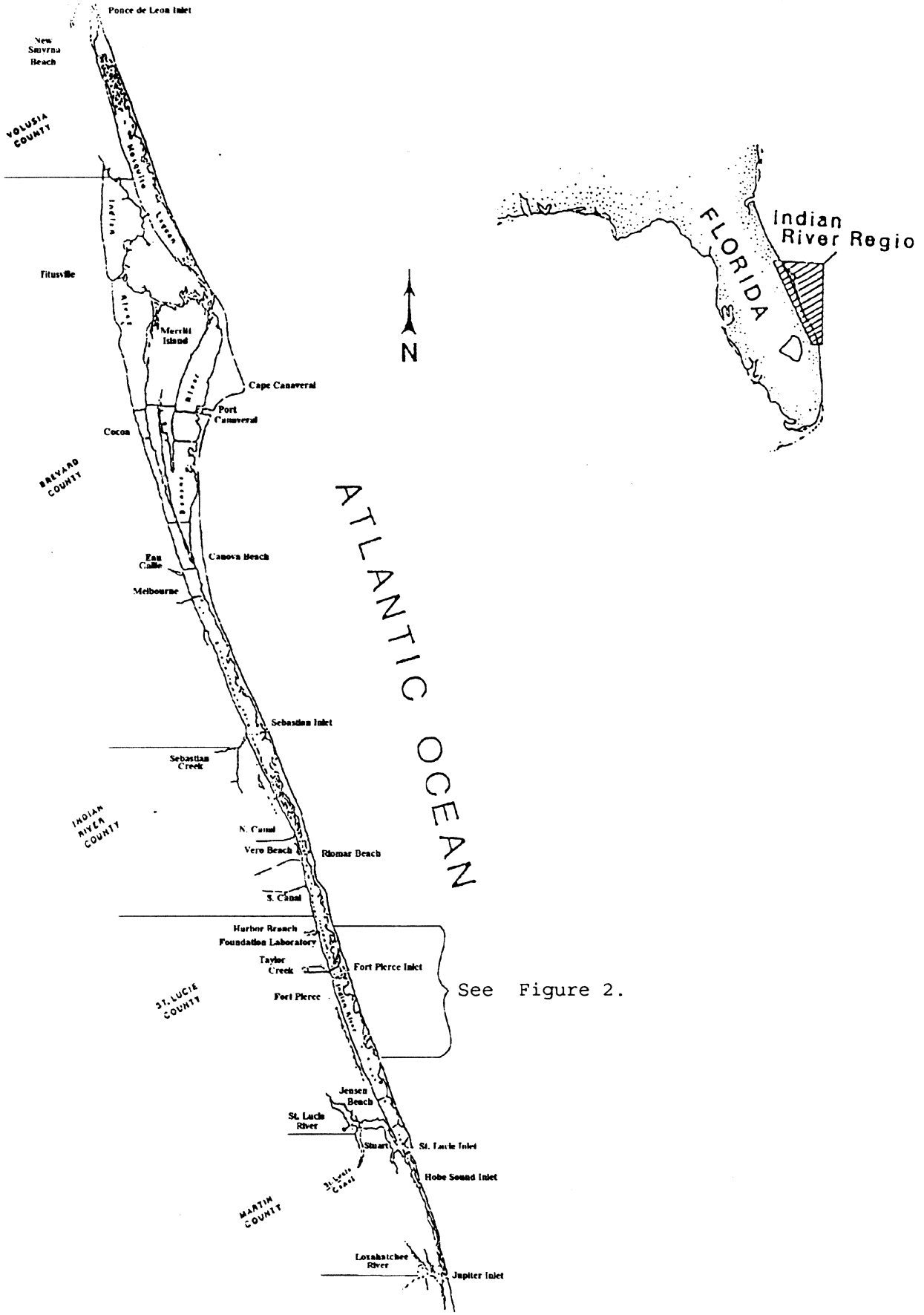

Figure 1. Map of the East Coast of Florida highlighting the Indian River Lagoc Region and study site.

The three spoil islands selected are located in St. Lucie County near Fort Pierce, Florida. St. Lucie #4 (SL4) is situated at 27 31.9 W and 80 19.1 N, to the east of Harbor Branch Oceanographic Institution. The other two sites, SL19A and SL19B are located at 27 20.6 W and 80 16.0 N and 27 21.2 W and 80 15.6 N, respectively, and are west of the Florida Power and Light (FPL) Nuclear Power Plant, Hutchinson Island (Figure 2).

MATERIALS AND METHODS

Spoil island selection was based on the following five criteria: 1) size; 2) proximity to an inlet; 3) vegetation present on the site; 4) substrate; and 5) historic recreational usage. To minimize cost and labor, small islands less than 1 hectare (2.49 acres) were selected. Because larids frequently tend to congregate close to inlets, islands in proximity to inlets should have a higher probability of success. Islands dominated by exotic vegetation were selected to serve two additional purposes: 1) to eliminate or minimize impact to native vegetation, in particular mangroves; and 2) to remove exotics from the Indian River Lagoon Aquatic Preserve. The substrate present at each site exhibited a wide range of textures and sizes suitable for shorebird nesting. The determination of recreational use was made through the use of the Indian River Lagoon Spoil Island Management Plan as well as site visits to candidate islands during holidays, weekends and weekdays. The only potential sites which met these criteria (SL4, SL19A and SL19B) were unfortunately designated for recreation and do receive recreational use by boaters. However, conservation and education islands were not options as they are ecologically important in their present state.

The first phase of island preparation began in November 1995. Concurrently, SL4 and SL19B were cleared manually with chainsaws and hand tools by staff from the FDEP, US Fish and Wildlife Service (USFWS), and FPL. Each site took two days to clear. Similarly, SL19A was cleared in December, 1995. Mechanical removal was not an option because of the shallow, rich seagrass meadows surrounding the islands, their small size, and high cost. Australian pine, Brazilian pepper and other

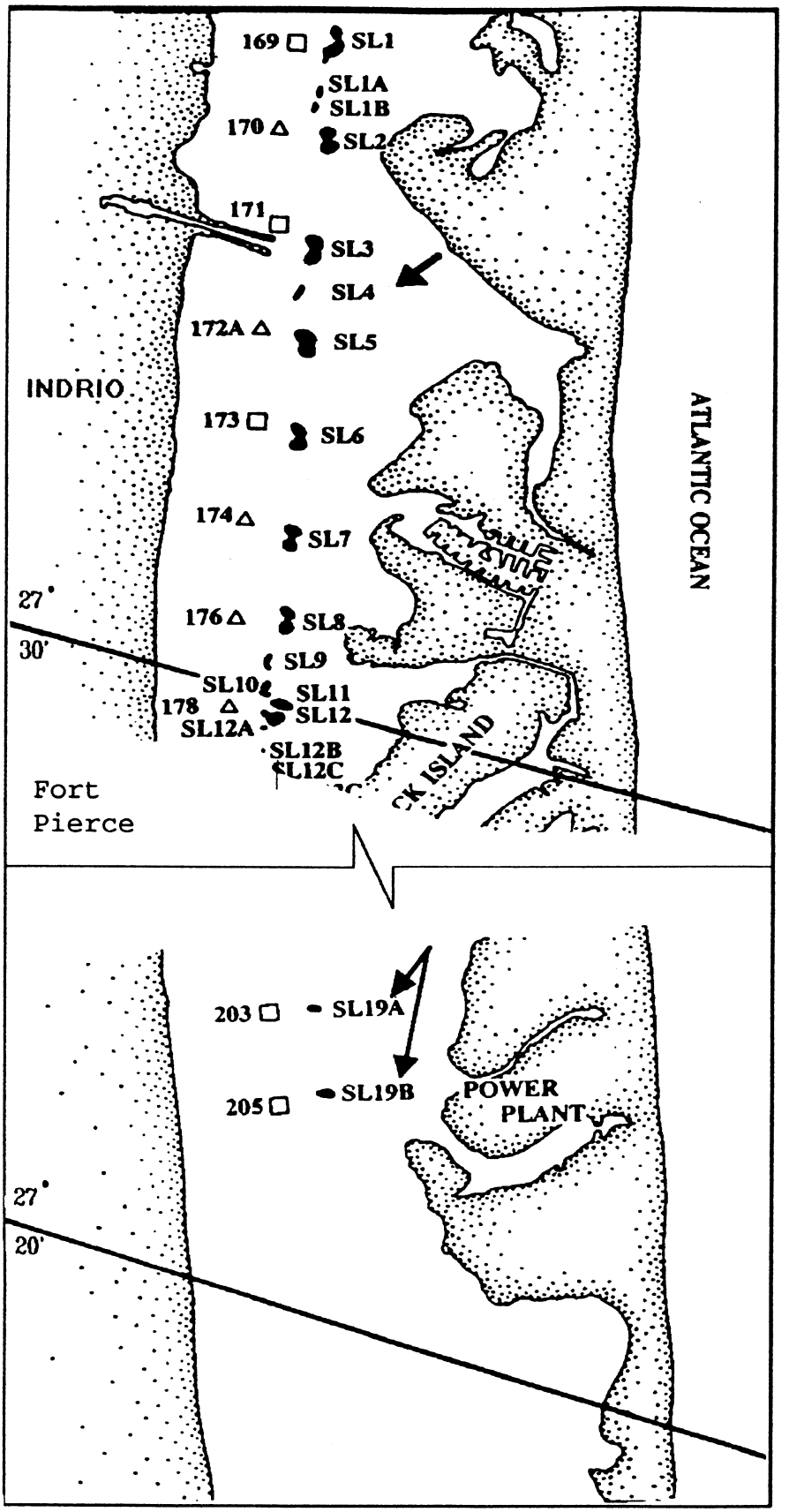

Figure 2. Inset from Figure 1 indicating Spoil Island locations (Brown-Peterson and Eames, 1989.)

vegetation were cut at ground level with the stumps left in place to minimize erosion and treated with a glyphosate herbicide to prevent regrowth. SL4 contained some native vegetation, which included one white mangrove (*Laguncularia racemosa*), one black mangrove (*Avicennia germinans*) and a patch of sea oxeye daisy (*Borrichia frutescens*) which were not removed.

Removal of the debris from the site was not a viable alternative; therefore, logs and branches were piled and left to dry for three months. In mid-February 1996, the debris on SL4 and SL19B was burned in compliance with a Division of Forestry open burn permit. However, after two failed attempts to burn the debris on SL19A because of adverse weather conditions, it was decided to complete the burn in the fall and concentrate enhancement efforts on SL4 and SL19B.

After several rain events, the carbon residue from the burn had mostly washed away. These areas were blended with the lighter colored substrate by spreading sand manually. Final preparation of the sites consisted of making the island as uniform as possible by raking the area smooth, covering stumps with sand, treating regrowth with glyphosate, and reducing the vertical structure on the island. Least terns prefer to nest in areas where predators are easily viewed from a distance without the interference of vertical structure (Smith, pers. comm.). Minimizing the abundant vertical structure on SL4 was therefore a priority. Australian pines undermined by erosion and partially submerged were not removed; however, their large root structures and branches were reduced as much as possible. Stumps on SL19B were cut as low to the ground as possible, but will need further reduction before the 1997 breeding season.

To give the illusion that terns were already utilizing the site or were possibly mating, life size least tern decoys were constructed and placed on each island. Templates for the decoys were obtained and transferred onto 2.5cm x 25cm x 1.21m (1"x10"x4') white pine planks using a Sulky Iron Transfer Pen and tracing paper. The decoy image on the tracing paper was ironed onto the plank using a household iron. The traced images were then

cut-out using a skill saw, and sanded. A .6mm (.25") hole was drilled for a 15cm x .6mm (6"X 1/4") dowel to support the decoy. Each decoy was primed with white spray paint, dried, and painted with least tern markings (grey wings; black cap and primaries; and yellow bill) using exterior grade latex paint. A total of 45 finished decoys and 50 rough decoys were constructed in one day by eleven FDEP and USFWS staff.

Addressing the need to ward off recreational use by boaters, eight signs were purchased. These signs read, "Do Not Disturb March 1-August 31, Tern Nesting Research Area, Disturbance May Cause Nest Abandonment" and acknowledged the partners of this project. The signs measured .6m x 1.22m (2'x4') with the largest letters (Do Not Disturb) measuring 7.5cm (3"). Four signs and 16 decoys and two signs and 13 decoys were placed on SL4 and SL19B in late March respectively. During this final trip to the islands, one least tern was observed scouting overhead at each island.

Each island was monitored twice a week beginning the first week of April 1996 and continued through August. The sites were observed at various times of the day for periods ranging from 20 minutes to four hours. The number of individuals per species, behavioral patterns (courting, mating, nesting, etc.), orientation on island/relative proximity to decoys, number of decoys, weather conditions, and evidence of disturbance by humans or predators were recorded. Precautions were taken not to disturb the birds during the monitoring process; therefore, the sites were approached with caution and the boat was positioned a safe distance offshore (Rodgers, and Smith, 1995).

RESULTS

SPOIL ISLAND SL4

As expected, species diversity was the greatest during the month of April as the last of the wintering species remain, migrants move through, and nesting species arrive. On SL4 a total of 12 species were recorded totaling

191 individual birds. The greatest number of individuals for the month of April, 92 royal terns (*Sterna maxima*), were observed on 18 April 1996 (Table 1). Also on that date, the first least terns were observed exhibiting courtship behavior, and several tern and shorebird species were loafing on the island.

However, monitoring of SL4 on 26 April revealed obvious evidence of human and canine pet disturbance. The first indication of disturbance was the lack of birds on the island. Upon closer inspection, trash, dog prints, and footprints were observed. Ten decoys were stolen and the remaining six decoys were found in the vegetation.

Two days later, additional signs with stronger language were posted and ten new decoys replaced those which were stolen. Unexpectedly, while posting signs, one pair of least terns were observed mating.

Subsequent monitoring days in May after the disturbance revealed little or no use of the site by any bird species. However, royal terns, least terns, brown pelicans (*Pelecanus occidentalis*), and double-crested cormorants (*Phalacrocorax auritus*) were observed loafing on a sand bar just north of SL4 at low tide (Table 2). Several least terns were flying overhead exhibiting courtship behavior and did briefly land on SL4. Signs that birds were possibly returning to the site occurred in late May when two least terns were seen loafing on the island.

On 4 June 1996, 35 least terns and three royal terns were seen loafing and an indication of the first nest was revealed. One least tern was observed sitting in incubating posture. It was located close to the decoys, high on the southern sand bar, and in an area where nesting was anticipated (Table 3). The following week, on 13 June, the nest was confirmed when the least tern was located in the same location and exhibiting incubating behaviors such as egg rolling and territorial tail raising. Another adult least tern returned continually, assumed to be the mate, to feed and relieve the incubator. The bird incubating would stand up periodically, circle the scrape, and then move the egg with its bill. The pair acted defensively as

other least terns occasionally approached.

A total of 38 least terns, one brown pelican, and four double- crested cormorants were also on the island. The least terns were actively exhibiting courtship behavior and one pair was observed mating. Of the 38 least terns, five were juveniles. They are probably fledglings from successful nearby rooftop colonies, the nearest of which is located approximately 10km away.

The next day, 30 least terns, two least tern fledglings, four double-crested cormorants, and one brown pelican were observed. The same least tern was incubating while one other least tern appeared to be selecting a nest site. The other least terns were loafing, foraging, and courting possible mates.

Observations on June 22, 1996, revealed 48 least terns, several juveniles, and as many as four additional nests. Three least terns now were incubating, two were site selecting, two pairs were mating, and there was evidence of additional scraping on the top of the sand spit. All birds flushed and immediately returned as an osprey (*Pandion haliaetus*) flew over the island.

Three of the new nests were oriented North to South, almost in a straight line, with the first nest (A) at the Northern most point. The remaining nest was located midway on the eastern slope of the sand spit between two wrack lines (Figure 3).

Observations on June 24, 1996, revealed 11 least terns loafing/courting and five active nests. However, 59 least terns and two royal terns were observed to the north of the island on a sandbar which is exposed during spring tides.

During monitoring on 1 July, 40 least terns and three tended nests were observed (Table 4). It is believed that nest "A" hatched successfully by this date, when observations revealed one least tern landing near the vegetation

Table 1: Number of birds by species monthly data summary at Spoil Island SL4
* Number of birds by species loafing on Sand bar North of SL4

APRIL, 1996

	04/04/96	04/18/96	04/24/96	04/26/96	* 04/26/96	04/29/96	Total
Least Tern	0	10	0	6	14	6	36
Royal Tern	18	92	0	0	15	0	125
Forster's Tern	2	4	0	0	0	0	6
Caspian Tern	0	3	0	0	0	0	3
Oystercatcher	0	1	0	1	1	0	3
Sanderling	3	2	0	0	0	0	5
Wilson's Plover	0	6	0	0	0	0	6
Ruddy Turnstones	0	5	0	0	0	0	5
Brown Pelican	4	5	0	0	5	0	14
Double-crested cormorant	5	12	0	0	28	0	45
American Anhinga	2	2	0	0	0	0	4
Laughing Gulls	0	2	0	0	2	0	4

Table 2: Number of birds by species monthly data summary at Spoil Island SL4
* Number of birds by species loafing on Sand bar North of SL4

MAY, 1996

	05/01/96	* 05/01/96	05/12/96	* 05/16/96	05/19/96	* 05/19/96	05/23/96	05/25/96	05/28/96	* 05/28/96	Total
Least Tern	1	10	0	1	0	6	2	2	10	0	18
Royal Tern	0	42	0	10	0	0	0	0	0	46	52
Forster's Tern	0	0	0	0	0	0	0	0	0	0	0
Caspian Tern	0	0	0	0	0	0	0	0	0	0	0
Oystercatcher	0	0	0	0	0	0	0	0	0	0	0
Sanderling	3	0	0	0	0	0	0	0	0	0	3
Wilson's Plover	0	0	0	0	0	0	0	0	0	0	0
Ruddy Turnstones	0	15	0	0	0	0	0	0	0	0	15
Brown Pelican	0	6	0	0	0	1	0	0	0	0	7
Double-crested cormorant	0	16	0	0	0	9	0	1	0	0	25
American Anhinga	0	0	0	0	0	0	0	0	0	10	0
Laughing Gulls	0	0	0	0	0	0	0	0	0	0	0

Table 3: Number of birds by species monthly data summary at Spoil Island SL4
* Number of birds by species loafing on Sand bar North of SL4

JUNE, 1996

	06/04/96	06/13/96	06/14/96	06/22/96	06/24/96	* 06/24/96	Total
Nests	1	1	1	5	5	0	
Least Tern	35	38	30	48	11	59	221
Least Tern Fledglings	0	5	2	5	0	0	
Royal Tern	3	0	1	0	0	2	6
Forster's Tern	0	0	0	0	0	0	0
Caspian Tern	0	0	0	0	0	0	0
Oystercatcher	0	0	0	0	0	0	0
Sanderling	0	0	0	0	0	0	0
Wilson's Plover	0	0	0	0	0	0	0
Ruddy Turnstones	0	0	0	0	0	0	0
Brown Pelican	0	1	4	0	4	0	9
Double-crested cormorant	0	4	1	0	0	0	5
American Anhinga	0	0	0	0	0	0	0
Laughing Gulls	1	0	0	0	0	0	1

Table 4: Number of birds by species monthly data summary at Spoil Island SL4
* Number of birds by species loafing on Sand bar North of SL4

JULY, 1996

	07/01/96	07/04/96	07/05/96	07/06/96	Total
Nests	3	5	4	5	
Least Tern	40	115	20	65	240
Least Tern Fledgling	5	10	0	0	
Royal Tern	0	0	0	0	0
Forster's Tern	0	0	0	0	0
Caspian Tern	0	0	0	0	0
Oystercatcher	1	4	0	2	7
Sanderling	0	0	0	0	0
Wilson's Plover	0	0	0	0	0
Ruddy Turnstones	0	0	0	0	0
Brown Pelican	13	9	3	3	28
Double-crested cormorant	4	0	2	1	7
American Anhinga	0	0	0	0	0
Laughing Gulls	0	0	0	0	0

with a bay anchovy (*Anchoa mitchilli*) and immediately leaving that area 9 (Figure 3). Due to various equipment shortfalls and limited approach distances, sighting the chicks was not possible. The eggs hatched within the expected incubation period of 20 to 22 days.

As of observations on 4 July 1996, the remaining nests had not hatched, though it seemed one nest, "D", may have been lost due to high tides, or hatched. However, two additional nests were observed. There were 115 least terns sighted on the island, the highest density this season. Nine brown pelicans, four oystercatcher (*Haematopus palliatus*), as well as 10 least tern fledglings were sighted (Table 4). This increase in numbers may be attributed to recreational boater activities taking place on other spoil islands and loafing sites near the Fort Pierce Inlet. No human disturbance was observed during the four hour observation period.

Throughout the holiday weekend, volunteers policed the nesting sites, particularly SL4 because of the active nests. One boater was observed approaching the island on 6 July (Saturday) but promptly left when asked by the volunteer. Sixty-five least terns, two oystercatchers, one double crested cormorant and three brown pelicans were sighted.

SPOIL ISLAND SL19B

This island was used primarily as a loafing site. As of 4 July, no least terns were observed nesting. In April 1996, royal terns dominated the site though least terns were common, exhibited courtship behaviors, and seemed to be pair bonding (Table 5). It was very common to see a pair of loafing oystercatchers.

During the month of May, the number of least tern observed increased steadily (Table 6). However, royal tern populations dominated with a maximum of 68 observed in one day. For reasons unknown, 1 dead herring gull and 4 dead royal terns were found on the island; one of the royal terns was banded. (Band return data has not yet been received from

SPOIL ISLAND SL4

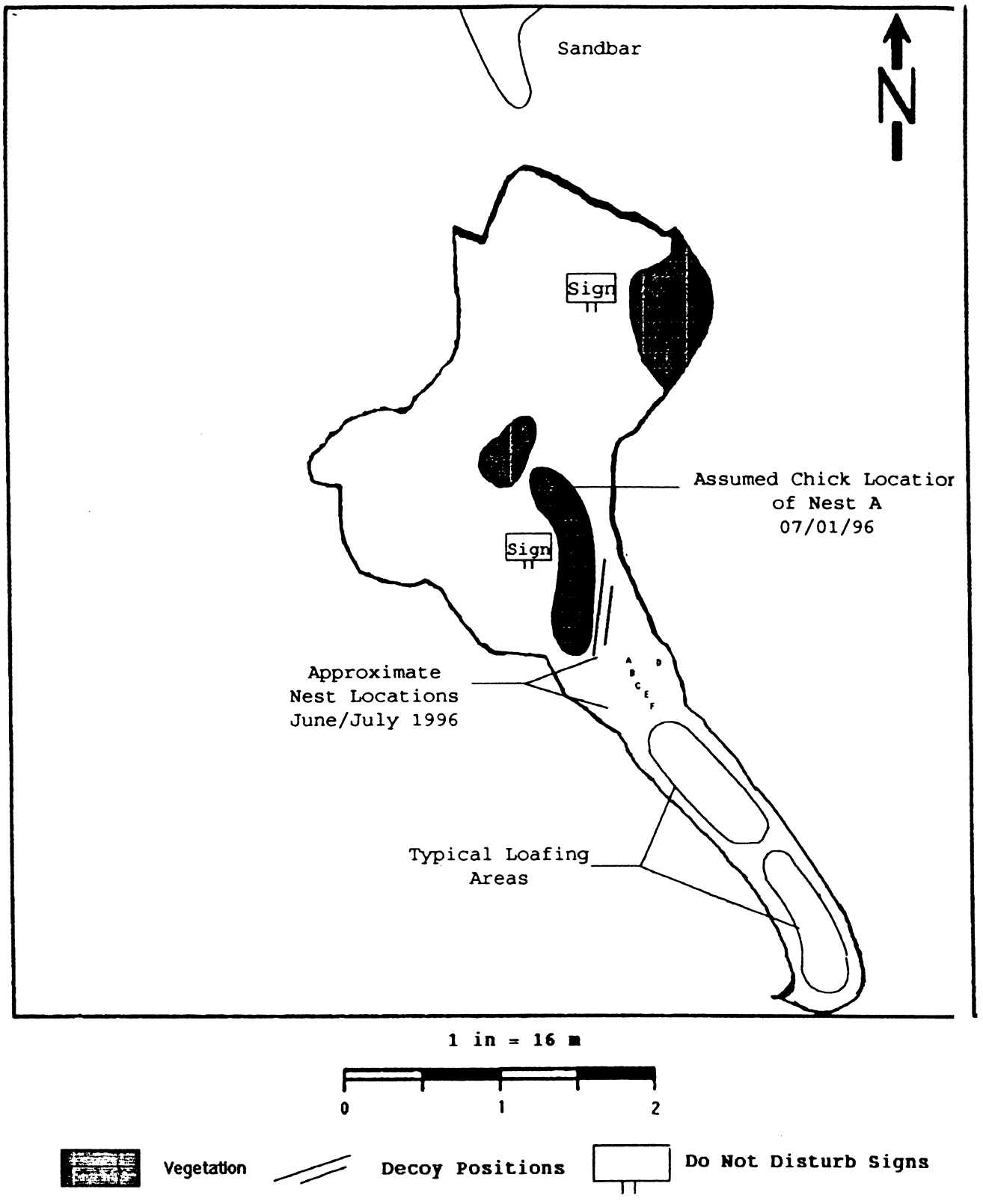

Figure 3. Spoil Island SL4 indicating nest locations, loafing areas and chick location.

USFWS.)

Three times during the month, disturbance by boaters was observed. Typically, the loafing and foraging birds would relocate to SL19A located North of the site or return when the boaters evacuated the area.

Similar to SL4, fledglings were observed utilizing SL19B in June. The greatest number of least terns (30) also were observed during June; however, their behavioral activity was confined to loafing and foraging. Disturbance by recreational fishermen was observed three times.

CONCLUSION

Nesting did not take place this season on SL19B as a result of several factors: by July, grass covered all of the decoys and the area's most suitable for nesting, disturbance by boaters was frequent, and vertical structure's on the island was a hinderance. However, the site offered a much needed avian loafing area in this part of the Indian River Lagoon.

At both sites, the number of least terns increased as the season progressed. Fledglings utilized both sites, and adult least terns loafed progressively higher on the island toward suitable nesting areas through the season.

The 26 April disturbance by vandals on SL4 was significant. Because least terns typically begin nesting around May 1st, the disturbance a few days prior was likely to deter nesting during the first wave.

RECOMMENDATIONS

Due to the frequency of human disturbance, further actions are warranted to deter access. Signs will be posted approximately 50m offshore to protect nesting shorebirds around the site provided they do not interfere with safe navigation (Rodgers and Smith, 1995). A new rule will likely be

Table 5: Number of birds by species monthly data summary at Spoil Island SL19B

APRIL, 1996

	04/12/96	04/14/96	04/18/96	04/20/96	04/24/96	Total
Least Tern	2	5	5	4	5	21
Royal Tern	13	12	20	40	27	112
Forster's Tern	0	0	0	0	0	0
Caspian Tern	0	0	0	0	0	0
Oystercatcher	2	0	2	2	0	6
Sanderling	0	0	0	0	0	0
Wilson's Plover	0	0	0	0	0	0
Ruddy Turnstones	0	0	0	0	0	0
Brown Pelican	18	4	23	6	0	51
Double-crested cormorant	3	2	0	2	0	7
American Anhinga	0	0	0	0	0	0
Laughing Gulls	0	0	0	0	0	0
Herring Gull	0	0	1	0	0	1

Table 6: Number of birds by species monthly data summary at Spoil Island SL19B

MAY, 1996

	05/01/96	05/04/96	05/12/96	05/19/96	05/23/96	05/27/96	05/30/96	Total
Least Tern	5	10	30	20	15	0	20	100
Royal Tern	68	40	0	25	26	0	10	169
Forster's Tern	0	0	0	0	0	0	0	0
Caspian Tern	0	0	0	0	0	0	0	0
Oystercatcher	0	2	0	2	2	1	0	7
Sanderling	0	0	20	0	0	0	0	20
Wilson's Plover	0	0	0	0	0	0	0	0
Ruddy Turnstones	0	0	0	2	0	0	6	8
Brown Pelican	5	4	0	1	3	0	0	13
Double-crested cormorant	0	4	0	0	0	0	0	4
American Anhinga	0	0	0	0	0	0	0	0
Laughing Gulls	0	4	0	0	0	0	0	4
Herring Gull	1	2	0	0	0	0	0	3

adopted by the Florida Legislature in the fall of 1996 which allows the FDEP to deny access to FDEP managed lands to protect sensitive resources. Spoil islands fall under this criteria.

Recognizing the needs of recreational boaters, SL3 will be enhanced before the 1997 nesting season. Some fringing Australian Pine and Brazilian Pepper will be removed or thinned to open a greater beach area. Fire rings and a picnic table will be placed on the island. More importantly, an informational sign will be posted explaining why the island was enhanced, brief life history information about least terns, and why it is imperative for the public to refrain from using SL4 on a seasonal basis.

Final enhancement of SL19A will begin once lower humidity levels allow for a successful burn. The island will be prepared in the same manner as SL4 and SL19B.

Minimizing the vertical structure on SL19B is a priority. Thus, to reduce the stumps, a stump grinder will be employed. The grass will not be removed until February 1997 to aid with stabilizing sediment during winter storms. At that time, the grass will be removed and the area treated with a herbicide.

Aggressive volunteer recruitment will take place during the fall and winter to secure a group to assist with monitoring the sites, particularly on weekends. Contacts to local boating and environmental groups have been made.

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ESTUARINE RECONNECTION OF A SALT MARSH CREEK: EFFECTS ON MACROFAUNA AND PHYSICAL PARAMETERS

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ABSTRACT

Lost Creek (Volusia County, Florida), a previously isolated marsh creek, was reconnected to tidal influences by rotary ditches in an attempt to improve water quality and macrofaunal parameters within the system. Before-After, Control-Impact (BACI) analyses compared conditions at Lost Creek and a nearby control site (Ten Mile Creek) before and after rotary ditching. Total Kjeldal nitrogen decreased significantly in Lost Creek after reconnection; no other water quality parameters varied significantly. Macrofaunal (fish and crustaceans) and crustacean abundances changed signifanctly after reconnection. The mean difference in number of macrofaunal species did not change. Jaccard percent similarities increased significantly after reconnection.

INTRODUCTION

As of 1991, a total of 195 impoundments occupying approximately 16,000 ha of coastal wetlands were found in east-central Florida (Rey et al. 1991). Deterioration of water quality and a decline in the productivity of commercially important fishery species within estuarine systems are just

a few of the environmental impacts attributed to salt marsh impoundments (Rey 1985).

Basic water quality parameters obtained from closed impoundments generally differ from the values observed in open impoundments or natural marsh systems. Due to lack of tidal inputs, closed impoundment water levels are largely controlled by evaporation and precipitation which can result in broad salinity fluctuations. Gilmore et al. (1982) measured salinities as low as 2.0 ppt and as high as 200 ppt within a closed impoundment in contrast to a range of 25-38 ppt from an impoundment fitted with an open 80 cm diameter culvert. Water temperature, dissolved oxygen levels, and turbidity can also differ within an enclosed system since water circulation is controlled mainly by wind rather than tidal currents. Lack of tidal mixing within impoundments can also restrict import and export of nutrients resulting in both extreme short and long term oscillations (Rey and Crossman 1987).

Drastic differences have also been noted when comparing macrofaunal populations from closed impoundments and natural salt marshes (Harrington and Harrington 1982; Rey and Crossman 1987; Vose and Bell 1994). Natural salt marshes are utilized as nursery grounds by a variety of commercially and recreationally important fishery species (Bozeman and Dean 1980; Boesch and Turner 1984; Rozas and Hackney 1984), but due to limited access, this function is often restricted in impoundment systems (Zale et al. 1987). For example, the disappearance of 11 fish species, including 4 commercially important species, was reported after impoundment of an east-central Florida salt marsh (Harrington and Harrington 1982). Abundance of fish and decapod species can also differ significantly between impoundments and a natural marsh environment. Vose and Bell (1994) and Karlen (1993) documented significantly higher densities within isolated systems when compared to control sites.

Rotary ditching has recently been used in several Florida impoundments in an attempt to enhance impacted systems while maintaining adequate mosquito control. Rotary ditching follows the same basic principles of

Open Marsh Water Management (OMWM) by connecting isolated bodies of water to tidal influences and allowing for inundation and destruction of mosquito breeding depressions, improved water circulation, and year round access for organisms. Water quality and macrofaunal parameters were examined at an impacted creek recently reopened to tidal influences through rotary ditches and compared to conditions at a nearby control site. The above parameters were also examined prior to ditching in a baseline study (Karlen 1993). Comparison of the post-ditching and baseline data allowed for evaluation of the effectiveness of salt marsh restoration through rotary ditching.

STUDY SITE

Lost Creek (impacted site) and Ten Mile Creek (control site) are both located to the north of Ponce de Leon Inlet in Volusia County, Florida (Fig. 1). Sampling of water quality parameters and macrobenthic communities occurred monthly at 5 stations within each creek. Preceding isolation, Lost Creek (29° 05'N, 80° 57'W) provided drainage into the Halifax River for approximately 100 acres of marshland. This connection to the Halifax River was disconnected prior to 1943 by deposition of dredge spoil from maintenance or construction of the Intracoastal Waterway (Taylor and Cochrane 1991). In November 1992, the East Volusia County Mosquito Control District reestablished tidal flushing within Lost Creek through the installation of rotary ditches, approximately 2.5' x 3' in original size. Ten Mile Creek (29° 06'N, 80° 57'W), situated approximately 0.6 miles north of Lost Creek, was utilized as a tidally influenced control site. The Halifax River is connected to this system on the east end while Rose Bay is located at the west end. The distance between the two creek systems is small enough to allow for the same climatic influences, but separation by a large body of water (Rose Bay) diffuses any potential impact on the Ten Mile Creek reference environment from the altered hydrology of Lost Creek.

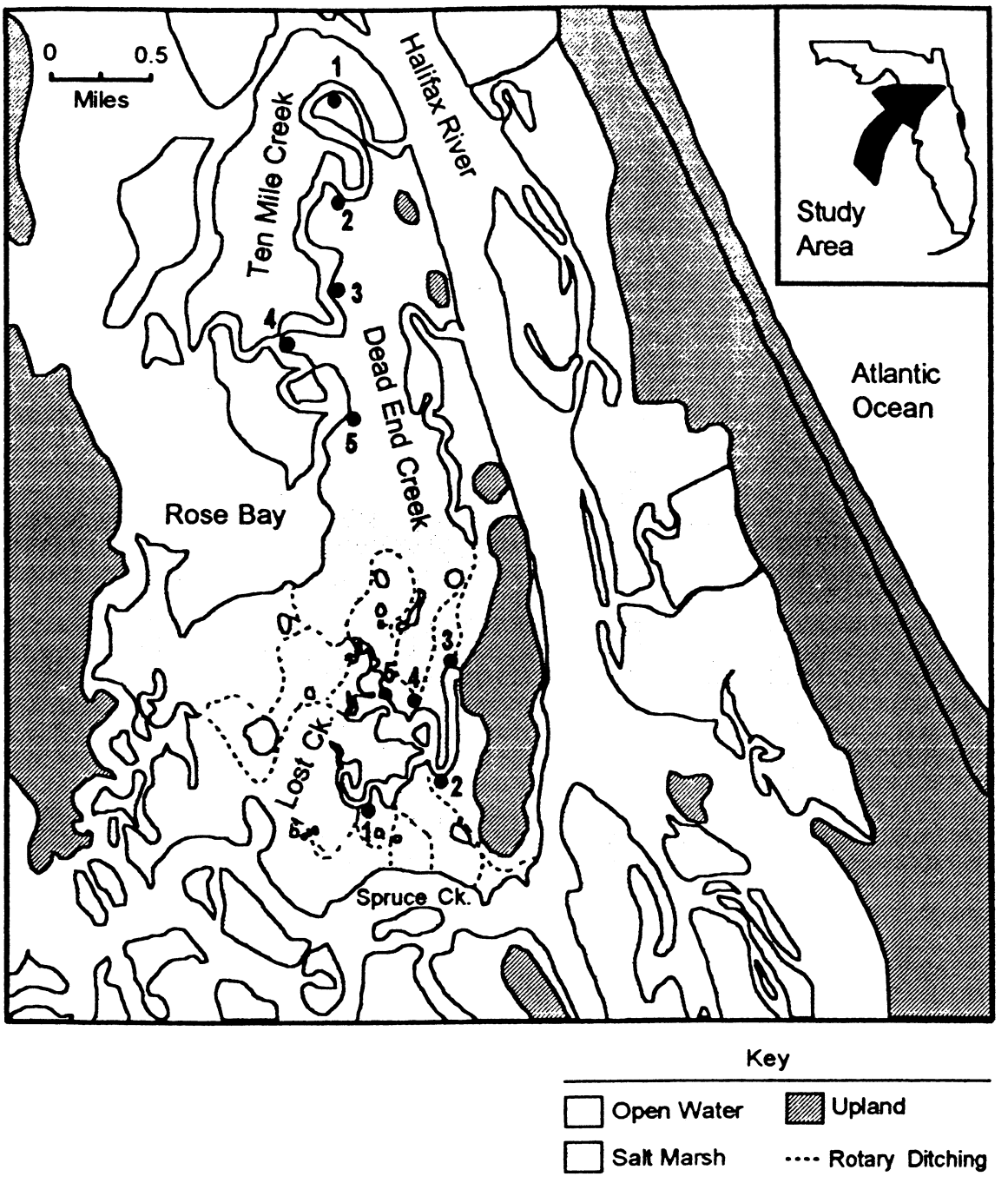

Figure 1. Location of sampling stations and rotary ditches (dark lines) at Lost Creek (impacted site) and Ten Mile Creek (control site).

MATERIALS AND METHODS

All pre-ditching data were compiled by Karlen (1993) between May and November 1992. Rotary ditches were installed at Lost Creek in November 1992. Post-ditching samples of water quality and macrofauna were collected from the five stations at both Lost Creek and Ten Mile Creek between September 1994 and June 1995.

Monthly water temperature, salinity and surface dissolved oxygen levels were measured from midchannel locations at all stations in both creek systems. Chlorophyll a, water color, turbidity, total suspended solids, nitrates-nitrites, total Kjeldahl nitrogen, total phosphate, and soluble ortho-phosphate were determined from four water samples collected at midchannel locations from each station. Volusia County Environmental Management Laboratory analyzed all water samples before and after rotary ditching of Lost Creek.

A 9.14m x 1.83m (30ft x 6ft) seine net equipped with a 1.83m x 1.83m (6ft x 6ft) center bag and 0.635cm (0.25in) Ace type mesh was pulled a distance of approximately 10m perpendicular to the creekbank in order to obtain monthly macrofaunal samples from each station. The collected organisms were placed in plastic containers and fixed in 7% formalin for a minimum period of 48 hours, rinsed with tap water, identified to the lowest possible taxon, enumerated and transferred to ethanol.

Before-After, Control-Impact (BACI) analyses (Stewart-Oaten et al. 1986; Faith et al. 1991) were used to detect significant changes in the relationship between Lost Creek and Ten Mile Creek with regard to water quality and macrofaunal parameters. A t-test was used to detect any significant change in pre-ditching and post-ditching differences of mean parameter values from Lost Creek and Ten Mile Creek. If the data failed a normality test, the t-test was replaced with a nonparametric Mann-Whitney Rank Sum test. A significant decrease in mean difference after estuarine reconnection for any given parameter indicates a convergence between Lost Creek and Ten Mile Creek.

Macrofaunal percent similarities between Lost Creek and Ten Mile Creek were calculated for each sampling date using the Jaccard Index of Similarity. A t-test was employed to compare pre-ditching and post-ditching percent similarities.

RESULTS

No significant changes were detected between pre-ditching and post-ditching differences between Lost Creek and Ten Mile Creek for water temperature, salinity, dissolved oxygen, chlorophyll a, turbidity, total suspended solids, total phosphate, soluble ortho-phosphate, and nitrate-nitrites (Table 1). However, the mean difference for all water quality parameters, with the exception of turbidity and total suspended solids, decreased after rotary ditching. Total Kjeldhal nitrogen (TKN) was the only water quality parameter to experience a significant change in difference between Lost Creek and Ten Mile Creek after rotary ditching.

Mean TKN values were higher at Lost Creek on every sampling date prior to and after rotary ditching (Fig. 2A). Ten Mile Creek levels remained relatively constant throughout both sampling periods, while Lost Creek experienced a reduction in range after rotary ditching. Comparison of the mean differences in TKN between the two creeks before and after tidal reconnection showed a significant change ($P=0.0025$), resulting from a decrease in the values observed at Lost Creek (Fig. 2B; Table 1).

Macrofaunal abundances for Lost Creek and Ten Mile Creek generally followed similar trends of increase and decrease both before and after rotary ditching. However, prior to estuarine reconnection, Lost Creek macrofaunal abundances were much higher than those seen at Ten Mile Creek (Fig. 3A).

Table 1. Mean difference (± 1 S.E.) between Lost Creek and Ten Mile Creek before and after rotary ditching and results of t-tests (t,P; α=0.05; df=15) or Mann-Whitney U-tests for various water quality parameters.

	Mean Difference (± 1 S.E.)			
Parameter	Before	After	t/U	P
Water temperature	0.7425 ± 0.811	-0.0289 ± 0.566	0.794	0.4393
Salinity	3.944 ± 1.58	-0.599 ± 2.17	1.69	0.1127
D.O.	1.040 ± 1.188	0.403 ± 0.675	0.466	0.6481
Chlorophyll a	24.282 ± 10.00	10.023 ± 2.37	61.0*	0.6126
Turbidity	3.910 ± 1.42	17.34 ± 12.57	68.0*	0.7363
TSS	7.330 ± 3.74	26.01 ± 9.06	55.0*	0.1124
Total phosphate	0.1611 ± 0.0634	0.0650 ± 0.0212	1.51	0.1517
Ortho-phosphate	-0.0101 ± 0.0291	-0.0011 ± 0.006	69.0*	0.1520
TKN	1.377 ± 0.254	0.445 ± 0.090	3.62	0.0025
No_x	-0.0785 + 0.0299	-0.0067 + 0.0189	70.0*	0.1206

* U-value for Mann-Whitney Rank Sum test.

After rotary ditching, the range in macrofaunal abundances at Lost Creek decreased and values became more similar to those observed at Ten Mile Creek. Differences in macrofaunal abundance before and after rotary ditching were significantly different (P=0.0024) (Fig. 3B), due to a reduction in the monthly values observed at Lost Creek.

The crustacean component of the macrofauna made up a large portion of the total abundance at both Lost Creek (47.5%) and Ten Mile Creek (37.2%) after tidal reconnection. Prior to rotary ditching, crustacean abundance at Lost Creek was consistently higher than Ten Mile Creek (Fig. 4A). Once tidal influences were reestablished, crustacean abundances at Lost Creek were often less than or nearly equal to the levels observed at

Ten Mile Creek. Differences for crustacean abundances before and after rotary ditching were significantly different (P=0.0113; Fig. 4B).

The mean number of macrofaunal taxa was higher at Lost Creek for every sampling date, even though a higher total number of species was collected from Ten Mile Creek (45 taxa at Lost Creek vs. 51 taxa at Ten Mile Creek; Fig. 5A). Species richness for the two sites generally followed the same trends in increase and decrease with the exception of the 20 September 1994 and 19 May 1995 sampling dates. There was no significant change (P=0.3149) in the pre-ditching and post-ditching differences for macrofaunal species richness (Fig. 5B).

After rotary ditching, a total of 38 macrofaunal species were found in both creek systems compared to only 17 species in common before ditching (Fig. 6A). Seven species were found only in Lost Creek and thirteen species were found only in Ten Mile Creek after ditching. Before tidal reconnection, 10 macrofaunal species occurred only at Lost Creek and 16 species occurred only at Ten Mile Creek. The mean monthly number of macrofaunal taxa common to both sites ranged from 6 species on 20 September 1994 to 16 species on 13 March 1995 during post-ditching samples. Jaccard percent similarities for macrofauna at Lost Creek and Ten Mile Creek differed significantly between pre-ditching and post-ditching values (P=0.0024; Fig. 6B). The mean percent similarity prior to rotary ditching was 24.9 $\pm$ 6.45% compared to a mean of 50.3 $\pm$ 3.73% after tidal reconnection of Lost Creek.

DISCUSSION

Rotary ditches were used to reconnect Lost Creek, a previously isolated system, to tidal influences. Through increased water circulation and faunal access, the ditches were expected to alter the physical and biological environment of Lost Creek, making it more similar to Ten Mile Creek, a natural salt marsh.

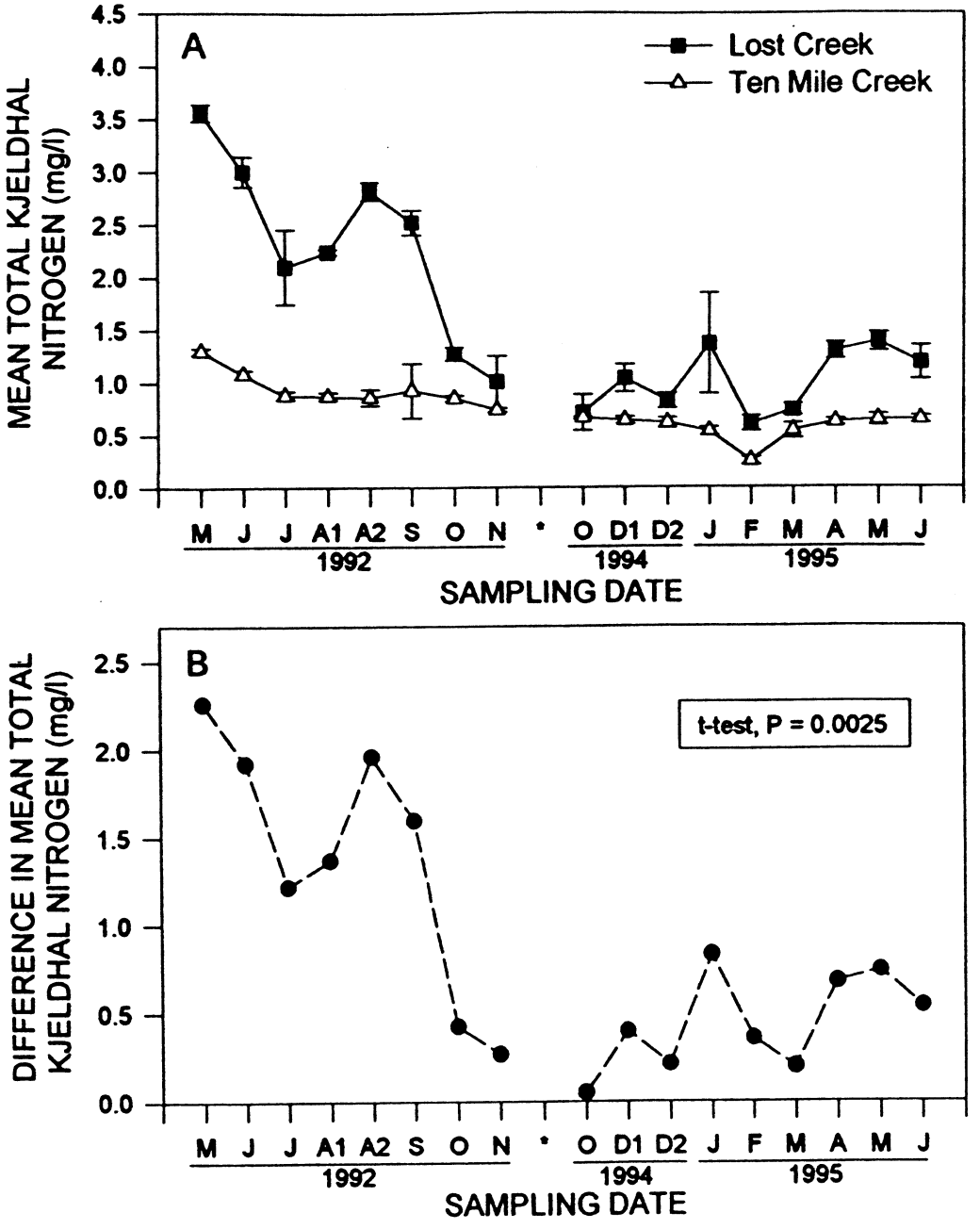

Figure 2. (A) Mean (± 1 S.E.) total Kjeldhal nitrogen and (B) difference in mean total Kjeldhal nitrogen, between Lost Creek and Ten Mile Creek before and after rotary ditching, May 1992 to June 1995 (* = rotary ditching, November 1992; pre-ditching data from Karlen 1993) (A1 = 4 August 1992, A2 = 31 August 1992, D1 = 2 December 1994 and D2 = 19,21 December 1994).

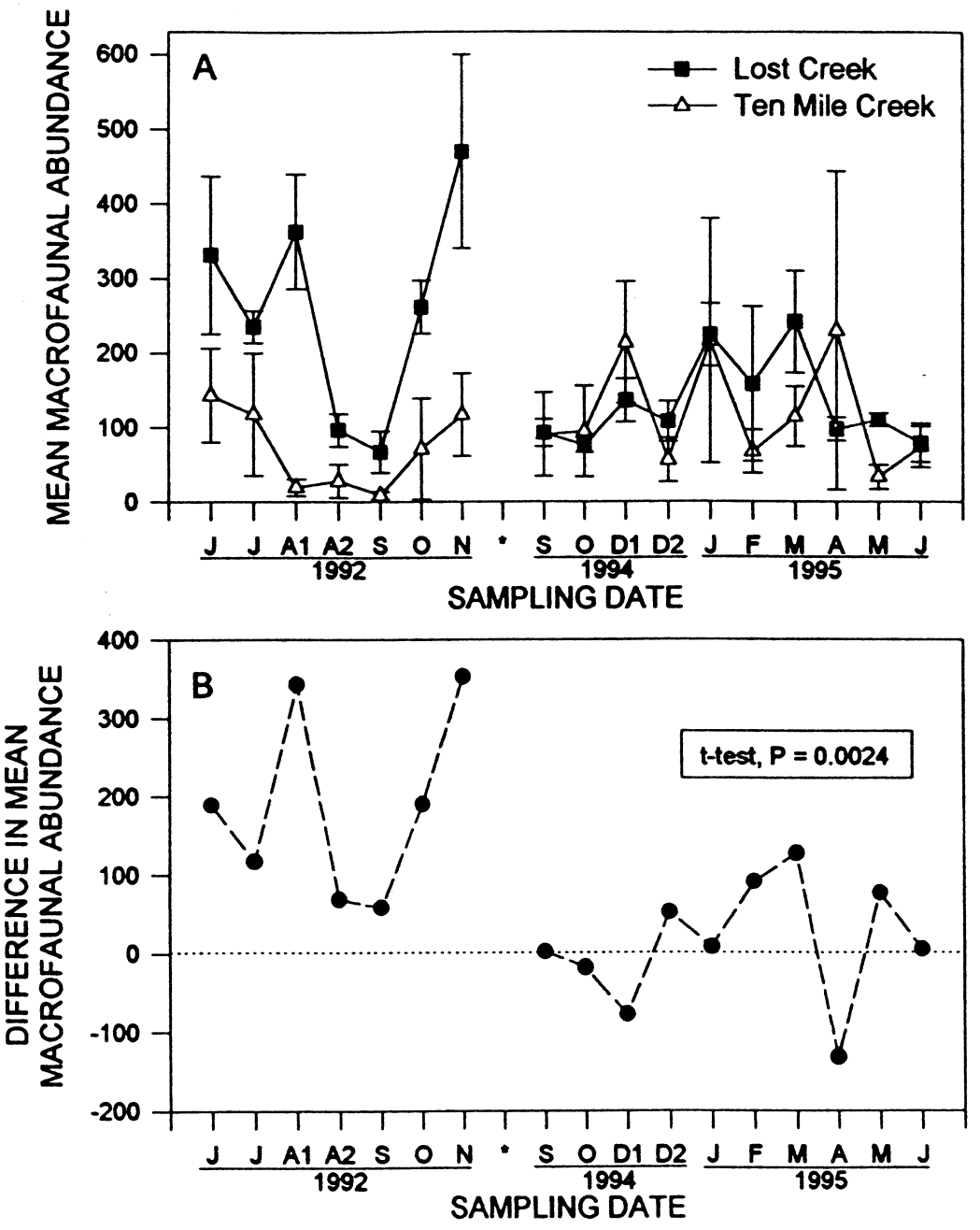

Figure 3. (A) Mean (± 1 S.E.) macrofaunal abundance and (B) difference in mean macrofaunal abundance, between Lost Creek and Ten Mile Creek before and after rotary ditching, June 1992 to June 1995 (* = rotary ditching, November 1992; pre-ditching data from Karlen 1993) (A1 = 4 August 1992, A2 = 31 August 1992, D1 = 2 December 1994 and D2 = 19,21 December 1994).

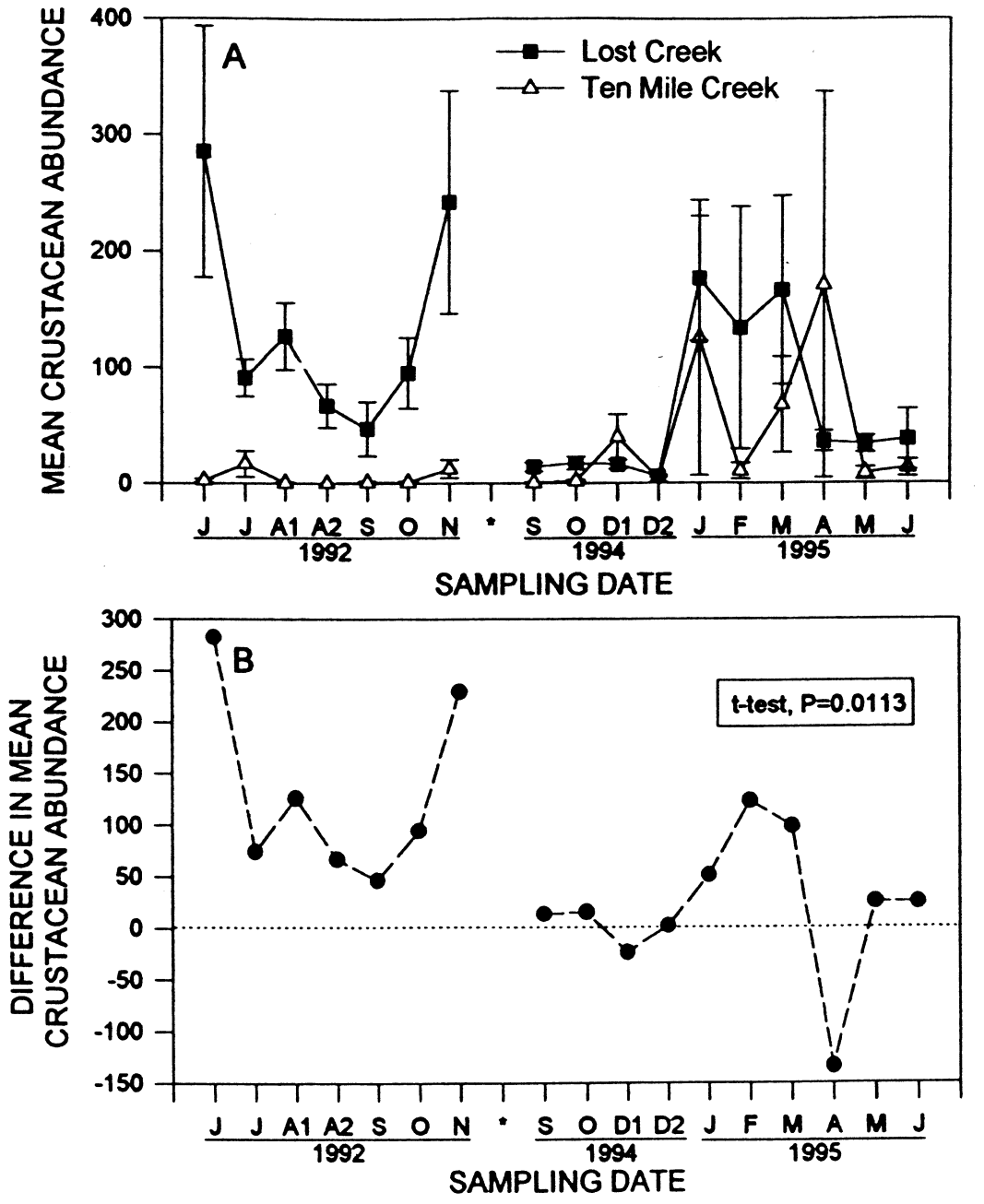

Figure 4. (A) Mean (± 1 S.E.) crustacean abundance and (B) difference in mean crustacean abundance, between Lost Creek and Ten Mile Creek before and after rotary ditching, June 1992 to June 1995 (* = rotary ditching, November 1992; pre-ditching data from Karlen 1993) (A1 = 4 August 1992, A2 = 31 August 1992, D1 = 2 December 1994 and D2 = 19,21 December 1994).

The mean difference in most water quality conditions did not change significantly after Lost Creek was reconnected to the estuary. The only water quality parameter which changed significantly was total Kjeldhal nitrogen (TKN). TKN was higher at Lost Creek than Ten Mile Creek on all sampling dates both before and after rotary ditching, but a significant decrease in mean monthly values at Lost Creek was seen after reconnection. Before tidal influences were reestablished within Lost Creek, nitrogen was probably imported into the system through runoff of residual water from the highmarsh, rainfall (Whiting et al. 1989), and leachates from submerged vegetative material (Turner 1993) and allowed to accumulate due to the absence of tidal flushing. While the amount of runoff and rainfall to Lost Creek would not be affected by rotary ditching, tidal removal of submerged vegetative material, which has a leachate release rate of 15 times greater than above ground detritus (Turner 1993), could account for a significant difference in nitrogen concentrations.

With the exception of turbidity and total suspended solids, all water quality parameters showed a decrease in the mean difference after ditching. The elimination of extreme values of physical parameters in Lost Creek may be particularly important for the survival of many benthic and macrofaunal species (Gamble et al. 1990). During the pre-ditching study, algal blooms, resulting from high nutrient levels, created wide dissolved oxygen fluctuations and subsequent fish kills in Lost Creek on two separate sampling dates (31 August and 15 September 1992) (Karlen 1993). Due to the 3 to 4 week sampling intervals, the occurrence of similar short term extremes in any given water quality parameter may have been underestimated at Lost Creek. However, no fish kills were observed during the post-ditching study, suggesting that tidal flushing through rotary ditches has created conditions which are more suitable for many marsh organisms.

An increase in the mean difference between Lost Creek and Ten Mile Creek was observed for turbidity and total suspended solids after tidal reconnection. Increased water circulation through the rotary ditches is probably resuspending the fine sediment materials within Lost Creek,

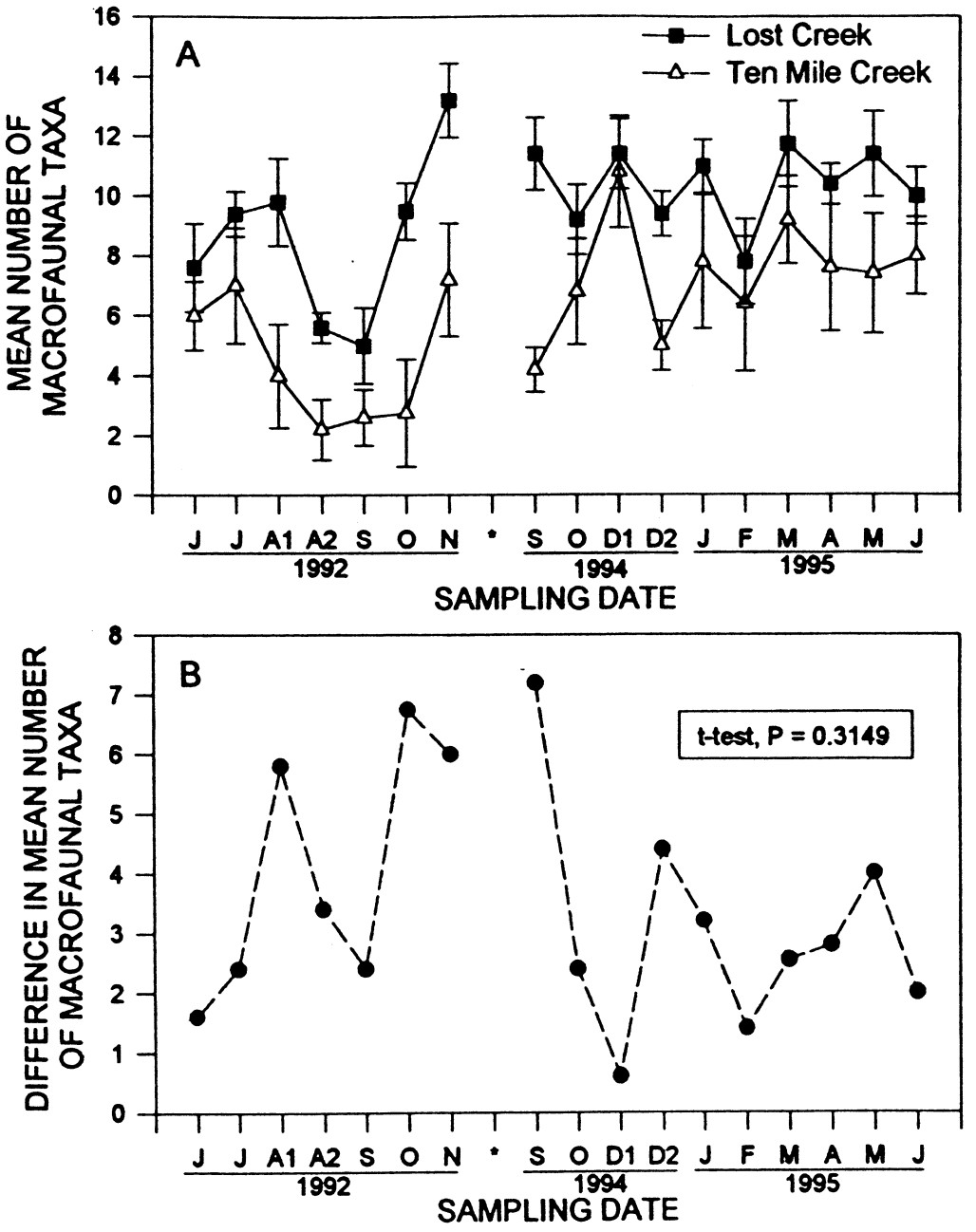

Figure 5. (A) Mean (± 1 S.E.) number of macrofaunal taxa and (B) difference in mean number of macrofaunal taxa, between Lost Creek and Ten Mile Creek before and after rotary ditching, June 1992 to June 1995 (* = rotary ditching, November 1992; pre-ditching data from Karlen 1993) (A1 = 4 August 1992, A2 = 31 August 1992, D1 = 2 December 1994 and D2 = 19,21 December 1994).

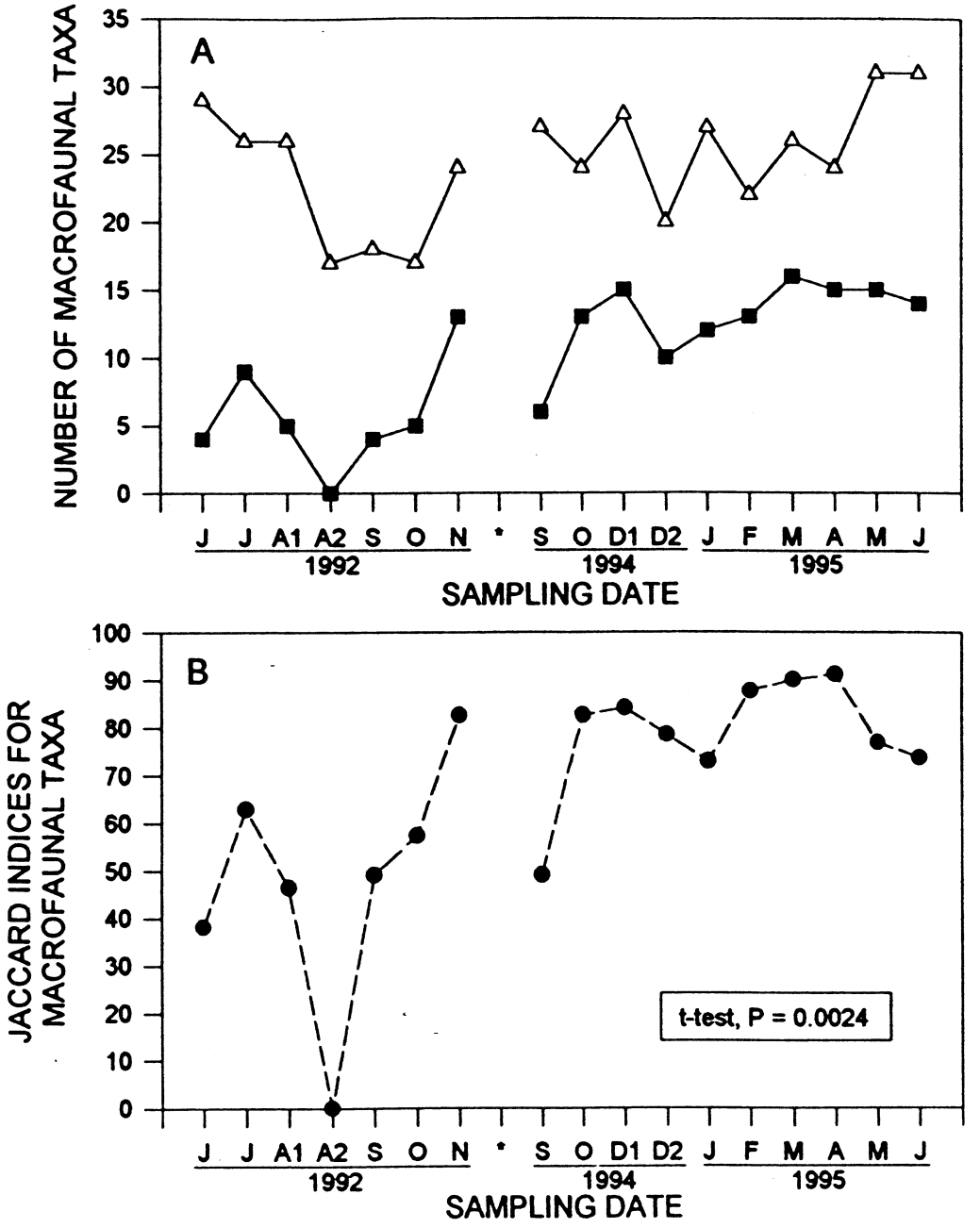

Figure 6. (A) Comparison of total number of macrofaunal taxa and the number of macrofaunal taxa common to both sites and (B) Jaccard Similarity index for macrofauna, between Lost Creek and Ten Mile Creek before and after rotary ditching, June 1992 to June 1995 (* = rotary ditching, November 1992; pre-ditching data from Karlen 1993) (A1 = 4 August 1992, A2 = 31 August 1992, D1 = 2 December 1994 and D2 = 19,21 December 1994).

resulting in wider extremes of turbidity and total suspended solids. Through erosion of the ditch banks and the resultant increase in tidal exchange, levels of turbidity and total suspended solids within Lost Creek should gradually diminish as the majority of fine sediments are swept out of the system.

During the course of this study, the volume of water exchanged through rotary ditches may not have been large enough to provide adequate circulation for the improvement of water quality conditions within Lost Creek. Towards the end of the post-ditching study (May and June 1995), certain ditches had become noticeably wider through erosion of the marsh bank. Water quality conditions at Lost Creek are expected to become more similar to those at Ten Mile Creek as the banks of the ditches continue to erode and a larger volume of water is exchanged between Lost Creek and the connected estuary.

A significant change was observed for mean difference of macrofaunal abundance (fish and crustaceans) between Lost Creek and Ten Mile Creek, resulting from a decrease in the monthly abundances observed at Lost Creek after tidal reconnection. Vose and Bell (1994) have documented a similar occurrence in an Upper Tampa Bay salt marsh. Upon breaching of a previously impounded area, they recorded a decrease in fish densities of approximately 80%. For macrofaunal populations in enclosed systems, abundances can increase and be maintained due to low predation risks (Rey and Crossman 1987; Zale et al. 1987). Unlimited macrofaunal access to Lost Creek through rotary ditches allowed for increased emigration of residents and may have increased immigration of predators, both resulting in lower numbers of individuals within the system and increased similarity to a natural salt marsh creek.

When the macrocrustacean component of the macrofauna was examined independently, mean difference in abundances differed significantly before and after rotary ditching. This indicates that the change in the total macrofaunal abundances was due primarily to a decrease in mean monthly crustacean abundances at Lost Creek after reconnection. Once again,

movement of motile macrocrustaceans out of Lost Creek through the rotary ditches and the invasion of predatory fish into the system are most likely the cause for the decrease in crustacean abundances.

Lost Creek and Ten Mile Creek showed similar seasonal trends in the mean number of macrofaunal taxa, although Lost Creek values were consistently higher in both pre-ditching and post-ditching studies. Collection efficiency may account for the consistently lower mean monthly number of taxa found at Ten Mile Creek. Seine efficiencies, which are dependent upon species, season and physical conditions (Allen et al. 1992), were probably lower at Ten Mile Creek both before and after ditching due to strong tidal currents, variable depths, and the presence of oyster reefs.

With the tidal reopening of a marsh creek, numbers of species would generally be expected to increase significantly since access is no longer restricted (Gilmore et al. 1982). The original conditions of Lost Creek prior to ditching may account for the absence of this result. Monthly mean numbers of macrofaunal species were higher at Lost Creek than at Ten Mile Creek even before tidal reconnection was established. High macrofaunal diversity is generally unusual in an impoundment which is completely isolated (Gilmore et al. 1982; Harrington and Harrington 1982; Rey and Crossman 1987; Zale et al. 1987). At Lost Creek, which was not a true impoundment, the flooding of the high marsh during spring tides allowed periodic macrofaunal recruitment into the system. The total number of taxa collected from Lost Creek after ditches were constructed did increase from the numbers recorded in the baseline study (30 before, 45 after). The higher number of species seen after reconnection could be a result of increased access through the ditches or simply a sample bias due to a larger number of sampling excursions.

After installation of rotary ditches in the Lost Creek system, the Jaccard index showed an increase in the percent similarity of macrofaunal species at Lost Creek to the natural marsh environment at Ten Mile Creek. This is a result of an increase in the number of species common to both systems, again resulting from the increased macrofaunal access to Lost Creek

through rotary ditches. Vose and Bell (1994) also saw higher levels of percent similarity to a natural marsh, from 0-53% to 0-73%, after berm removal of an impounded area in Upper Tampa Bay, Florida, but the increase was not statistically significant.

While this study detected a convergence between the impacted and control site for TKN, macrofaunal abundance, and percent similarity, rotary ditches have continued to increase in width due to erosion, indicating that the Lost Creek habitat may still be responding to tidal reconnection. The final equilibrium state which may be reached at Lost Creek as tidal influences stabilize will most likely not resemble the precise conditions of Ten Mile Creek. Since Lost Creek and Ten Mile Creek are not identical in sedimentary environment, location, and hydrology, it would be expected that the Lost Creek environment will eventually reach a stable state which is intermediate between the conditions that existed prior to estuarine reconnection and those seen at Ten Mile Creek.

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CASE STUDY OF THE ECOSYSTEM MANAGEMENT DEVELOPMENT IN THE BRETON SOUND ESTUARY, LOUISIANA.

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ABSTRACT

The Breton Sound Estuary in Louisiana, an area 269,503 km^2 in extent, annually loses an average of 408 km^2 (1,024 acres) of marsh. Major hydrologic changes have resulted from the construction of flood-control levees along the Mississippi River in the 1920's, the Mississippi River Gulf Outlet (a major navigation canal) in 1963, and a labyrinth of smaller canals for oil and gas extraction. The estuary was, to a great extent, decoupled from alluvial water, sediment and nutrients; then extensively connected to marine processes. The most immediate impact of these actions was a landward shift in salinity, followed by an inland shift in oyster production, and land loss. Several freshwater diversion projects have been implemented to offset these negative impacts. The environmental administration of the area is now moving towards ecosystem-level management. The authors discuss this real-world case study.

INTRODUCTION

The Breton Sound Estuary is located in southeast Louisiana and encompasses approximately 269,503 km^2 (676,400 acres) in Plaquemines and St. Bernard parishes (figure 1) . The estuary is bounded on the west by the Mississippi River, on the east by the south-west bank of the Mississippi

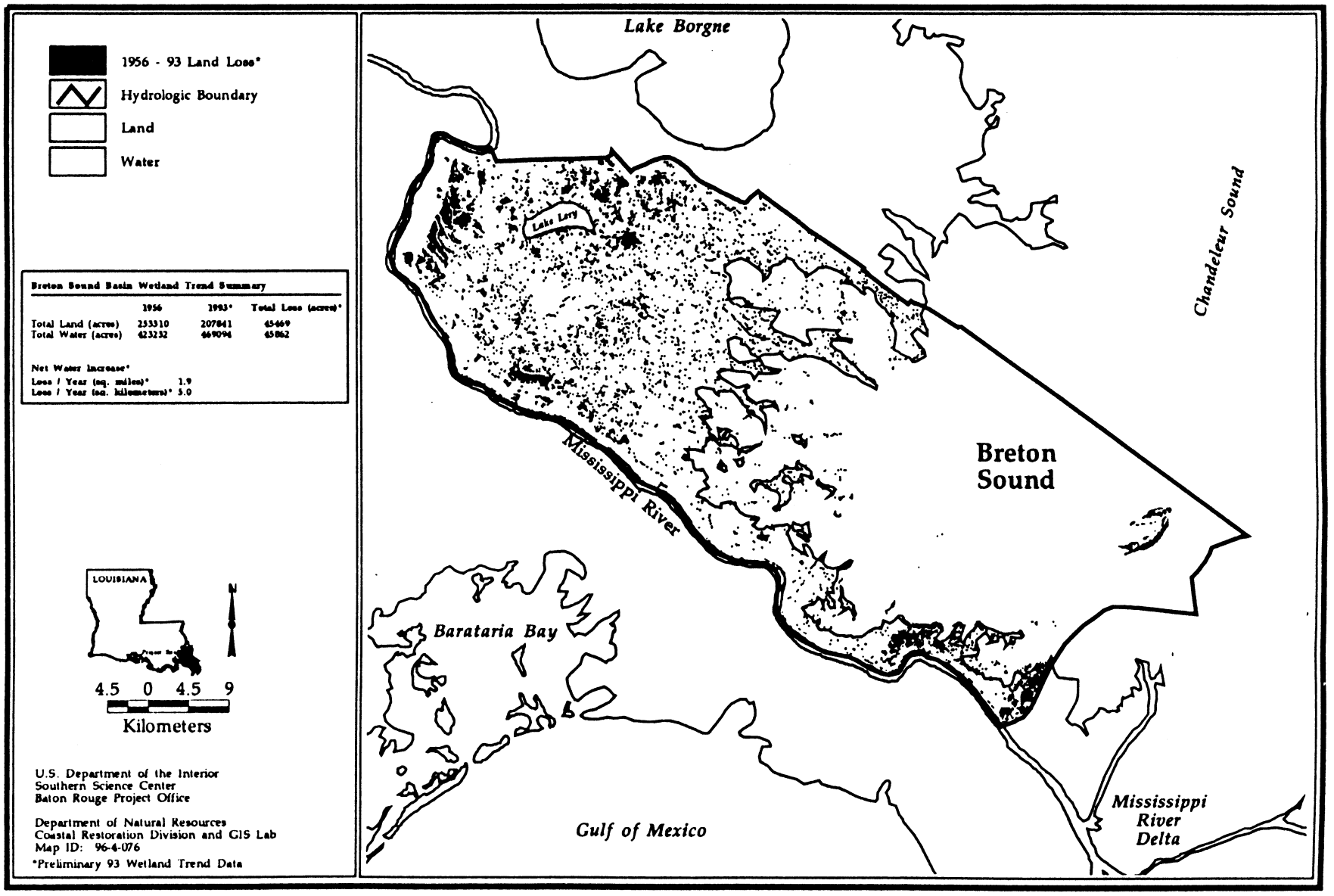

Figure 1. Breton Sound Estuary 1956-93 Wetland Trend Data

River Gulf Outlet (MRGO) , on the north by Bayou La Loutre, and on the south by Baptiste Collette Bayou and Breton Island.

Since 1932, 18,105 km^2 (45,440 acres)of coastal wetlands have been lost within the Breton Sound Estuary. The fresh, upper end has suffered the greatest net loss (Chabreck and Linscombe, 1982) as freshwater marshes ultimately give way to a more salt-tolerant community or deteriorate into open water (McKee and Mendelssohn 1989). Loss of fresh marsh has resulted in a truncated salinity gradient in many areas and a corresponding decrease in biodiversity associated with land loss and less diverse saltwater communities. Current projections indicate that during the next 50 years, another 13,260 square kilometers (33,280 acres) will disappear unless preventative measures are implemented (LCWCRTF 1993).

The primary causes of this land loss are sediment deprivation and saltwater intrusion which have resulted from the construction of flood control levees along the Mississippi River, navigation projects in the region, and canals associated with mineral extraction projects (Louisiana Wetland Protection Panel 1987, Gagliano and wicker 1989, Ramsey and Moslow 1987, Sasser et al. 1986, Templet 1987, Turner and Cahoon 1988). Because the sediment and freshwater deficit are the leading causes of marsh breakup in Breton Sound, effective management of the Breton Sound Estuary requires reintroduction of alluvial freshwater and sediment(Gagliano and van Beek 1976, LCWCRTF 1993).

A major impetus for freshwater diversions into the area was the loss in over 50,000 acres of oyster supporting bottoms in Breton Sound, presumably as a result of saltwater intrusion and the concomitant increase in oyster losses due to predation and disease (Pollard 1973). A case in point was the Bayou Lamoque Freshwater Control Structure (figure 2). The result of this diversion was a 300% increase in oyster production of the area and a 200% increase in harvested yield (Bowman et al. 1995). The original structure, built in 1957, was ultimately deemed to provide insufficient amounts of water, leading to the enlargement of the structure in 1977, which increased its capacity to approximately 340 cms(12,000

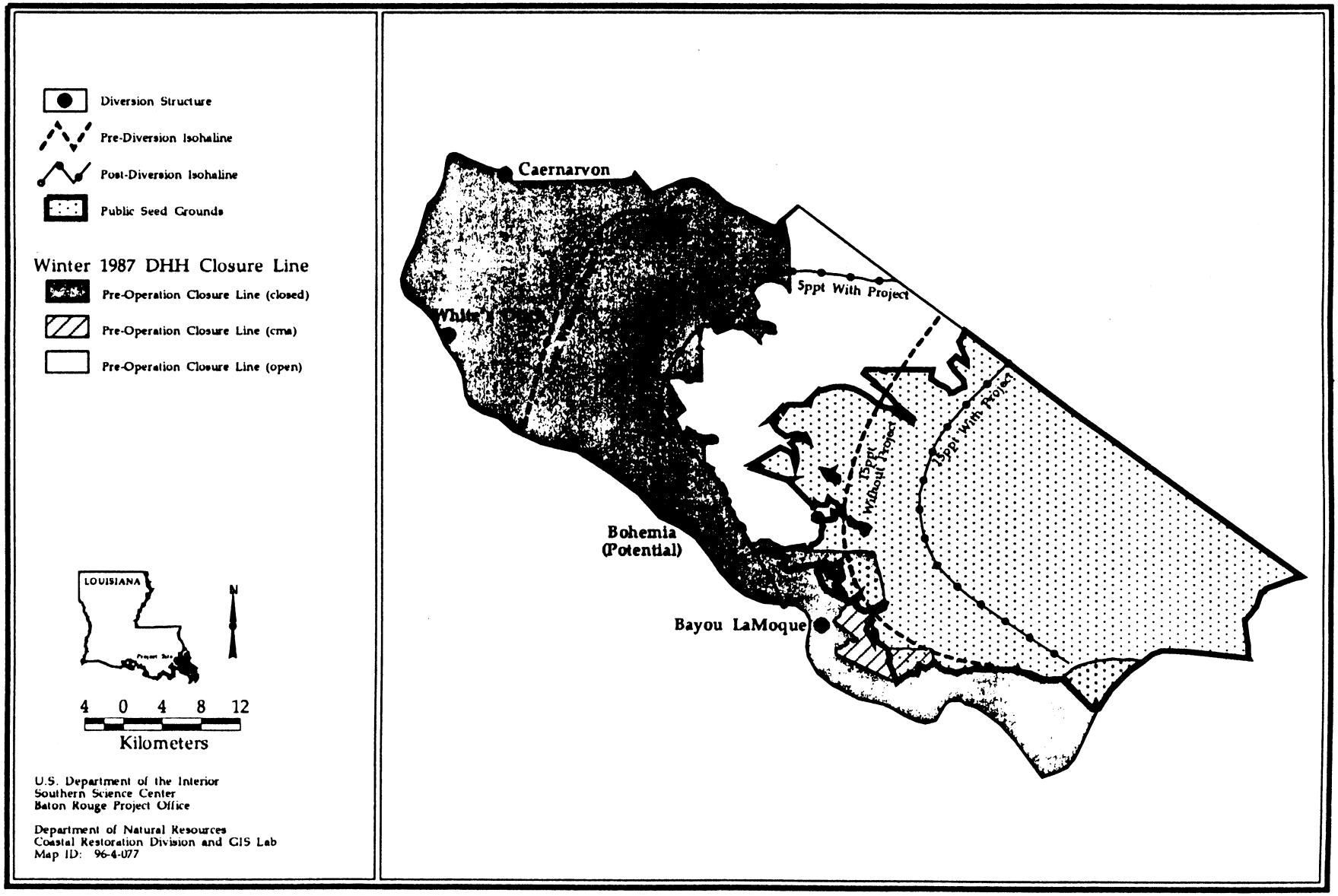

Figure 2. Breton Sound Estuary freshwater diversions, public oyster seed grounds, and closure lines. Isohalines are for pre- and post- diversion operation at the Caernarvon structure.

cfs).

The White's Ditch Diversion (figure 2) has been in operation since 1975 and was intended to benefit muskrat populations in the vicinity by stimulating reversion of brackish marsh into intermediate marsh. Its capacity is 250 cfs. As an example of sediment contribution by a freshwater diversion project, in 1990, it added 70,338,000 M^3 (57,000 acre-feet) of freshwater and 11,793,600 kg (13,000 tons) of riverine sediment to Breton Sound Basin (Roberts et al. 1991).

In the early 1960's, the Louisiana Department of Wildlife and Fisheries (LDWF) along with Plaquemines Parish Government requested the USACE to construct the Caernarvon freshwater diversion project (figure 2) to enhance oyster productivity on the Breton Public Seed Grounds.

The operational objectives of the Caernarvon structure were later expanded. According to the USACE (1966), "The structure is designed to introduce freshwater and alluvial material from the Mississippi River to the Breton Sound Estuary to help deter saltwater intrusion and establish favorable salinity conditions in the area. salinities would be managed to reduce saltwater intrusion, enhance growth of vegetation, reduce land loss, and increase production of commercial and recreational fish and wildlife." It was anticipated that these objectives would be achieved by increasing primary productivity and biodiversity, decreasing marsh loss, which would in turn result in an increase in the abundance of commercial and recreational fisheries, furbearers, and other wildlife.

A 1984 Environmental Impact Statement conducted by the USACE anticipated that $6,054,000 of average annual benefits to commercial fish and wildlife production and recreation (figure 3) would result from the Caernarvon diversion (USACE 1986). The 1984 EIS projected that as a result of the Caernarvon diversion, oyster productivity would be shifted to an enlarged area delimited by the projected five and fifteen parts-per thousand isohaline contours (figure 2). Although benefits were anticipated for numerous species, increased oyster productivity represented the

majority of predicted benefits resulting from operation of the structure, accounting for $5,416,000 of these benefits (figure 3).

The Caernarvon freshwater diversion structure began operations in 1991 and can deliver flows exceeding 227 cms (8,000 cfs).

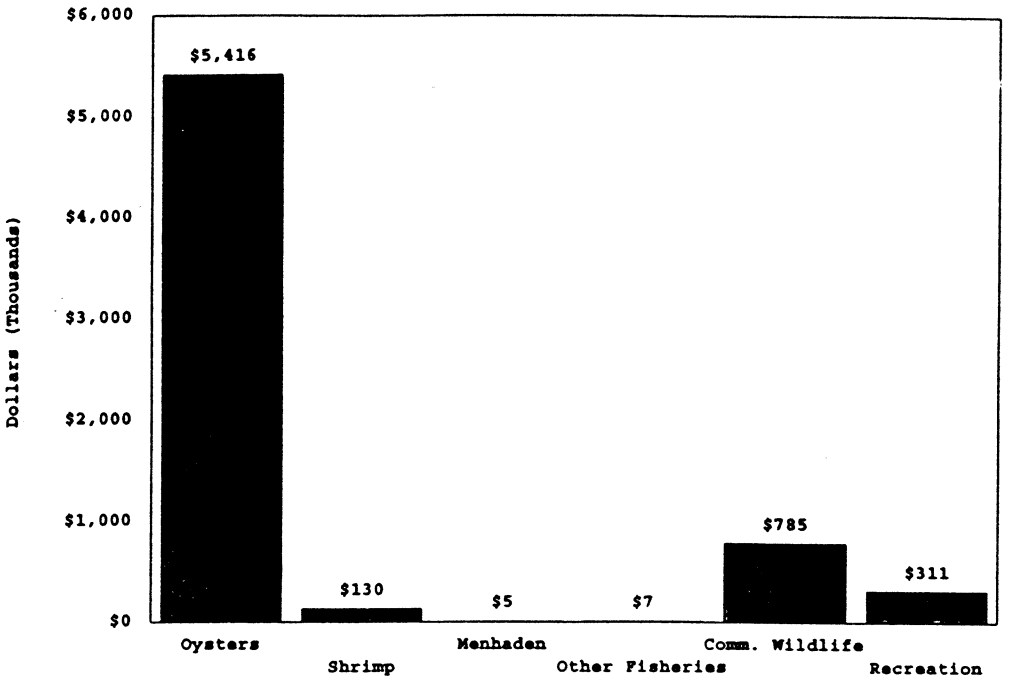

Figure 3. Summary of average annual economic benefits at Caernarvon project area. Estimates from 1984 USACE EIS assuming 8 1/8 % discount rate and 1983 prices.

Caernarvon Data

Much of the data collected in the Breton Sound Estuary to date is in connection with operations of the Caernarvon Diversion Structure. Of particular concern are commercially important fishery species and furbearers. vegetation composition is also of interest since this reflects any habitat shifts resulting from implementation of freshwater diversions. The impact of freshwater diversions like Caernarvon on these resources is indicated by comparing the abundance of these resources before and after diversion operation.

The production potential of oysters in the Breton Sound Estuary has been

assessed by the LDWF since the early 1970's (figure 4). From 1988-94, wildlife observations in the Caernarvon outfall area have been assessed by counting the number of alligator nests and muskrats before (1988-1990) and after (1990-94) operation of the Caernarvon structure (Geaghan 1995). Pre- and post-operational sampling for various fish plant and species were also conducted (figures 5, 6, and 7).

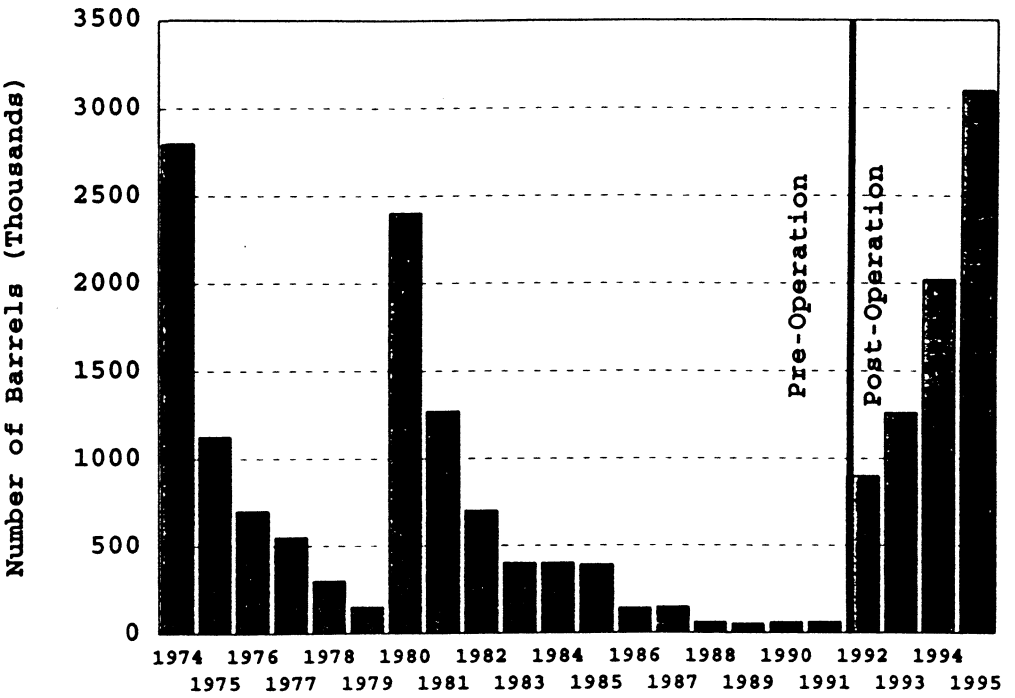

Figure 4. Oysters available on Breton Sound public oyster seed grounds (Bowman 1995).

OBSERVATIONS

Oyster, finfish, shellfish, and other wildlife have exhibited increases when comparing Caernarvon pre-operational and post-operational time periods. In 1991, the Caernarvon freshwater diversion began operating and lowering salinities in the outfall area. Oyster populations on the public grounds in Breton Sound began to rise the following year (figure 4). Also during 1991, the area experienced more than 113 inches of rainfall and extremely high river stages; therefore, it is difficult to assess the actual impact that the Caernarvon structure may have had on the increased oyster production potential observed during 1992. However, these populations have continued to rise each year since Caernarvon operation -a pattern

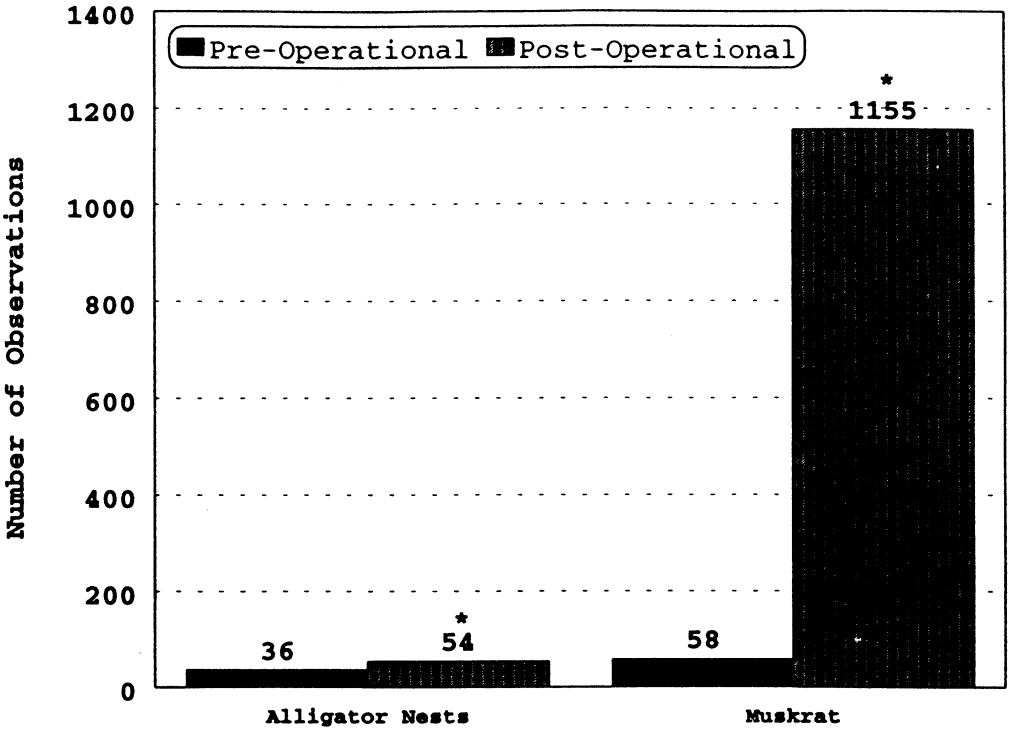

Figure 5. Wildlife Observations at Caernarvon. Observations during pre- (1988-90) and post- (1991-94) periods (Geaghan 1995). Asterisk indicates significant increase at 5% level.

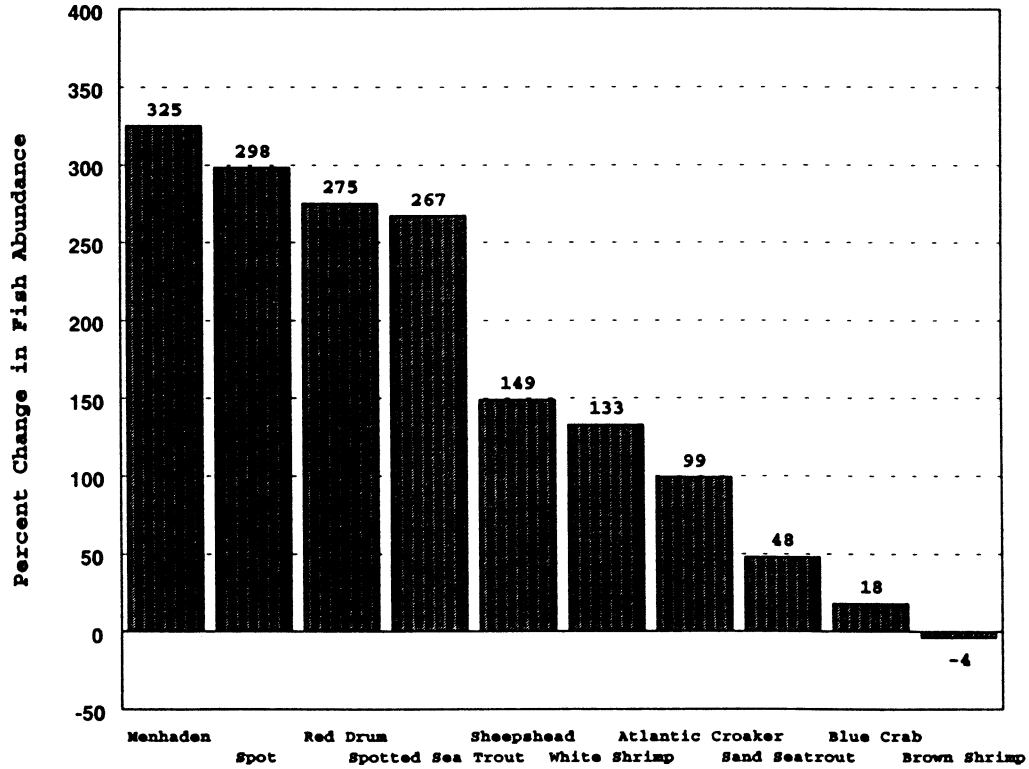

Figure 6. Percent change in finfish and shellfish at Caernarvon Area. Change between pre- (1988-90) and post- (1991-94) periods (Geaghan 1995). Abundance based on 6-ft trawl sampling method.

which had not been observed prior to operation of the freshwater diversion. Results from data collected by LDWF (Geaghan 1995) from 1988-1994 indicate increases in both finfish, shellfish, and wildlife abundance in the project area (figures 5 and 6).

Due to the absence of an adequate control site, the increases mentioned above cannot be attributed solely to the effects of the Caernarvon Project. However, we feel that these trends are at least in part due to a return towards estuarine conditions more aptly suited for these wildlife resources. Not surprisingly, a decreased abundance of salt marsh species and an increased abundance of fresh marsh species were observed when comparing the pre-operational and postoperational periods of the Caernarvon freshwater diversion (figure 7).

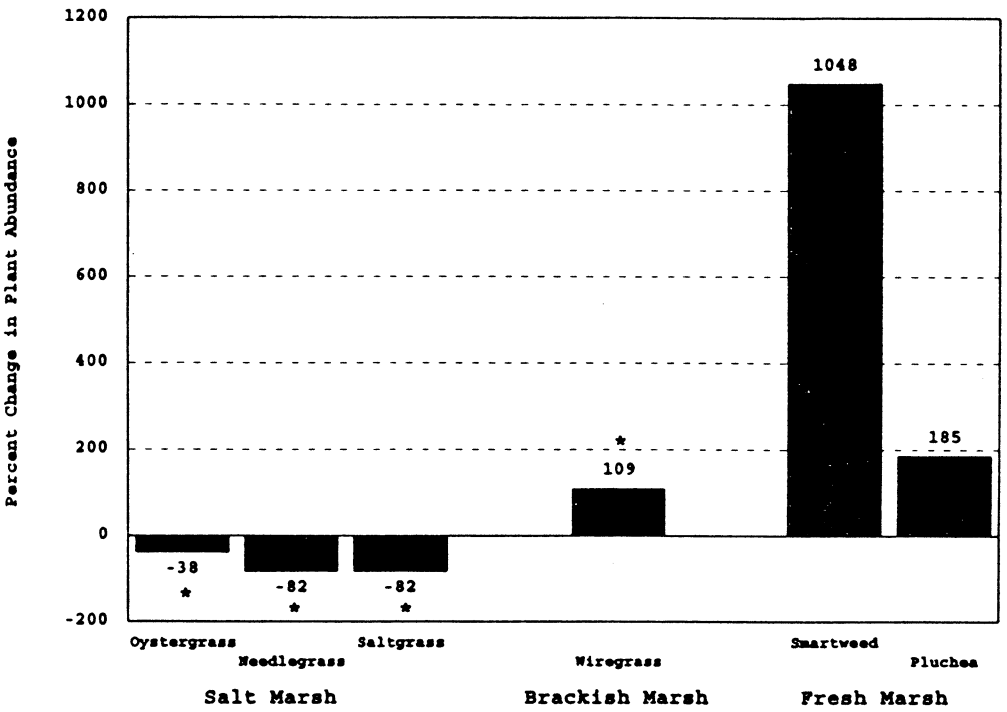

Figure 7. Percent change for vegetation at Caernarvon area. Percent change between pre- (1988-90) and post- (1991-94) periods (Geaghan 1995). Asterisk indicates significant change at 5% level.

DISCUSSION

The current approach does not factor in the combined effects of the three

structures influencing the estuary (figure 2). More importantly, it does not examine the long-term implications in terms of resource base sustainability, which is extremely important in an area such as the Breton Sound Estuary most of which has a relative sea-level rise rate of 0.6-1.2 meters (2-4 feet) per century (CEI 1993).

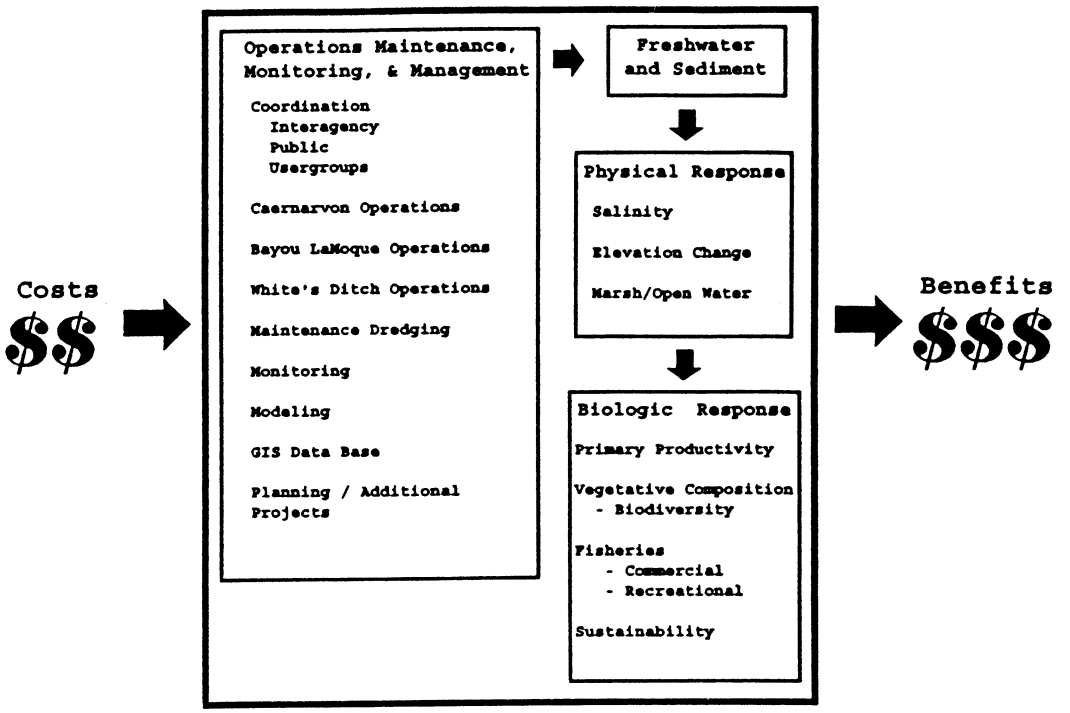

Figure 8. Conceptual Management Model of the Breton Sound Ecosystem Management Plan.

The measurements of the outputs described above provide an index of short-term species-specific responses only. Longterm sustainable management of this resource base must examine the underlying causal processes.

The authors propose that an economic model be developed that will serve as a predictive tool for assessing the various alternatives available (figure 8). Such a model should provide for the comparison of alternative operation schedules of all three structures in terms of their relative merits for long-term sustainability of the resource base (i.e., the marsh), the resulting dependent biological productivity over time, as well as effects on short-term species-specific responses, and the socioeconomic costs of any

adjustments required by user groups such as oyster fishermen.

In order to develop and implement the resulting public policy in an equitable fashion, the economic costs and benefits should be discussed with affected user groups, other local interests, and resource agency representatives. The final plan should be adjusted to accommodate such concerns.

The initial inputs for the conceptual model are the costs associated with interagency planning and management, freshwater diversion operations and maintenance, and the social costs resulting from plan implementation such as relocation of oyster leases in over-freshened areas (figure 8). Model outputs would be the economic benefits realized from increased wildlife and fisheries productivity, increased recreational opportunities, and infrastructure protection by virtue of preserved marsh. Net economic benefits would be the output benefits minus input costs.

In order to establish specific operational objectives consistent with this Breton Sound management strategy, we recommend the following steps:

1. Determine the most desirable seasonal isohalines based on short-term fish and wildlife objectives.

2. Based on hydrologic modeling, determine the operational management plan at each proposed diversion location necessary to achieve the desired isohalines.

3. Determine resulting sediment contributions involved through the operational plan evaluated above. Determine the emergent area gained or sustained through alluvial contributions minus the area lost due to subsidence in order to determine increased sustainability attributable to the operational plan under consideration.

4. Estimate social costs such as oyster lease relocations.

5. Discount future productivity in keeping with the predicted diminishment of the resource base, over the life of the project (50 years). Sum all years to determine total economic benefits.

6. Compare other acceptable seasonal isohaline scenarios, repeating steps 1) through 5) until site specific flows are satisfactory for short-term isohaline and long-term sustainability objectives.

7. Implement the most satisfactory (i.e., based on public and interagency response) overall operational plan. This may not be the one providing greatest total benefits if short-term effects are unacceptable.

SUMMARY

The Breton Sound Estuary has experienced deleterious alterations to its hydrology. Fortunately, restorative efforts have begun -- initially focusing on specific species, and subsequently by concentrating on multiple objectives for single freshwater diversions. The operations of existing structures currently remain at a project-specific regime.

It is proposed that ecosystem management of Breton Sound commence in earnest. A process of balancing trade-offs of long-term sustainability with short-term species-specific objectives is proposed. The paradigm recommended is one that is predictive of economic benefits.

ACKNOWLEDGMENTS

We thank Phil Bowman, LDWF, and Dr. James Geaghan for their contributions towards understanding responses to diversion operations in Breton Sound. We also thank Greg Steyer, Steve Underwood, Rick Raynie, and Diana Steller for their efforts in data collection and analysis

in the Breton Sound Estuary. We thank Rachel Wilson at LDNR for her contributions towards development of appropriate ecosystem modeling components and efforts in the Breton Sound Estuary. We thank Dr. Oliver Houck of Tulane University's Law School for his contributions towards interagency and public cooperation in achieving ecosystem-level restoration in Louisiana. We thank Christina Saltus in the Geographic Information Systems (GIS) Lab at LDNR for time and effort in producing figures I and 2. Finally, we thank Margo Olinde for editorial review of this manuscript.

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LONG-TERM MONITORING OF ESTUARINE FAUNAL USE IN THE NUECES DELTA MITIGATION PROJECT, CORPUS CHRISTI, TEXAS: 1989 THROUGH 1995.

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ABSTRACT

Long-term monitoring is essential in comprehending the impacts of regulatory decision-making regarding functional wetland mitigation. This study monitored faunal and hydrological aspects of a 'failed' mitigation project. The original goal of providing replacement Spartina alterniflora habitat failed, but the project did produce estuarine habitat as non-vegetated bay bottom. Sampling occurred monthly at mitigation and natural marsh stations from June 1989 through August 1995. While functional habitat differences exist, changing environmental conditions and time reveal faunal convergence between locations. Experimental plantings in 1991 and 1992 produced mixed results. Design modifications to 2.4 ha in 1994 resulted in the creation of successful salt marsh habitat. Monthly sampling occurred at this location from October 1993 through August 1995. Continued comparisons are essential for evaluating further design modifications necessary to ensure total project success.

INTRODUCTION

In March 1987 the U.S Army Corps of Engineers (USACE) and the Port of Corpus Christi Authority excavated an 80 ha (36 ha supporting emergent vegetation) borrow area of upland habitat in the Nueces River Delta to satisfy mitigation related to the Corpus Christi Ship Channel 45-Foot Dredging Project. Plans also called for creating a network of channels and ponds to maximize water circulation and edge effect, simulating natural salt marshes. Tidal exchange began September 1988 to allow for substrate saturation before initial planting in February 1990. The aim was to create a self-sustaining and persistent salt marsh that provided suitable habitat for a diverse group of species. Successful establishment and persistence of Spartina alterniflora was the sole criteria for judging project success and habitat function.

A major problem in evaluating successful wetland mitigation is lack of long-term monitoring and comparisons with naturally occurring wetlands (PERL 1990). Addressing that issue, the Center for Coastal Studies (CCS) began to monitor the faunal and hydrological aspects of the Nueces Delta Mitigation Project (NDMP) in June 1989. Our reports documented failure of the project within six months of planting (Ruth 1990; Nicolau and Adams 1993; Nicolau 1993; Nicolau 1995). Incorrect elevation and slope of designated Mean Sea Level (MSL) areas, and high salinity, hindered development. While a failure according to project objectives, it did not hinder successful colonization by a wide range of species. Functional habitat differences exist between mitigation site and natural marsh, but time and changing environmental conditions are resulting in convergence. A major portion of successful species utilization within this site relates directly to proximity of the natural marshes and deltaic hydrology. Experimental transplant projects conducted during years 2 and 3 demonstrated that successful development and persistence of Spartina are possible under the right conditions.

Plans to modify elevations involved construction of a 2.4 ha test marsh on an existing MSL area, after raising interior elevations approximately 25 cm. Evaluation of construction techniques and planting success would

precede full scale area modifications. Additional planting of 16 ha would follow evaluation in an attempt to successfully satisfy original mitigation requirements. Plans also called for the planting of a 0.6 ha area of submerged aquatic vegetation next to this test area. Planting of these areas took place in May 1994.

This study represents the continued commitment of the USACE, USFWS, and the CCS to monitor this mitigation site. All participants encouraged continuation of monthly sampling and analysis of information concerning habitat function, once transplanting appeared successful. While the recent success is encouraging, we are far from satisfying original mitigation requirements. Ongoing research will still prove vital in evaluating modifications necessary to ensure eventual success. In all wetland mitigation, creation, or enhancement projects, past knowledge regarding site dynamics should contribute to successful creation of functional marsh habitats.

STUDY SITE

Located on the southeastern Texas coast, the site lies in the Nueces River Delta, 2 km west of Nueces Bay and 120 m north of the Nueces River (Figure 1). Primary water exchange is through a direct connection to Nueces Bay, augmented by a network of natural deltaic channels. The Choke Canyon/Lake Corpus Christi (CC/LCC) reservoir system severely limits fresh-water inflows into Nueces Bay, with the core wetlands receiving water only during rare flooding events. Reservoir operations theoretically reduce freshwater inundation frequencies from 2.3 flood events to 1.2 events annually. Lack of significant rainfall, except for year-three of this study, resulted in even fewer flooding events. The dynamics of the Nueces Delta system are critically dependent on these infrequently occurring events (HDR 1991).

The climate is subtropical and semiarid with typically, mild winters. Moderate rainfall and high evaporation rates produce an annual net moisture deficit. Prevailing southeasterly winds often serve as a primary

source of atmospheric moisture. Tides are primarily diurnal with an average amplitude of 10 cm. Astronomical tidal fluctuations range from 0-60 cm, with wind direction and velocity being the major controlling factors influencing inundation duration and tidal range within the estuary. Salinities in Nueces Bay are extremely variable. Historical salinity readings ranged from 2.0 ppt during freshening events to 45.0 ppt during times of low rainfall and river flow, producing a normal range of 31.0-38.0 ppt. Martinez (1975) reported on additional hydrological parameters for Nueces Bay with water temperatures averaging 23.0^{o} C, pH 8.0, and dissolved oxygen 4.9 mg l^{-1}.

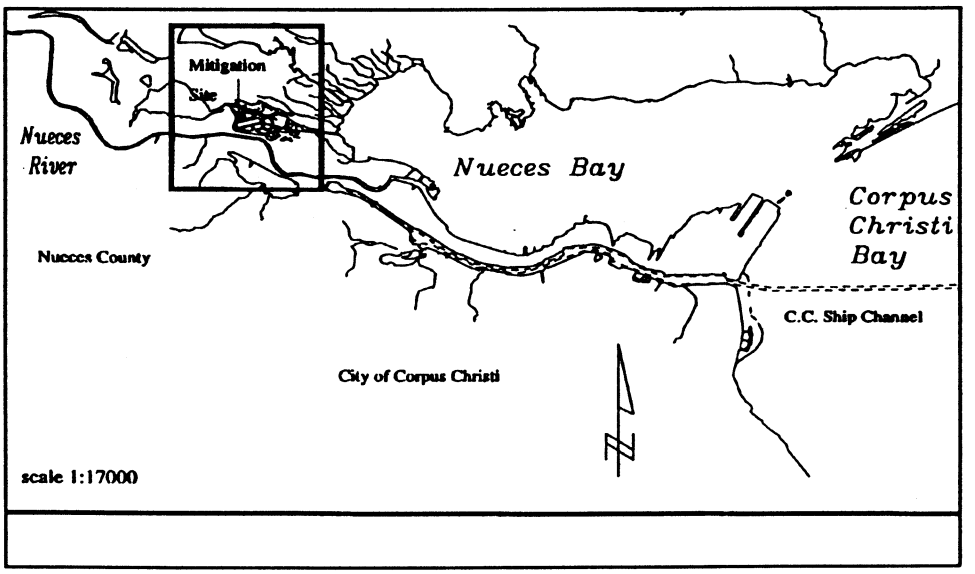

Figure 1. Map detailing location of study site within the Nueces-Corpus Christi Bay estuary.

METHODS AND MATERIALS

In year one, monthly sampling occurred at three mitigation stations (1, 2, and 3) and one natural marsh station (4). In year-two, Station 5 in the natural marsh replaced Station 1 in the mitigation site. Beginning in year-five sampling occurred at six stations: two long-term nonvegetated mitigation stations (2 and 3); two long-term natural marsh stations (4 and 5); and two newly transplanted stations (6 and 7) (Figure 2). Sampling

from June 1989 through August 1993 occurred along three permanent transects: 1) the MSL area designed to support Spartina, 2) adjacent channel edge, and 3) tidal pond. No faunal sampling occurred in July and August 1989. In September 1993 sampling ended at transect 3 due to construction modifications at Stations 6 and 7, and natural sedimentation at Station 4. A PVC push core, 10.12 cm diameter, sampled benthic infauna to a depth of 10 cm. Sampling consisted of three replicates, combined in 0.5 mm mesh biobags. Epibenthic invertebrate and nekton sampling employed a 1 m wide benthic marsh net (1.1 mm mesh) secured to an aluminum frame or sled.

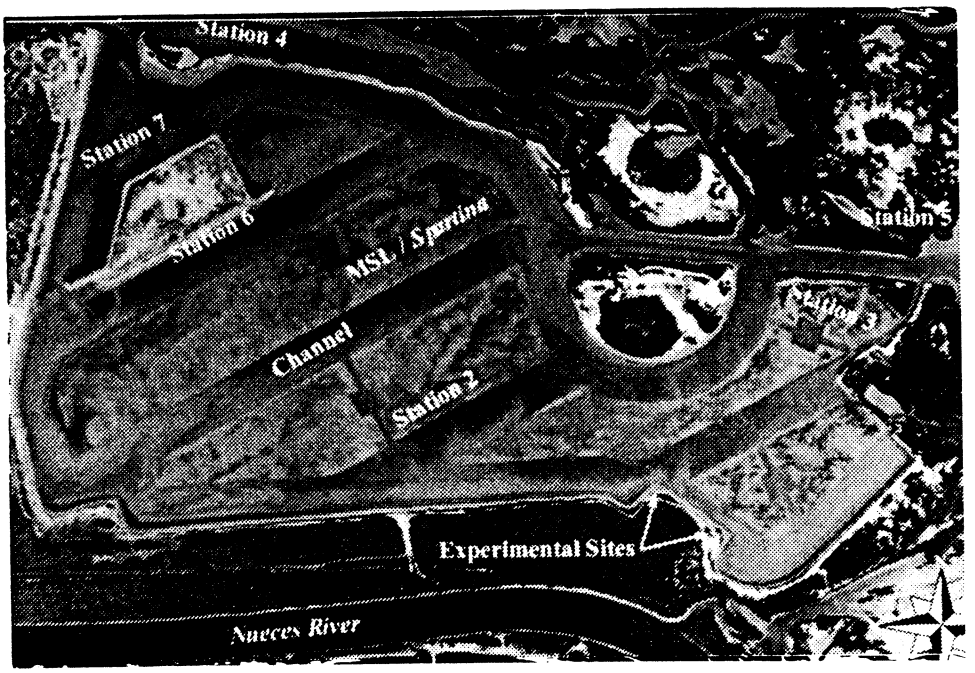

Figure 2. Aerial photograph of mitigation site, depicting locations of sampling and experimental stations. (Lanmon Aerial Photography No. 3295-1)

Sampling involved pulling the sled once at each transect for a distance of 25 m. Sample fixation was in a 10% formalin/seawater mixture with rose bengal. Laboratory analysis consisted of washing all samples through nested sieves (minimum 0.5 mm), with organisms sorted, counted, and identified to the lowest possible taxon. Bird counts consisted of visual observations within a 50 m^2 area around each station on a semimonthly basis, with usage modes (i.e., feeding, resting, or flyover) noted. A Hydrolab Surveyor II recorded hydrological parameters at each station.

Paired t-tests compared monthly hydrological parameters between stations. One-way ANOVAs provided all faunal density analysis after being log transformed (density + 1). Calculation of species diversity utilized the Shannon Diversity Index (H'). All analysis between mitigation and natural marsh stations consist of mean values of transects and stations. Values for Stations 6 and 7 represent mean values of transects.

An experimental planting of Spartina and Scirpus maritimus occurred in the mitigation site to determine if the substrate, elevation, and slope of these locations were suitable for colonization (Figure 2). A nearby natural marsh location provided transplant material. Sampling consisted of visual and photographic inspection conducted on a semimonthly basis. Two quantitative surveys to ascertain percent survival of the newly transplanted Station 6 occurred in May and August 1994 along randomly sampled transects. Survey of seagrass area at Station 7 occurred in May 1994 in randomly selected 1 m^2 plots. Survival determination depended on the presence of any green living material in the sample. Additional surveys conducted at Station 6 consisted of aerial and slide photo analysis to determine approximate percent cover of Spartina.

RESULTS

Mitigation site MSL areas underwent submergence 95% of the days sampled during the six years. In the natural marsh the figure was 82%. Stations 6 and 7 two year values were 79% and 96%, respectively. Mean DO, pH, and water temperatures values were consistent, with no significant differences found between the mitigation and natural marsh. Significant differences in pH and water temperature occurred during the two year sampling period between Stations 6 and 7 and the mitigation site and/or natural marsh. DO ranged from 3.4 to 12.9 mg l^{-1} with mitigation site and natural marsh mean values of 7.7 mg l^{-1} and 7.6 mg l^{-1}. Mean values for Station 6 and 7 were 7.4 mg l^{-1} and 7.3 mg l^{-1}. The range of pH was from 7.0 to 10.4 with an approximate mean value of 8.2 for all locations. Water temperature ranged from 6.4° C in December 1989 to

36.0° C in June 1995. Mean value for the mitigation site was 22.9° C, natural marsh was 22.6°C, Station 6 was 24.5°C, and Station 7 was 22.2° C. Salinity concentrations varied during the study with significant differences ($p<0.05$) between mitigation site and natural marsh in the first, third, and overall sampling years. Significant differences also existed between Stations 6 and 7 and the mitigation site and/or natural marsh during the two years sampling occurred. A high of 50.0 ppt in October 1989 came at the end of a drought and before mandated releases began from the CC/LCC reservoir system. The low of 0.0 ppt in May 1992 followed flooding events beginning in December 1991 (Figure 3).

Although Spartina planted in February 1990 failed to survive, limited natural plant succession of Batis maritima, Salicornia virginica, and Distichlis spicata growth occurred along areas disturbed by site construction. Lower salinity levels prompted the planting of small experimental plots of Spartina and Scirpus in April, June, and August 1991 in an area conducive to moderate tidal inundation (Figure 2). Limited growth occurred at this location through the summer. Above average rainfalls and severe flooding of the Nueces River in early 1992 resulted in maximum vegetative growth during year-three. The most obvious changes occurred in Scirpus and Spartina. Observations revealed distinct changes in stem heights and densities. The Station 4 pond transect produced substantial amounts of Scirpus where it had been previously nonexistent. Furthermore, the existing natural Spartina stand spread across and down the bank, separating it from the mitigation site. The 1991 experimental plots doubled in size and two more plantings occurred in April and May 1992. May transplants died within 30 days due to excessive submergence.

Mandated freshwater releases kept salinity levels low and the persistence of mild winters allowed natural vegetative succession to continue throughout most of years four and five. Succession continued at Station 4, and both experimental plots continued to expand. Quantitative surveys of the newly transplanted marsh took place in May and August 1994 revealing 25.1% and 11.9% live plants, respectively. A visual inspection in the seagrass area at Station 7 took place in May, but high tides and turbid conditions hindered sampling. June 1994 sampling revealed no

surviving transplants. A mild winter and stable salinity levels allowed further natural succession throughout the sixth year. In the newly transplanted marsh, surviving transplants showed notable increases in percent cover. By March 1995 estimates of survival were approximately 50%, and by early summer the percent cover of Spartina was nearly 60%.

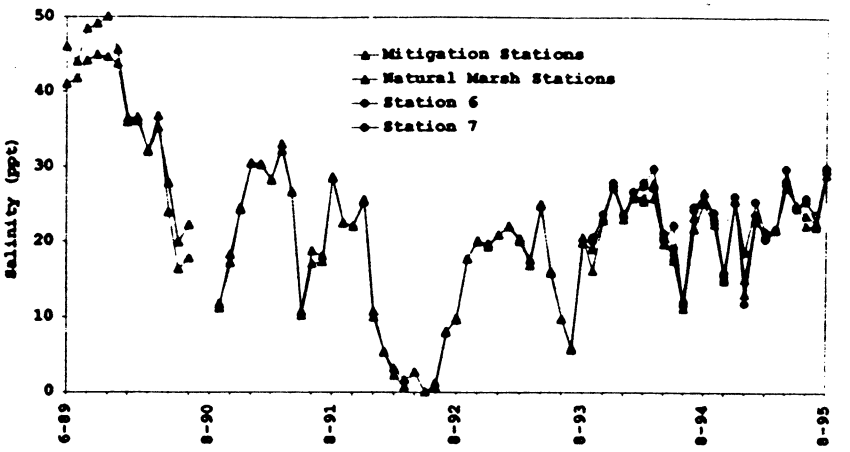

Figure 3. Mean monthly salinity measurements for all locations, June 1989 through August 1995.

Benthic cores yielded 56 species representing six phyla. Annelids produced 23 species of polychaetes, and one species of oligochaete and hirudinid. Arthropods yielded 16 species of crustaceans and two species of insects. Molluscs produced six species of bivalves and four species of gastropod. Unidentified platyhelminthes, nemerteans, and bryozoans accounted for one species each.

The mitigation site contained 50 infaunal species, 14 of which did not occur in the natural marsh. Overall mean monthly density of 3550 m^{-2} consisted of: 1242 polychaetes m^{-2}, 2033 crustaceans m^{-2}, 162 molluscs m^{-2}, and 113 other specimens m^{-2} collected. Natural marsh stations yielded 41 species, including five species not found in the mitigation site. Mean monthly density of 2091 m^{-2} included: 1598 polychaetes^{-2}m , 251 crustaceans m^{-2}, 85 molluscs m^{-2}, and 156 other specimens m^{-2} (Figure 4). Range was from 214 m^{-2} to 26,779 m^{-2} in mitigation stations and from 152 m^{-2} to 10,385 m^{-2} in the natural marsh (Figure 5). Maximum densities

occurred in late winter/early spring with a more even distribution in the natural marsh. No significant differences (P=0.326) existed between densities over the entire period but significant differences did exist in year-one (p=0.008). Species diversity values for mitigation site and natural marsh stations were 0.74 and 0.89, respectively.

Station 6 contained 29 infaunal species. Mean monthly density of 2559 m^{-2} included: 1904 polychaetes m^{-2}, 215 crustaceans m^{-2}, 229 molluscs m^{-2}, and 211 other specimens m^{-2} collected. Station 7 yielded 25 infaunal species. Mean monthly density of 1991 m^{-2} consisted of: 1292 polychaetes m^{-2}, 274 crustaceans m^{-2}, 290 molluscs m^{-2}, and 134 other specimens m^{-2} collected (Figure 4). Twenty-four species that occurred in the other stations were absent from Stations 6 and 7. Densities ranged from 166 m^{-2} to 6562 m^{-2} at Station 6 and from 207 m^{-2} to 8508 m^{-2} at Station 7 (Figure 5). No significant differences existed between Station 6 and Station 7 or between these stations and the mitigation site or natural marsh (all values p>0.180). Stations 6 and 7 yielded overall diversity values, for the two year period, of 0.95 and 0.85, respectively.

Five species accounted for 84.3% of all mean monthly benthic density (Figure 6). They were the amphipod Corophium louisianum, and the polychaetes Streblospio benedicti, Mediomastus ambiseta, Laeonereis culveri, and Capitella capitata. These species accounted for 85.6% of all organisms collected in the mitigation site and C. louisianum dominated. In the natural marsh these same species accounted for 83.6%, and exhibited a more even distribution, with S. benedicti the dominant species. The same five species represented 77.1% and 74.6 % at Stations 6 and 7 and the dominant organism at both stations was S. benedicti. Higher densities in the mitigation site directly result from the large numbers of C. louisianum collected at Station 2. Removal of this species from the data yields almost equal mean monthly densities for both locations, with the natural marsh stations producing 1906 m^{-2} and the mitigation stations producing 1773 m^{-2}. Discounting C. louisianum at Stations 6 and 7 yielded new figures of 2405 m^{-2} and 1762 m^{-2}, respectively.

Over the six year sampling regime, marsh net samples produced 61 species

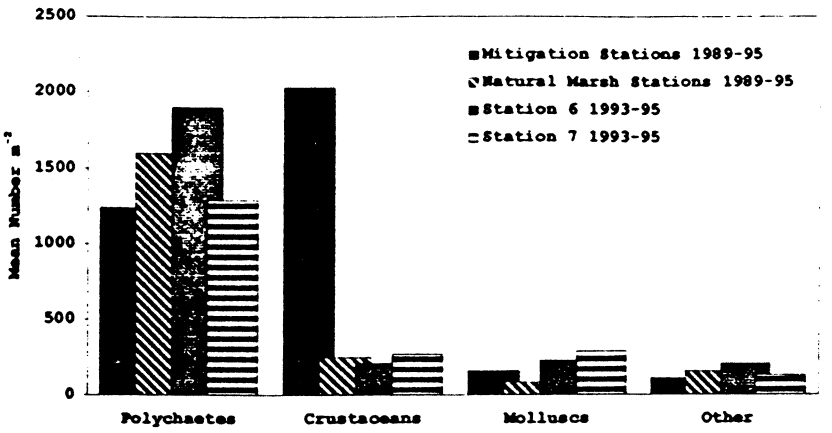

Figure 4. Mean monthly densities of major benthic taxa collected at all locations.

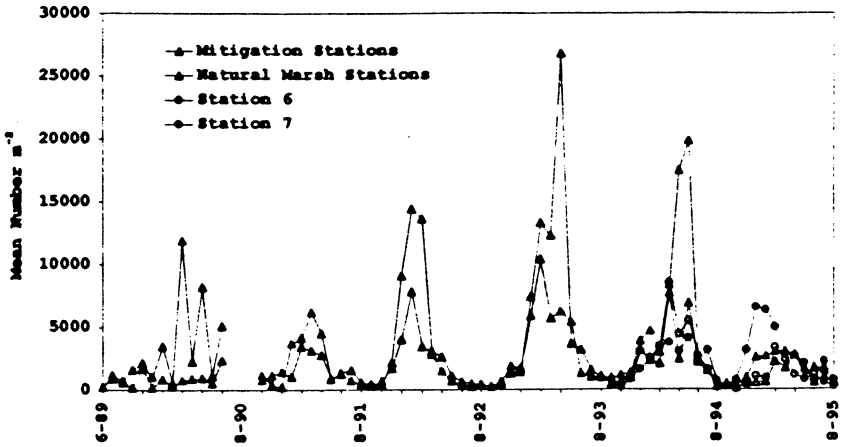

Figure 5. Comparison of mean monthly benthic densities at all locations.

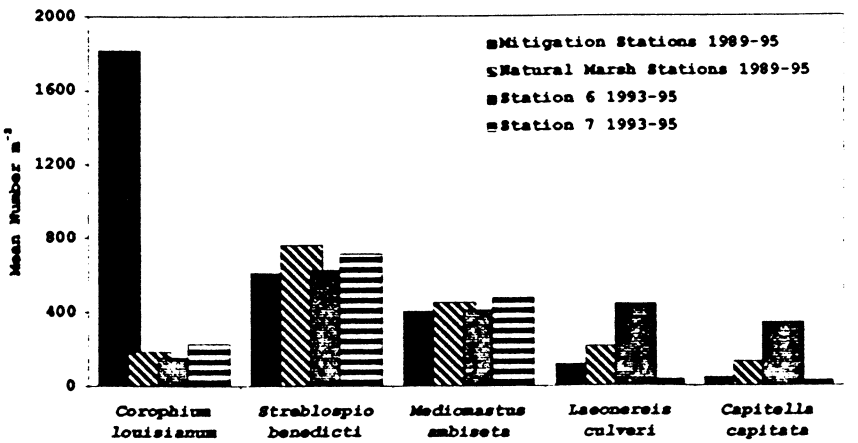

Figure 6. Mean monthly densities of dominant species collected from benthic core samples over the six year period at all locations.

representing six phyla. Vertebrates dominated with 38 species of fish collected. Arthropods produced 16 species of crustaceans and one species of insect. The remaining species consisted of three cnidarians, one ctenophore, one bryozoan, and one mollusc. The mitigation site yielded 55 species, ten of which did not occur in the natural marsh. The mean monthly density of 3.72 m^{-2} included: 2.89 crustaceans m^{-2}, 0.61 fish m^{-2}, and 0.22 other specimens m^{-2} collected (Figure 7). Natural marsh stations produced a total of 51 species, including six species not found in the mitigation site. A mean monthly density of 5.66 m^{-2} consisted of: 4.60 crustaceans m^{-2}, 0.93 fish m^{-2}, and 0.13 other specimens m^{-2} collected (Figure 7). Range was from 0.17 m^{-2} to 17.79 m^{-2} in mitigation stations and from 0.00 m^{-2} (no water periods) to 38.33 m^{-2} in the natural marsh (Figure 8). Maximum densities in the mitigation site and natural marsh occurred in the winter and summer months, respectively. Due to periods of no water occurring in the natural marsh it is difficult to assess accurate distribution patterns. No significant differences ($P=0.658$) existed between mitigation and natural marsh station densities over the entire period but significant differences ($p=0.012$) did exist between locations in year-two. Species diversity values for mitigation site and natural marsh stations were 0.81 and 0.72, respectively.

Station 6 contained 25 epifaunal and nektonic species. Mean monthly epifaunal density of 3.46 m^{-2} consisted of: 2.36 crustaceans m^{-2}, 0.86 fish m^{-2}, and 0.23 other specimens m^{-2} collected. Station 7 yielded 21 species with a mean monthly epifaunal densities of 2.31 m^{-2} which consisted of: 1.54 crustaceans m^{-2}, 0.38 fish m^{-2}, and 0.38 other specimens m^{-2} collected (Figure 7). Thirty-three species occurring in other stations were absent from Stations 6 and 7. Densities ranged from 0.32 m^{-2} to 15.68 m^{-2} at Station 6 and from 0.00 m^{-2} (no water period) to 4.04 m^{-2} at Station 7 (Figure 8). Maximum densities tended to occur in the winter and spring months at Stations 6 and 7, respectively. No significant differences existed between Station 6 and Station 7 or between these stations and the mitigation site or natural marsh (all values $p>0.180$). Stations 6 and 7 yielded overall diversity values, for the two year period, of 0.80 and 0.86, respectively.

Six species accounted for 86.5% of mean monthly epifaunal density at all stations (Figure 9). They were the daggerblade grass shrimp Palaemonetes pugio, the brown shrimp Penaeus aztecus, the naked goby Gobiosoma bosci, the ctenophore Beroe ovata, the white shrimp Penaeus setiferus and the blue crab Callinectes sapidus. These species accounted for 87.9% and 86.2% of all organisms collected in the mitigation site and the natural marsh, respectively. The dominant organism collected in both locations was P. pugio. From 1993 through 1995, these six species represented 79.4% and 85.8% of all mean monthly epifaunal density at Stations 6 and 7, respectively. The dominant organisms collected at Station 6 and 7 were P. pugio and P. aztecus, respectively.

Aquatic and wading shorebirds, herons, egrets, pelicans, waterfowl, and other avian species used the site. Observations revealed increasing numbers of species the first four years with 27, 49, 59, and 66 recorded. In year-five the number observed declined to 59, and in year-six rose to 63. Except for year-one, the number of species observed through year-four was very similar. The mitigation site yielded 25, 35, 51, 57 species while the natural marsh yielded 17, 36, 48, and 55 species. Both locations produced a total of 50 and 48 species each the next two years, respectively. During the fifth and sixth years Station 6 produced 36 and 41 species observed with 30 and 31 species recorded at Station 7. Many species absent during high water periods occurred in large numbers when water levels were low, while some species demonstrated no preference.

For the first four years avian usage was higher in the mitigation site for all seasons. In the summer of year-five, following test marsh construction, Station 6 produced the highest numbers of individuals observed. In year-six avian usage was higher at Station 6 for all seasons. Analysis of usage modes showed higher instances of flyover, followed by resting and feeding, for both the mitigation site, natural marsh, and Station 7. Station 6 revealed higher percentages of resting followed by feeding and flyover. Besides the normal avian usage of the area, a large scale nesting event (57 nests with 95 eggs) occurred at Station 6 during year-five. The creation of levees provided habitat that appeared to be conducive to nesting by Gull-billed Terns, Black Skimmers, and Forster's Terns, but most chicks failed to survive due to predation, high tides, and heavy rainfall. Final

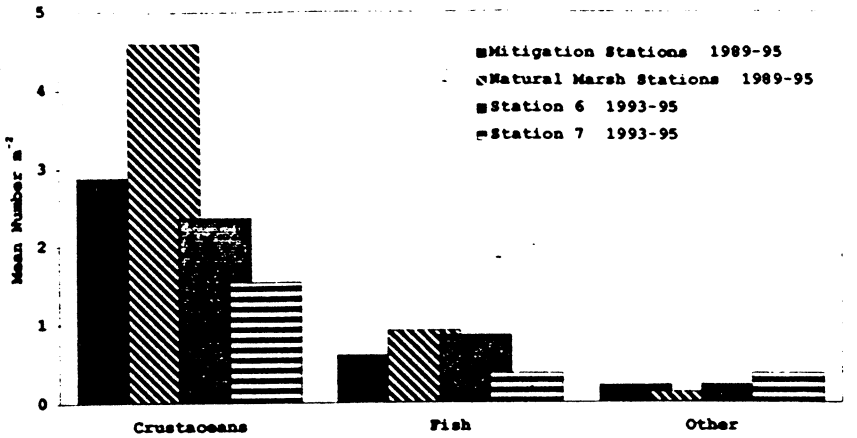

Figure 7. Mean monthly densities of major epifaunal taxa collected at all locations.

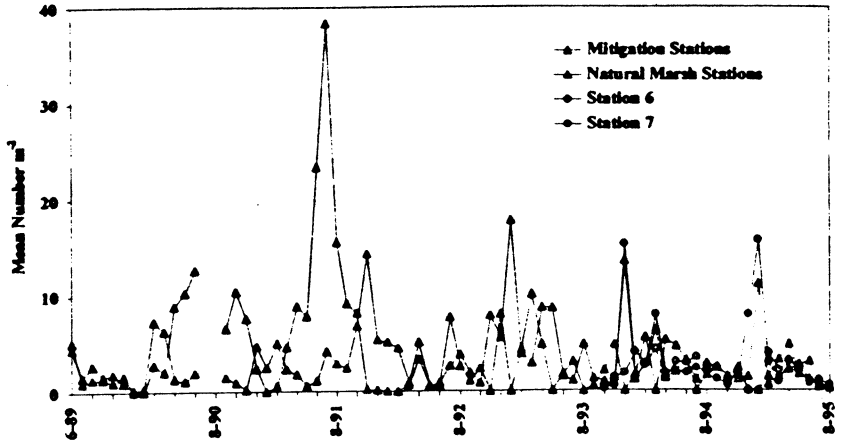

Figure 8. Comparison of mean monthly epifaunal densities at all locations.

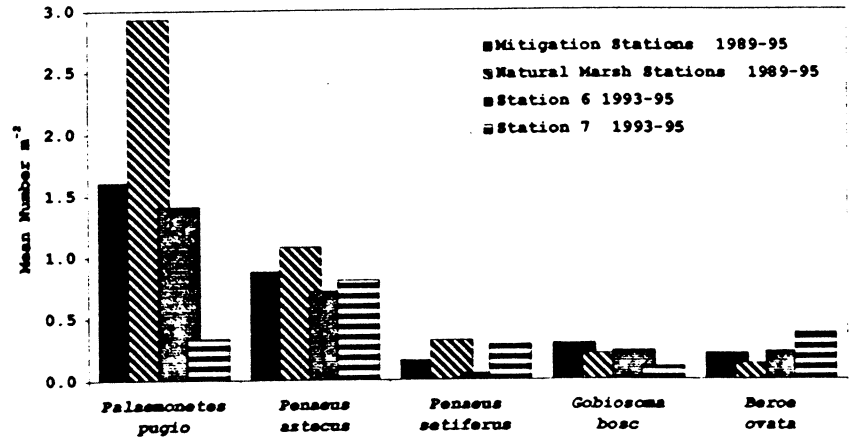

Figure 9. Mean monthly densities of dominant species collected from marsh net samples over the six year period at all locations.

examination of the area for nesting activity occurred July 1994 with no eggs observed. In conducting surveys of the area, observation of Station 3 revealed one juvenile Gull-billed Tern feeding from its mother. This was the only juvenile bird observed throughout the area.

DISCUSSION AND CONCLUSIONS

Although highly influenced by the surrounding natural marsh system, this created site did not constitute a functioning salt marsh. Until the 1994 transplanting effort, comparisons have been between a nonvegetated bay bottom and a naturally functioning salt marsh. Recommendations to modify elevations to provide viable transplant habitat remain valid, as demonstrated by the success seen at Station 6. A beneficial modification feature were the ponds and creeks formed during construction. Typically, constructed marsh designs produce low edge-to-area ratios; resulting in continuous stands of marsh and reduced edge habitat. Edge habitat is important to species utilization and habitat values. Proper elevation, edge-to-area ratio, and inclusion of drainage features are necessary to ensure created marshes hydrologically and morphologically resemble natural systems (Minello et al. 1994; Peterson and Turner 1994).

Although not quantified, except at Station 6, Spartina alterniflora, and other vegetation underwent distinct changes in growth rates and spatial distribution. Many marsh plants suffer from stress due to high salinity and low nitrogen supplies. It appears that increased freshwater influxes served to stimulate growth by reducing soil salinities and enriching depleted nitrogen supplies (Zedler et al. 1992). Regarding the failure of the Ruppia maritima seagrass area of Station 7, turbid water conditions are natural in the Nueces River Delta and this location did not afford the adequate amounts of sunlight necessary for seagrass survival.

As stated, a primary factor influencing salt marsh ecosystem growth and development is salinity (Mitsch and Gosselink 1986). Short-term monitoring studies often discount temporal variability and fail to fully characterize complex salt marsh functions. Since inception, study site

salinities ranged from hyperhaline to fresh, influenced by floods and the mandated freshwater release/pass-through program. While the last two sampling periods may appear more "normal" than preceding years, data analysis continues to reveal how strong intermittent hydrological pulses continue to influence the Nueces River Delta. Studies correlate expanding benthic densities with increased freshwater inflow in Texas estuaries due to nutrient inputs that stimulate primary productivity. Population expansions result from the utilization of this increased productivity (Montagna and Kalke 1992). Increased benthic density from year-three to year-four lends support to these studies. Montague and Ley (1993) and Zimmerman et al. (1990) provide evidence of species biomass and dominance differences in areas experiencing salinity fluctuations or gradients. While comparative yearly density differences between stations may not be a direct result, due to small salinity variations, year to year salinity changes were often substantial. Patterns of distribution and abundance exhibited by many small benthic invertebrate species tend to differ both temporally and spatially (Kneib 1984). This fact, linked with changing salinity conditions may relate to density differences between locations.

Contrary to other research (Minello et al. 1991; Sacco et al. 1994), benthic densities were consistently higher in the nonvegetated mitigation site due to the large numbers of Corophium louisianum collected at Station 2. While not confined to this station, abundant distribution of this species at Station 2 apparently represented a preferred habitat. Sixth year analysis revealed higher densities in the natural marsh for the first time due to the absence of this species. No explanation exists at this time for the population decline recorded. Polychaetes dominated natural marsh samples possibly due to such factors as: higher organic content, finer sediment grain size, or reduced predation pressures due to restricted habitat caused by regular tidal inundation (Moy and Levin 1991; Minello et al. 1994). Polychaetes also dominated at Stations 6 and 7 in the newly transplanted area. Station 6 modifications now result in regular tidal inundation to the area. Although no modifications occurred at Station 7, this station lies at the mouth of a natural deltaic creek that provided large amounts of sediment and organic deposits during the floods observed in the 1991

through 1992 sampling season. These deposits resulted in a more natural substrate than that seen in other mitigation stations.

The diversity of epifaunal and nektonic species collected reinforces the importance estuarine habitats play in the life stages of many aquatic organisms (Minello et al. 1994; Peterson and Turner 1994). While patterns of distinct seasonal species abundance and habitat preference continue, changing environmental conditions have produced changes in abundance and distribution patterns within the study site, indicating a general downward trend in the total number of epifaunal organisms collected. These changing patterns continue to reinforce the inherent problems related to sampling motile fauna within estuarine system complexes. Numerous cause and effect conditions (salinity and temperature fluctuations, variations in sediment composition, turbidity, water depth, presence of vegetation, and duration and tidal amplitude) may possibly influence the distribution and abundance of motile estuarine species (Minello and Zimmerman 1992; Zimmerman and Minello 1984; Zimmerman et al. 1990). Minello et al. (1991) stated that though marsh habitats may support larger abundances of some animals, species composition may be entirely different. In determining habitat value we must take into account the intertidal nature of marsh vegetation and that it is not always available for exploitation. Small resident and transient fish are important components of estuarine marsh systems. They provide a necessary food source for such larger transient species as Cynoscion nebulosus and Sciaenops ocellatus traveling through the system (Peterson and Turner 1994).

Wetland areas are vital for many migratory avian species, providing areas for forage, resting, and refuge essential to the species' survival (Mitsch and Gosselink 1986). While the NDMP provided no functioning marsh habitat at first, it did create numerous areas of slightly submerged or tidally exposed mud flats that theoretically provided a large invertebrate food source (Baird et al. 1985). During critical migratory periods avian usage is always highest in the mitigation site during low water periods, with greater percentages of active feeding being recorded. Avian usage of this mitigation site continues to increase with the passage of time. As number

of surveys decreased from 35 to 24 times a year, overall total number of species observed increased. Many avian species are transient visitors to this location, but the area does support numerous permanent residents. The newly created marsh at Station 6 provides ideal habitat conditions, as evidenced by the large numbers and nesting of avian species recorded in year-five. In year-six, approximately half of the individuals recorded occurred at Station 6. Future research projects, documenting avian utilization of these intertidal areas, would further scientific knowledge regarding functional equivalence between created and natural habitats.

ACKNOWLEDGMENTS

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The Design Process Utilized To Restore Diked Salt Hay Farms To Natural Marshes

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ABSTRACT

Fifty years of diking have reduced marsh plain elevations, obliterated tidal channels, reduced *Spartina alterniflora* and increased *Phragmites australis* in marshes on the Delaware River Estuary. Prior to diking, the marshes were dominated by *Spartina alterniflora* however, the post diked marshes are now dominated by *Spartina patens*. The restoration goal is to obtain a mixed salt marsh ecosystem dominated by *Spartina alterniflora* but with significant areas of *Spartina patens*. The first step in the design processes was to obtain agreement among regulators, ecologists, engineers, and the public as to what a marsh is and what important functions a marsh performs. This was followed by a discussion and agreement as to the essential sedimentary and hydrodynamic characteristics necessary to achieve the desired ecosystem. The engineers then used a two-dimensional hydrodynamic numerical model to design marsh channels that would be nonerosive, would achieve the proper channel cross-sectional shape, and produce a suitable hydro-period on the marsh plain to facilitate growth of the desired species. We will describe both the social/political and the scientific/engineering steps necessary to achieve a final design acceptable to all involved parties.

INTRODUCTION

The restoration of salt hay farms in the Delaware River Estuary to natural marshes using natural engineering techniques presents many difficulties which must be overcome. These range from agreeing on definitions of

what a marsh is, to determining what the optimum hydro-period of the restored marsh will be. The projects described below are in progress. Construction has begun at one site, but the other sites have not obtained all of the necessary permits (Fig. 1). The restoration processes included more than deriving an engineering design. The process required extensive data collection, engineering analysis, and consensus building to assure the public and permitting agencies that restoration of a marsh would be advantageous and successful.

BACKGROUND

Salt marshes have a unique inter-relationship between vegetative, hydraulic, and soil characteristics which allow the marsh to function. Salt marshes along the east coast of North America have historically been modified by altering the marsh hydrology and/or vegetative cover for farming activities. During the early 1900's the States of New Jersey and Delaware encouraged transforming natural salt marshes for agricultural purposes. While this may initially have involved road building and harvesting of natural vegetation, construction of dikes around the perimeter of the marsh to eliminate daily tidal inundation often followed. The objective of dike construction was to control tidal inundation to promote production of *Spartina patens*. These dikes were generally 2-6 feet in height, designed only to keep daily tidal flows out of the marsh. The dikes were often over-topped during storms; therefore, a series of drainage ditches and water control structures were also constructed to drain the marshes after storms. Additionally, these water control structures were designed to flood the marshes, generally during the spring, to provide salts and nutrients required to encourage *Spartina patens*.

Diking and farming have had a dramatic effect on the ecology and hydrology of the salt hay farms. The natural meandering creeks, no longer carrying tidal waters twice a day, have been filled in to accommodate farming activities and replaced with linear drainage features. These drainage features terminate at the water control structures (Fig. 2).

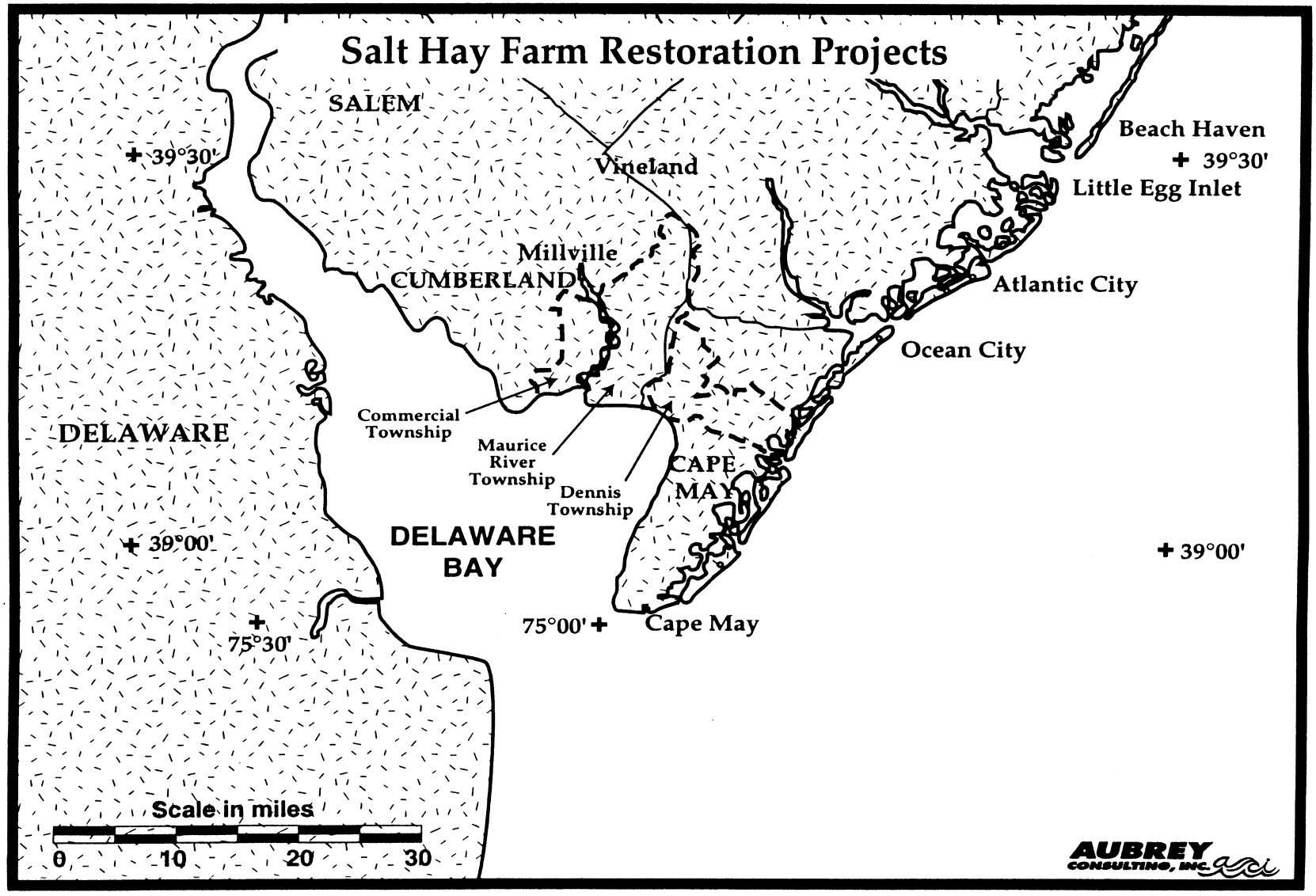

Figure 1: Location of Salt Hay Farm restoration sites.

Depending on the farming technique applied, the marsh plain was wetted only monthly or yearly. Therefore, while the elevation of the marsh plain was not immediately affected by farming activities, the construction of the perimeter dikes hydraulically transformed the marsh plain from a normal marsh with low, transitional, and high regions into high marsh. The result of these activities was the reduction in marsh area occupied by *Spartina alterniflora* and establishment of a less diverse flora community dominated by *Spartina patens* and *Distichlis spicata*.

The alteration of the habitat had several unanticipated side effects which resulted in long-term consequences to the salt hay high marsh ecosystems. First, the elimination of daily tide made conditions favorable for the establishment of *Phragmites australis*. Rains coupled with elimination of the daily tides reduced the salt content of the soils. The lowered salinity and reduced flooding lessened the natural environmental stresses that normally would have inhibited *Phragmites*. Second, construction of the perimeter dikes eliminated the influx of sediment to the marsh plain. Under normal conditions, marshes act as a trap for sediments which, along with the accumulation of peat, helps to maintain the elevation of the marsh plain. Elimination of the sediment supply, sediment compaction, and oxidation of the drained sediments all result in a lowering in elevation of the marsh plain.

Construction of perimeter dikes also had consequences that extended beyond the salt hay farm ecosystem. Salt hay farming was a widespread activity in the lower Delaware River estuary. The construction of the perimeter dikes segregated thousands of acres from normal estuarine processes. Elimination of the marshes from the daily tidal inundation reduced the tidal prism of the creeks outside the diked areas that previously supplied water to these systems. Reduction in the tidal prism reduced the volume of tidal waters which flowed through the tidal inlets and creeks. This resulted in sedimentation within these creeks, reduction in their size, and in some instances, the disappearance of the smaller tidal creeks.

Marshes have long been known to be net exporters of organic material in

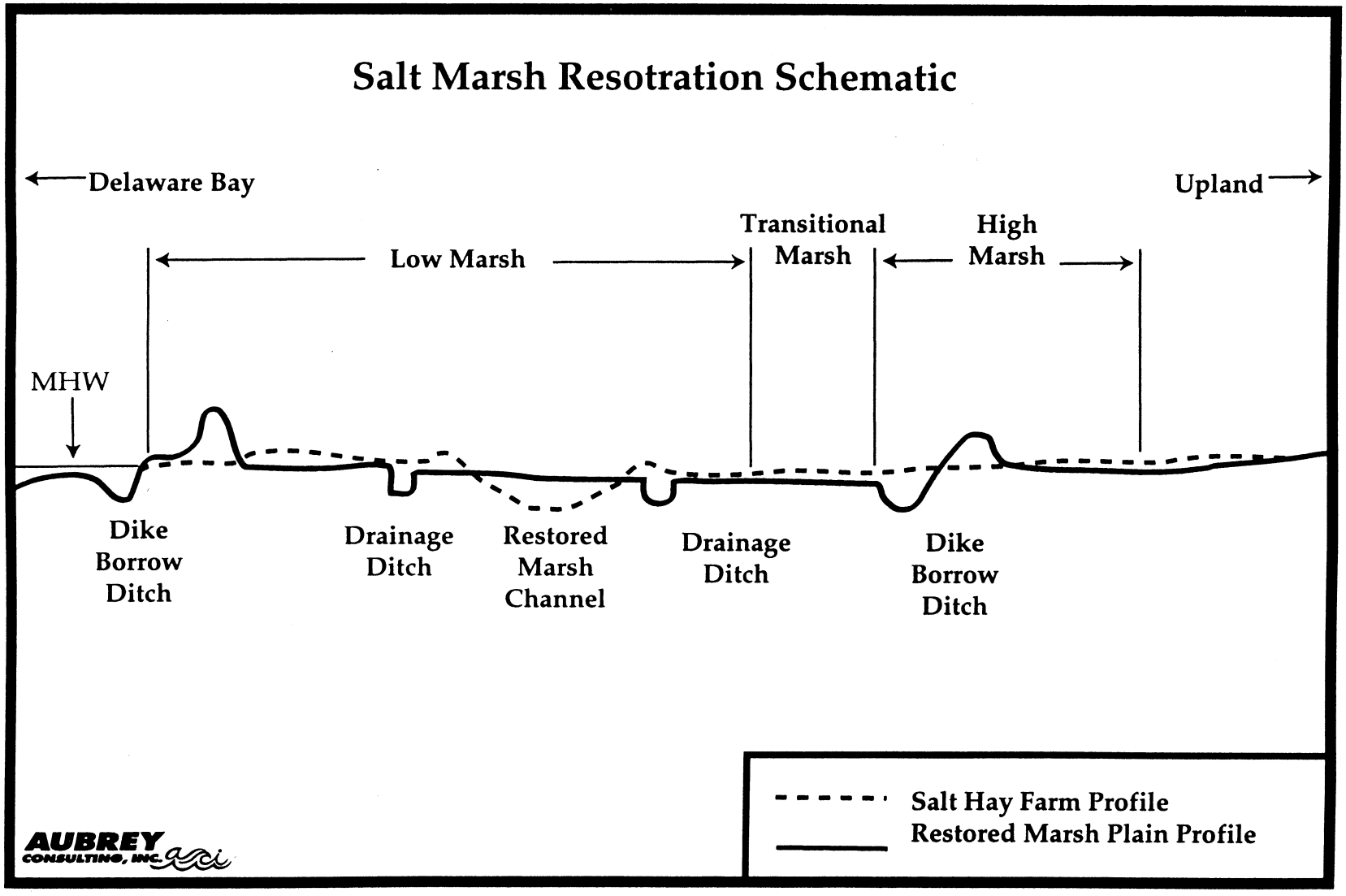

Figure 2: Idealized topographic profile of Salt Hay Farm and restored marsh.

the form of algae, detritus and animals. This material exported from the marsh forms one basis of the estuarine food chain, directly and indirectly providing food for fishes. Removing large tracts of marsh from the daily functioning estuarine system reduced detrital export to the Delaware River Estuary system.

Diking of the salt marshes also increased the potential for catastrophic upland flooding and reduced the export of sand from the tidal creeks to the river littoral system. The construction of the dikes transformed the marshes into an almost uniform high marsh. Elimination of the natural transition from low to high marsh and construction of the dike changed the natural flood attenuating capabilities of the marsh. Normal flood waters were effectively held out by the dikes, but during severe storms dikes frequently failed. When the dikes failed, sea water moved rapidly into and flooded areas previously protected by the dike. Additionally, the elimination of low marsh species and the reduction in elevation of the marsh plain minimized the ability of the marsh to absorb locally generated waves and abate storm surge. As a result, the salt hay farm perimeter dikes provided a false sense of security from storm surge.

The marsh plain effectively stores water during severe storms. Typically, storm surge increases gradually as the winds associated with the storm increase in intensity. As the storm passes, the winds abate quickly leaving the storm water on the marsh plain. When this occurs, relaxation of the storm surge increases the ebb tidal velocities to an extent that sand is eroded from the channel bottoms and is transported out of the creeks into the littoral system. Construction of salt hay farms altered this sequence and eliminated this source of sediment to the littoral system. As a result, sediment supply to many of the river/bayfront littoral systems along the Delaware River estuary was greatly reduced, contributing to bayfront erosion.

The State of New Jersey has restored several salt hay farms to natural salt marshes. Their approach has been to breach the dikes and allow the restoration to occur naturally. Although this process may take several

different pathways and produce widely different results; over time the marsh will successfully restore itself (Shisler, 1989). For the purposes of this paper we are defining successful marsh restoration as the successful re-establishment of normal tidal flows and transitional low to high marsh areas populated with *Spartina* species. The dike breaching approach applied by the State of New Jersey has in several cases resulted in the establishment of large open water areas which have been slow to revegetate for several reasons. The uncontrolled breaching of the dikes allowed erosion to lower the marsh plain elevation, remove vegetation, and produce braided stream channel systems which are unfavorable to sedimentation. The large shallow open water areas with emerging salt marsh plants attracted large numbers of migratory water fowl, such as snow geese which have impeded natural marsh revegetation as the result of over grazing.

These difficulties have not stopped restoration but have resulted in multiple restoration pathways and highly divergent timelines. Uncontrolled breaching may result in marshes with a large percentage of open water that may remain unvegetated for tens of years. Such marshes provide large areas of open water habitat but reduce the primary productivity from marsh plants. Other marshes may have a large percentage of high marsh habitat which flood infrequently and contain a small percentage of open water. This marsh type produces more macrophyte production but provides little fish nursery habitat. Often, oligohaline restored marshes have become dominated by *Phragmites* which although having large primary production, yields detritus high in refractory materials which are not readily exported to the estuary. Further, *Phragmites* tends to reduce the open water areas by filling the smaller tributaries with plant litter and rhizomes which ultimately raises the surface of the marsh plain, further reducing the aquatic habitat.

All of the restoration scenarios discussed in the preceding paragraph represent viable marsh ecosystems at different points on the restoration pathway. Each point along this pathway, however, has different characteristics and functions. Because the direction that restoration takes

is strongly influenced by the way the restoration is undertaken, it is extremely important to agree upon an objective and a restoration timeline prior to the beginning of the project. Such agreement will help minimize unjustified criticism when judging the success of the restoration project.

In a large restoration project, two critical factors which prevent misunderstandings are education of the stakeholders and obtaining their involvement in the project. The stakeholders include: the group responsible for the restoration, regulators, environmental groups, and the local public. Each stakeholder has valid, often conflicting concerns. The restorer is interested in a successful marsh restoration that satisfies whatever regulatory requirement initiated the activity. The regulator is interested in restoring the marsh for reasons that may vary from ecological damage restoration (i.e., wetlands destruction) to a desire for increased public access; as well as seeing that all applicable regulations are complied with. Environmental groups may be interested in ecological aspects ranging from the desired distribution and interspersion of habitats to the impacts of the use of herbicides to eradicate *Phragmites*. The local public may have concerns regarding potential flooding, groundwater effects, or just merely change. All of these are legitimate concerns, as perceived by the individual stakeholder, and must be identified and addressed early in the process in order to avoid potentially significant project delays. Overlaying all of these concerns is the financial considerations for the project. Each concern may result in an additional cost to the restoration project.

The key to the integration of the stakeholders' concerns is communication. This may take various forms but must be initiated as early in the restoration process as possible. Communication with the regulator must occur prior to the initiation of preliminary design work. If the objectives of the regulator and the restorer do not coincide, this is the time to resolve differences. All of the existing site conditions should be discussed so that the ultimate restoration will be achievable and can be permitted within the existing regulatory scheme.

Preliminary, targeted communication should begin with key local

individuals such as the local government. They will be able to identify problems early and they will begin to get information to their constituencies. Upon completion of the preliminary design, it can then be presented to the public. An effective forum for this communication is a well advertised, open house.

The open house forum presents all of the facts of the project and allows the public access to the individuals involved with the restoration, including the experts. This serves three functions. First, the public can hear first hand about the project and its benefits. Second, any public concerns will be identified early in the process. Third, the public can have input into the project. As a result, the public will feel some ownership in the project if their concerns are addressed early and secondly, information germane to the project may be discovered. Very often the public is aware of conditions that may affect the ultimate outcome of the project and that may have financial implications, such as the existence of contamination.

Project Pollution Mitigation Considerations

The New Jersey salt hay farms have been farmed since the late 1800's. As a consequence of farming operations, the sites contain both solid wastes and chemical/pesticide contamination. Until relatively recently, solid waste (trash) was handled individually, usually resulting in a private dump on a farmers property. This waste consisted of normal household and farm trash, frequently containing petroleum hydrocarbons (lubricating oil) and metals (paint residues). Typically, this resulted in surface contamination and is easily cleaned up through the removal of the trash and a shallow layer of soil.

Mosquito control is a particular concern in diked salt hay farms because the dried marsh surface develops cracks which are ideal for mosquito reproduction. Mosquito reproduction was traditionally controlled through the use of insecticides. Soil samples from the subject diked salt hay farms were analyzed for pesticide contamination. DDT and its metabolites were

the only pesticides detected, with concentrations at or below the natural background level detected in nearby open marshes. DDT and its metabolites are tightly bound to marsh sediments with a very low water solubility. The concentrations measured were below applicable cleanup criteria and also below levels likely to be a threat to fish, wildlife and other natural resources.

Applications of DDT in the US were halted in 1972 so present concentrations are determined by the rate at which DDT degrades in the environment. DDT is predominately degraded in the sediments by bacteria, fungi and algae (Cheng 1990, Johnsen 1976). In reducing (low oxygen) environments, such as marsh sediments, anoxic conditions combined with elevated levels of anaerobic bacterial populations promote relatively rapid and complete degradation of DDT (Johnsen 1976, Wedemeyer 1967),with half lives in estuarine and marine sediments of 3 and 4.4 years respectively (Lee and Ryan 1978).

Because DDT and related compounds degrade relatively rapidly in marsh soils, and because the organic content (and potential pesticide binding) of marsh soils is high, DDT effects are less likely in healthy marsh ecosystems than in other habitats. In addition, sedimentation within restored marshes will further attenuate DDT concentrations in surface layers and allow these systems to serve as partial filters removing pollutants from the water column which might otherwise degrade water quality.

The Affects of Sea-Level Rise on Marsh Restoration

Sea level rise within the Delaware River Estuary affects the entire ecosystem. The rate of rise (Emery and Aubrey, 1991) is on the order of 2-3 millimeters per year. Marshes respond to long-term changes in water levels through erosion and accretion processes (Redfield, 1972). Sediments carried by the tidal waters settle onto the marsh plain and

accumulate. The underground parts of marsh plants accumulate as peat as long as the sediments are anoxic. The result is that as long as relative sea level does not rise more 10-20 mm yr^{-1}, the high marsh maintains its level and low marsh maintains or may rise in level in relation to the tides. Increases in sea level result in the marsh migrating landward into areas that become intertidal. It is possible for low marsh to grow in elevation and become high marsh but also for the reverse to occur, probably as a result of changes in tidal access after storms. Therefore, the relative area of high and low marsh is in a continual transition in response to these two processes.

Elimination of tidal flows from the salt hay farms has resulted in a relative decrease in marsh plain elevation within the confining dikes. First, the elimination of sediment influx associated with normal tidal flow prevents the marsh plain from increasing in height by sediment accumulation. Second, the elimination of tidal flow results in a compaction of the marsh plain sediment through drainage and drying. Third, the organic material in the sediment oxidizes as air penetrates the better drained sediments. While these processes are acting to reduce the elevation of the marsh plain, sea level is increasing. Therefore, the relative marsh plain elevation within the salt hay farms is decreasing more rapidly than simple sea level rise. This is evident in several of the natural restorations which have resulted in large areas of open water and slow marsh grass recruitment.

Sea level rise has also had an adverse effect on the infrastructure of the surrounding communities. Communities surrounding the salt hay farms are relatively low and frequently have flooding problems. Most of the communities are small and have on-site septic systems and drinking wells. Sea level rise increases the mean bay water level elevation, and therefore has increased the ground water elevation and has moved the salt and fresh water interface near the margins of the estuary. As a result, the surrounding areas have experienced deteriorating efficiency of well and septic systems. Septic systems have been most heavily impacted. Deep, confined aquifers can be readily tapped to provide a fresh water supply unaffected by sea level rise. However, the rise in ground water elevation

has intruded into the septic systems. As a result, many do not meet local codes or minimum ground water separation distances. Some are even partially inundated with ground water.

Restoration Philosophy

Our major approach has been to apply as much natural or ecological engineering as is possible to the restoration. This is an approach which aims for the development of sustainable ecosystems using self design or self organization. It considers the entire ecosystem not just some parts or some species (Mitsch 1996). In this case we are concerned with the whole marsh ecosystem and with its relationships to the estuary, for the ultimate goal is to enhance estuarine fish production by increasing salt marsh production.

With this goal in mind, we want to achieve a marsh with good fish habitat: sinuous marsh tidal creeks with shallows and deep areas, a large proportion of vegetated marsh which is traversed by many small to large tidal creeks to provide lots of edge habitat, access to marsh production for small fishes, and good tidal channels connecting to the Bay so that the movement of fish or detritus into and out of the marsh is not inhibited. Although the primary purpose of this restoration is to create a fish nursery, subsidiary goals are the long-term restoration of the full range of natural salt marsh functions which will result in the establishment of a full range of marsh types. So, for example, we want high marsh to provide habitat for such species as the Northern Harrier and Black Rail, and gently sloping creek banks with mud flats to provide forage areas for shore and wading birds.

We also want to create a salt marsh that will maintain itself, one of the prime goals of Ecological Engineering. While we will excavate the main hydraulic channels to facilitate rapid evolution toward a natural marsh, the final placement of the tributaries, shoals, and the slopes of the banks, will all be left to self design. In other words, we will provide a suggestion of

how and where the channels should be shaped and positioned, but will accept whatever changes nature builds into them. Of course, if major disruptions are caused by storms some intervention and modification may be necessary.

Hydraulic/Hydrodynamic Restoration

Our analysis of marsh restoration has shown that the foundation of this process was reestablishment of tidal hydraulics. The tidal hydraulic channels of a marsh are similar to the venous system of a human. The tidal waters are the life's blood of the marsh. If these systems are not successfully reestablished the marsh could not successfully be restored. As previously discussed for unmanaged restorations, restoration can take pathways ranging from the creation of expansive open water areas lacking the essential tidal channels to marshes with a natural mosaic of low to high marsh species. We realize that a marsh is a dynamic environment which responds to small changes in processes. In a natural marsh, the smaller channels in the upper reaches are continually changing position through erosion and accretion processes. It was obvious to the design engineers that it was possible to completely design a channel system from the main stem channels at the tidal inlets to the smallest headwater channels within the high marsh areas. This could be accomplished by overlaying a mosaic of channels similar to the channel systems seen in the adjacent "natural marshes." However, this complete construction approach, antithetical to our ecological engineering approach, would involve intensive construction and be extremely costly. But, we realized that our ecological design approach required a critical examination of the initial channel system to ensure that the basic channel morphology was achieved.

Marsh channels decrease in size from the tidal inlet landward into the high marsh areas. The channels are sinuous and there is evidence that they periodically move by processes similar to river channel processes. Additionally, channels within the upper marsh occasionally appear to form and then become abandoned. Another critical channel parameter

associated with a marsh that has low open water areas is that the channels are not braided. Braided channels in this area are almost exclusively associated with marshes that have high ratios of open water to vegetated marsh plain. These braided channel areas were associated with high/eroding tidal velocities and hydroperiods which ensured the marsh plain remained covered with tidal waters for extended periods of time. Therefore, the two most important design parameters to incorporate into the initial channel design were low maximum tidal velocities and an initial stable channel cross section.

The first design task was to ensure that the channel system had cross sections which would be non-eroding. Examination of the sediments in the adjacent marsh channels and drainage channels within the salt hay farms suggested that the channel velocities should be kept below 1 meter/sec. Establishing a non-eroding condition would also ensure that the initial channel system would provide a large enough tidal prism to the marsh to fully wet and dry the marsh in one tidal cycle and minimize sheet flow velocities which would erode and prevent sedimentation on the marsh plain. This design also minimized the probability that a braided channel morphology would form. Additionally, the design velocity would not impede free access of fish and other fauna into the marsh.

The second design parameter considered was the initial construction cross-section. Our analysis suggested that establishing an initial stable cross section during the construction process was important for ensuring that the marsh evolved towards the desired low to high marsh mosaic and that establishment of marsh function would begin immediately. Our concerns were that if the channel cross section was initially unstable, the channel would not migrate uniformly and that the channel cross section might become constricted somewhere and result in the channel jumping its initial banks and seeking another course. Additionally, the channel morphology is important for fish habitat. A stable cross section permits small and/or juvenile fish to feed and remain in the shallows away from aquatic predators.

Sinuosity is a desired channel trait, but constructing channels with full sinuosity would be prohibitively costly. These conclusions evolved through the design review processes. Questions were asked such as "how do you know the channel will evolve by bending to the right and not left? or How do you determine the distance from bend to bend?" We concluded that constructing the channels with a slight bend or sinuosity would be sufficient because marsh channel/river dynamics would ensure that the channel would evolve with the desired sinuosity.

The next phase of the design process was to quantify these channel parameters for completing detailed construction plans. The channel design process used a four step design analysis to derive the final design. The first step was to complete a detailed topographic analysis of the restoration site. The second step produced a schematic channel and tidal inlet layout. Step three quantified the schematic channel layout producing channel, orientations, densities and quantified the size of the tidal inlets utilizing a 2-Dimension hydrodynamic numerical model. The fourth step verified the channel density and orientation by comparing the final channel design to natural marsh conditions near the restoration site.

The first step, acquiring the detailed topography, gave us a first order look at the type of marsh mosaic that would result after the restoration through a simple comparison between the Mean High Water (MHW) elevation at the site and existing ground surface elevations. The anticipated mosaic was obtained using a first order assumption that high marsh species such as *Spartina patens* would be established in the region between MHW and Mean Higher High Water (MHHW) and low marsh species such as *Spartina alterniflora* would be established between MHW and Mean Low Water (MLW).

Step 2, the initial schematic channel and tidal inlet layout, depended heavily on the detailed topographic data developed in Step 1. Topographically isolated low areas were identified to ensure that a channel would reach them to provide drainage. The initial channel layout was established schematically taking advantage of low areas and swales for the

main stem portions of the restoration site and ensured wetting and drying and drainage for the site. The location of tidal inlets were planned so that each served natural drainage basins or areas separated by existing high topography.

Step 3 of the design analysis required selecting an appropriate 2-Dimensional hydrodynamic numerical model to quantitatively evaluate the channel design and specify channel design parameters such as channel width, depth, number of tidal inlets etc. Our initial design guidance was the following: the maximum velocities should be 60 cm/sec (2 ft/sec) or less, the marsh plain had to fully wet and dry each tidal cycle, and there should be a minimum of tidal lag throughout the system. While these three design parameters were quite straight forward, completing the evaluation was a fairly complex task which required careful evaluation of the numerical model chosen for the design evaluation. Based on the specified evaluation criteria, the hydrodynamic numerical model selected had to have the following minimum capabilities:

- 2-Dimensional depth averaged
- Use a computational grid with complex geometry's and multiple tidal inlets
- Allow grid cells to wet and dry during a tidal cycle

Once selected, the model was configured to simulate hydrodynamic conditions within the restoration site. The computational grid for the numerical model used the results of the detailed topographic data obtained in Step 1. The initial channel size (order) and number of tidal inlets were chosen using the results of the topographic analysis described above. The input specified for the hydrodynamic model included friction coefficients for each order channel, Mannings n, and eddy viscosity. The tidal boundary conditions were chosen as the average tide during the growing season, June and July, because tidal elevations that produced wetting and drying during the peak growing season are controlling criteria for marsh species reestablishment. These relationships were discussed and agreed to by the engineering and biological teams and used to optimize the

hydrodynamic design.

The results of the initial model runs were examined to identify high velocity areas, phase lags, areas of poor drainage, areas of incomplete wetting and drying, and marsh plain hydroperiod. The model results were also used to identify tidal subsections and to verify that the initial schematic channel layout was correct. Additionally, tidal phase lags were critically examined because excessive tidal lags would reduce the possibility of marsh plain sedimentation, produce excessive erosion, and increase the potential for unanticipated channel cut off.

Once the results of each model run were analyzed, the model grid was adjusted by moving the channel locations, adding new channels and increasing or decreasing their size were required. This process was continued until the desired design parameters were achieved. Once the desired channel layout was achieved, the results were examined for practicality. Channels were examined for final size, location within the site, and location to other channels. Several times we found that channels were located too close together, increasing the probability of short-circuiting or cutoff, or they were located too close to the perimeter of the project, increasing the probability of tidal inlet formation. The design was modified to correct such conditions and reanalyzed using the numerical model. Once the channel layout was completed, the restored tidal subsections (drainage areas associated with each tidal inlet) were critically examined for extent of open water areas during the period mid-tide rising to mid-tide falling. In several instances, these results showed that there would be large areas which would only drain at the lowest tidal stages. To increase sedimentation potential within these areas, fetch berms were designed to reduce the possibility of locally generated wind waves which could produce local turbulence and inhibit or prevent deposition of fine grained sediment on the marsh plain.

The fourth step in the design specification process started with a geomorphologic examination of marsh channels in adjacent marshes which were "natural" and exhibited full marsh function. This analysis identified

design parameters such as the desired marsh channel density and transitions from the large main stem channels (first order) to the small head water channels (fourth order) within the upper marsh. We arbitrarily described the channel dimensions as first through fourth order for ease of reference. It became clear that our design could not specify one size channel for the entire marsh restoration project. The restoration design required a hierarchy of channels and outlined the desired transitions from the main stem to the smaller head water channels. Additionally, the number of tidal inlets and subdrainage areas within the marsh were determined. These results were compared to the design derived within Step 3. This step was viewed as one of the most important steps in the design process because it provided a quality control check of the restoration design. Again, the design was altered when inconsistencies were found.

The design processes was time consuming and required many model runs. This was necessary to ensure that the resulting design was close to natural conditions. As we had stated, we realized that the marsh would begin to evolve almost as soon as tidal flows were reestablished. This was an anticipated and desired process which must occur in a healthy marsh. However, by using the complex design analysis outlined above, our goal was to minimize counter productive evolution such as, channel cut offs which could isolate a section of the marsh from wetting and drying each tidal cycle.

Conclusions

We have described a long and detailed process by which we have designed the steps for initial construction involving the restoration of diked salt hay marshes. We have shown by changing them from isolated, artificially maintained systems being degraded by the invasion of undesirable plants, to natural, fully functional salt marshes whose functions will contribution to the productivity and fish habitat in Delaware Bay. The approach depends upon ecological engineering (self-design) once the initial

conditions favoring natural restoration within a reasonable, agreed upon time frame were designed. It is important in such a restoration that the resulting ecosystem should be self maintaining and stable within the natural limits of that particular type of ecosystem. This is desirable from the position of the organizations responsible for the restoration. It is much more important to society in general, that regardless of the fate of the responsible organizations, the restored ecosystem should be able to maintain itself into the indefinite future. We believe with careful planning we will achieved just that. The detailed monitoring of the restoration that is planned will provide the proof.

Acknowledgments:

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RESTORING UPPER SADDLE CREEK: LINKING THE GREEN SWAMP AND PEACE RIVER

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ABSTRACT

The Saddle Creek watershed forms the headwaters of the Peace River, and comprises approximately 165 km^2 in Polk Co., Florida between Lakeland and Auburndale. Saddle Creek is the northernmost tributary of the Peace River, and once served as an extension of the river into the Green Swamp. The original ecosystems included mixed hardwood swamps, cypress domes, shallow marshes, and wet pine flatwoods. Almost none of the original creek and floodplain remains in the area. Historical alterations have included phosphate mining, agricultural use, and stream channelization. The Upper Saddle Creek basin has 12 Nonmandatory Program parcels that are either funded projects underway or are unreclaimed and eligible for funding through the Florida Department of Environmental Protection (FDEP) Nonmandatory Land Reclamation Trust Fund. Three of the eligible parcels are located at Tenoroc, while the remaining 9 are on private land to the north owned by the Williams Acquisition Holding Company. The FDEP is incorporating an ecosystem management approach to mine planning. The strategy at Saddle Creek was for FDEP (in cooperation with other agencies) to plan reclamation on an ecosystem-wide basis rather than as individual mine units. The resulting reclamation plan will improve hydrology and form a wildlife corridor extending from Tenoroc to the Green Swamp via an underpass on I-4. This and future reclamation and preservation on Tenoroc will provide improved fish and wildlife habitat, and an ecological and recreational greenway

connection from Green Swamp to Lake Hancock.

INTRODUCTION

The Florida Department of Environmental Protection (FDEP) and the phosphate industry is currently implementing an ecosystem approach to phosphate mine reclamation planning in Central Florida. In the past, reclamation was traditionally designed on an individual mine parcel or program basis. The result was often a fragmented landscape of scattered plant communities and isolated habitats (Fig. 1). An ecosystem approach involves developing a reclamation plan that incorporates the recreation of environmental systems and drainage basins for the entire mine. This results in a more natural system for a larger land area. The advantage of this approach is that plant communities are contiguous from parcel to parcel with fewer gaps in between (Fig. 2). This tends to keep the developing ecosystem(s) separated from, but coordinated with any possible post-reclamation land use development planned for the area.

Central Florida is faced with an ever expanding population and increasing rate of urbanization (Long & Orne 1990). Naturally occurring ecosystems are threatened with the possibility of becoming isolated and fragmented with immigration and emigration of wildlife populations severely restricted. In addition, reclaimed land and land requiring reclamation in the future, will need a source of plant and animal colonizers for the replacement habitats. Reclaimed lands are generally unsuited for most intensive land uses other than agriculture and livestock grazing. As a result, they present a unique opportunity to establish a system of habitat preserves and linking corridors in the southern phosphate district.

FDEP is incorporating an Integrated Habitat Network (IHN) in mine reclamation planning for the entire southern phosphate district (Cates 1992) (Fig. 3). This plan is intended to provide mitigation for mining

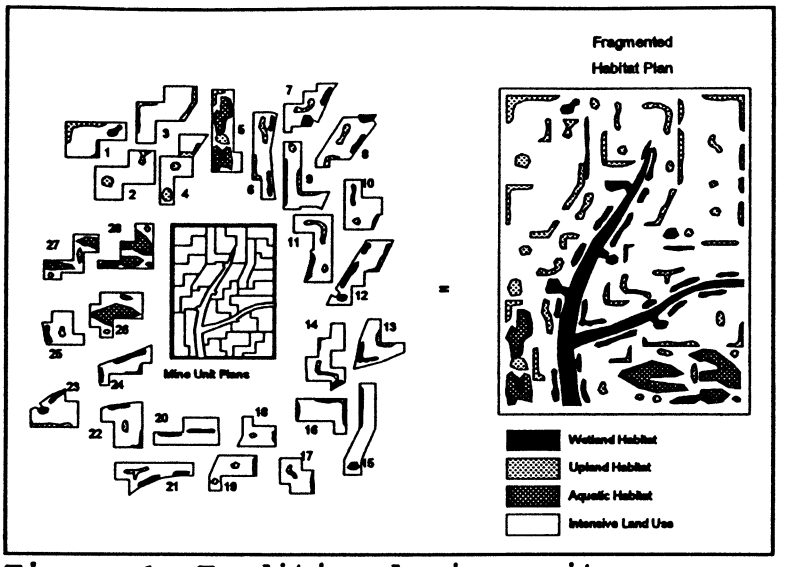

Figure 1. Traditional mine unit reclamation planning.

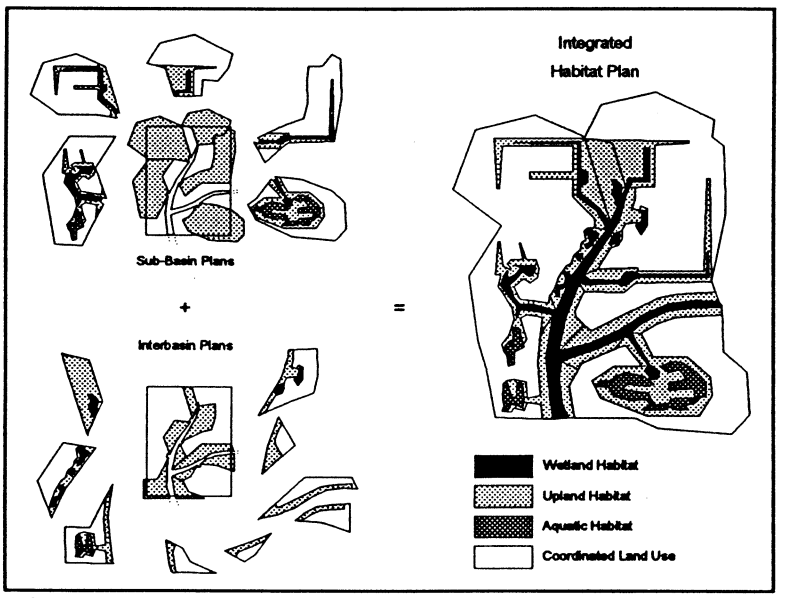

Figure 2. Basin reclamation to create an integrated habitat plan.

related impacts to fish and wildlife, as well as to produce a fully functional and persistent post-mining ecosystem capable of supporting the same complement of species as existed prior to mining. The IHN has 3 components: 1) a core reserve of protected habitat, 2) surrounding complimentary habitat, and 3) connections to habitats outside the planning

area (King & Cates 1994).

The IHN core reserve habitat consists of the bottomlands of rivers and streams. In the southern phosphate district, rivers and streams are ideal core habitats because they are unsuited to most intensive land uses, are relatively protected from development by wetland regulation, and are the least likely sites to be permitted for mining. They also provide habitat for a cross-section of a region's fish and wildlife (King & Cates 1994). The core habitat reserve in the southern phophate region includes the five major river systems that drain the area: the Peace, Alafia, Little Manatee, Manatee, and Myakka Rivers.

In order to function optimally as core reserve habitat, the riverine reserve should be as large and extensive as possible. A large reserve can support more diversity and greater wildlife populations. Therefore, the second component of the IHN is to surround the core reserve with complementary habitat not only to expand the reserve habitat, but also to provide a buffer between intensive land uses and the riverine systems.

The third component of the IHN is a series of protected habitat corridors linking the individual river systems. The goals of the habitat corridors is to prevent the core reserves from becoming isolated ecosystems and to extend protected core habitat into otherwise isolated reclamation tracts. Connecting individual river systems with protected habitat corridors would create the largest possible core reserve in the region.

Currently, FDEP is in the process of establishing core reserve habitat in the southern phosphate region. The problem is that there are gaps in the Peace and Alafia Rivers where historically weaker regulatory permitting has allowed portions of the rivers to be mined over. Of the utmost priority is the establishment of the Peace River core reserve habitat which ultimately will extend from the Green Swamp at the northern end to Charlotte Harbor to the south. The headwaters of the Peace River including the northernmost tributary Saddle Creek, has been subject to severe environmental alteration. In addition, the Saddle Creek basin has been severed from its historical

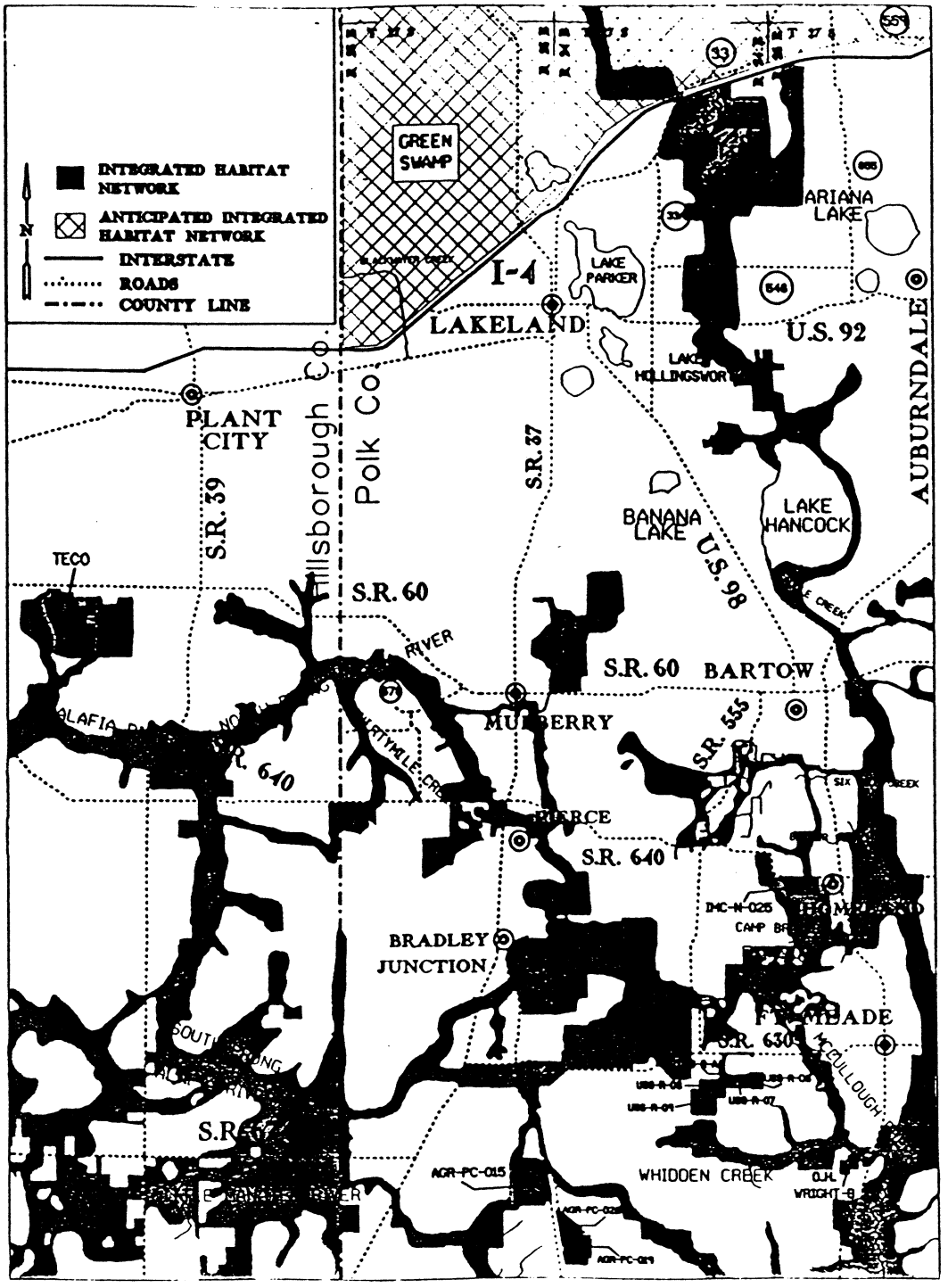

Figure 3. The Florida Department of Environmental Protection Bureau of Mine Reclamation's system of corridors/greenways in the Southern Phosphate District: The Integrated Habitat Network.

extension into the Green Swamp by Interstate 4.

This paper provides an example of ecosystem management and mine reclamation planning, and some of the methodology involved in establishing IHN core reserve habitat. The concepts of ecosystem management and the IHN are the 2 basic principles governing the reclamation of the Saddle Creek drainage basin.

STUDY SITE

The Saddle Creek watershed covers 165 km^2 in Polk Co., Florida between Lakeland and Auburndale, and forms the headwaters of the Peace River (Fig. 4). Saddle Creek is the northernmost tributary of the Peace River. Historically, Saddle Creek once served as an extension of the Peace River into the Green Swamp. The original ecosystems included cypress domes, bay heads, shallow marshes, wet pine flatwoods, mixed hardwood swamps, and hammocks. These areas were at one time interconnected and capable of detention and filtration of excess runoff and seepage. Presently, virtually no remnant of the original stream and floodplain remains in the area. Historical alterations have included extensive phosphate mining, agricultural development, and stream channelization (King et al. 1994).

In 1954, the upper Saddle Creek region began to be strip-mined for phosphate (Fig. 5). Over the years, several mining companies have been active in the area including American Cyanamid, Agrico, Coronet Chemical Company, and the Bordon Chemical Company 1975, approximately 70% of the region had been either strip-mined or used for clay waste disposal and left unreclaimed.

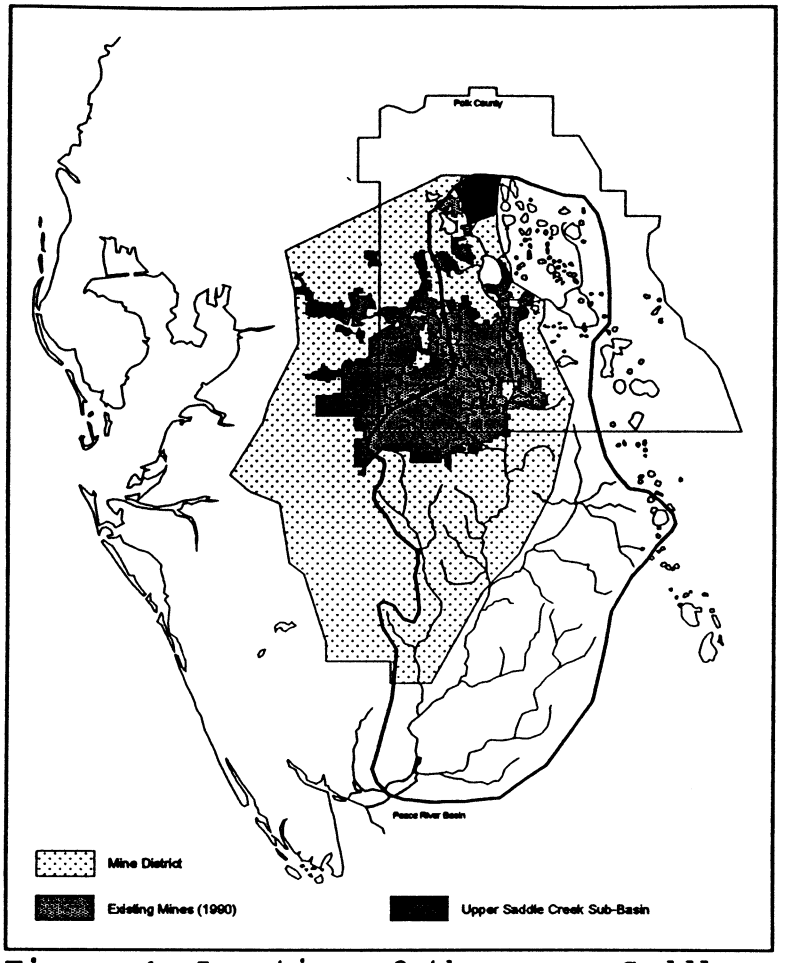

Figure 4. Location of the upper Saddle Creek region.

Although mining was completed by 1975, phosphate matrix continued to be pumped from the Ebersbach mine (south of Tenoroc) to benificiation facilities and clay settling areas at Saddle Creek until 1986 (Sharkey Watson, per. comm.). Mining activities, particularly the dams of clay settling areas, have impounded large areas of the basin and reduced the river's water supply. Former streams and floodplains were replaced by diversion ditches and surface drainage was rerouted through mine pit lakes connected into groundwater. The effect of these alterations was to exacerbate recognized problems in the upper Peace River: low total flow volume, localized flooding, and poor water quality (King et al. 1994). Also, upon the completion of Interstate 4 in 1962, a potential wildlife

barrier was created along the Florida peninsula from the east to west coast and isolated the Green Swamp from the Peace River.

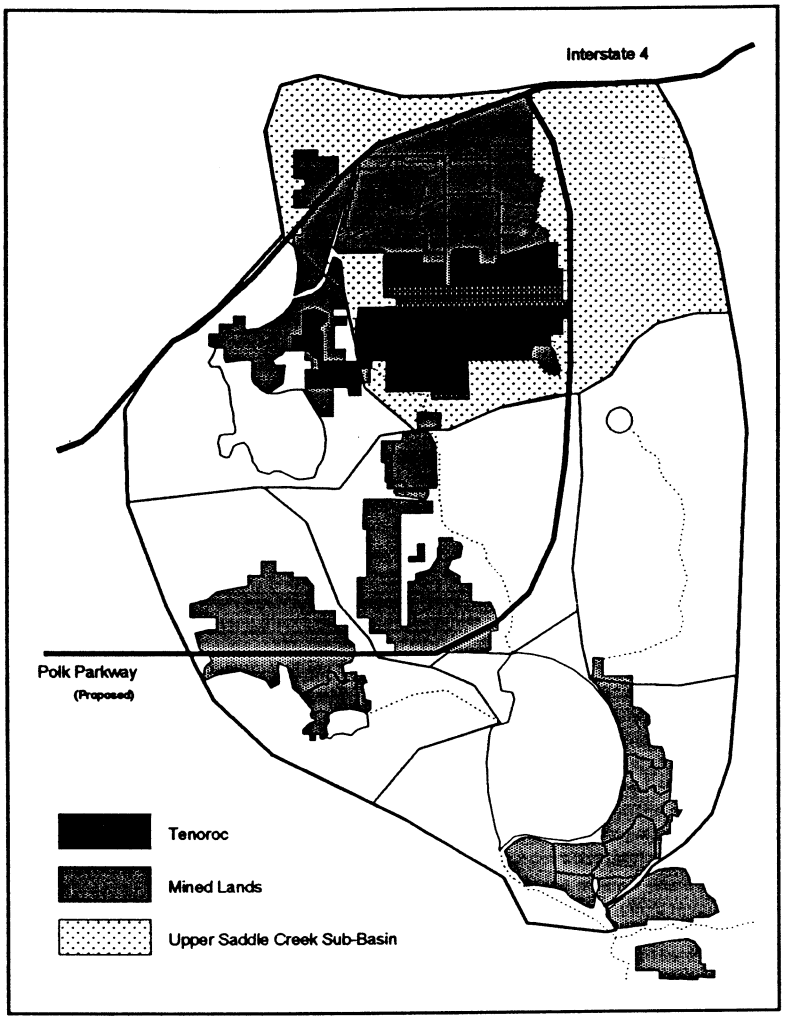

Figure 5. Tenoroc and the Saddle Creek sub-basin.

The Williams Acquisition Company has been the major landowner north of Tenoroc since ca 1973, when the land was purchased from Continental Oil. Currently on the Williams Co. property there are 9 reclamation programs totaling 1222 ha (3017 ac) in different stages of completion. The State of Florida acquired Tenoroc from the Borden Chemical Company in 1982. This 2446 ha (6040 ac) fish management area began allowing public fishing access in 1983. Of the 1293 ha (3192 ac) eligible for funding

through the Nomandatory Land Reclamation Trust Fund, 799 ha (1972 ac) have already been reclaimed leaving 3 parcels totaling 494 ha (1220 ac) left to reclaim.

The goals of the Saddle Creek drainage basin reclamation project are: protect remaining habitat in the mined out area at and near Tenoroc; replace stream forest and wetland habitat lost due to mining; extend reclaimed forest through the mined area to a potential bridge location at I-4; restore Saddle Creek and its floodplain within the former mine area; enhance flow from each sub-basin by unimpounding former mine areas and minimizing flows through mine pit lakes; and enhance water detention and treatment within each sub-basin by directing flows through natural and reclaimed wetlands.

METHODS

There are two means available to accomplish the goals of the Saddle Creek reclamation. The first is that in the Saddle Creek drainage basin there are several projects proposed or underway that require environmental mitigation (Fig. 6). These projects include three Developments of Regional Impact (Bridgewater DRI, Williams Parkway Center DRI, Polk Commerce Center DRI), two new highways (Polk Parkway and the East-West Expressway), renovation of I-4 (Florida Department of Transportation), a natural gas pipeline (Sunshine Pipeline Company), and a 500 kv transmission line (Florida Power Corporation).

Secondly, there are 12 nonmandatory reclamation program parcels in the Saddle Creek basin that are eligible for funding through FDEP's Nonmandatory Land Reclamation Trust Fund. As noted above, 3 of the program parcels are located on Tenoroc, while the remaining nine are located on private land just north of and adjacent to Tenoroc owned by the Williams Acquisition Holding Company. Reclamation plans for these parcels can be designed or modified to improve hydrology and wildlife habitat for the watershed.

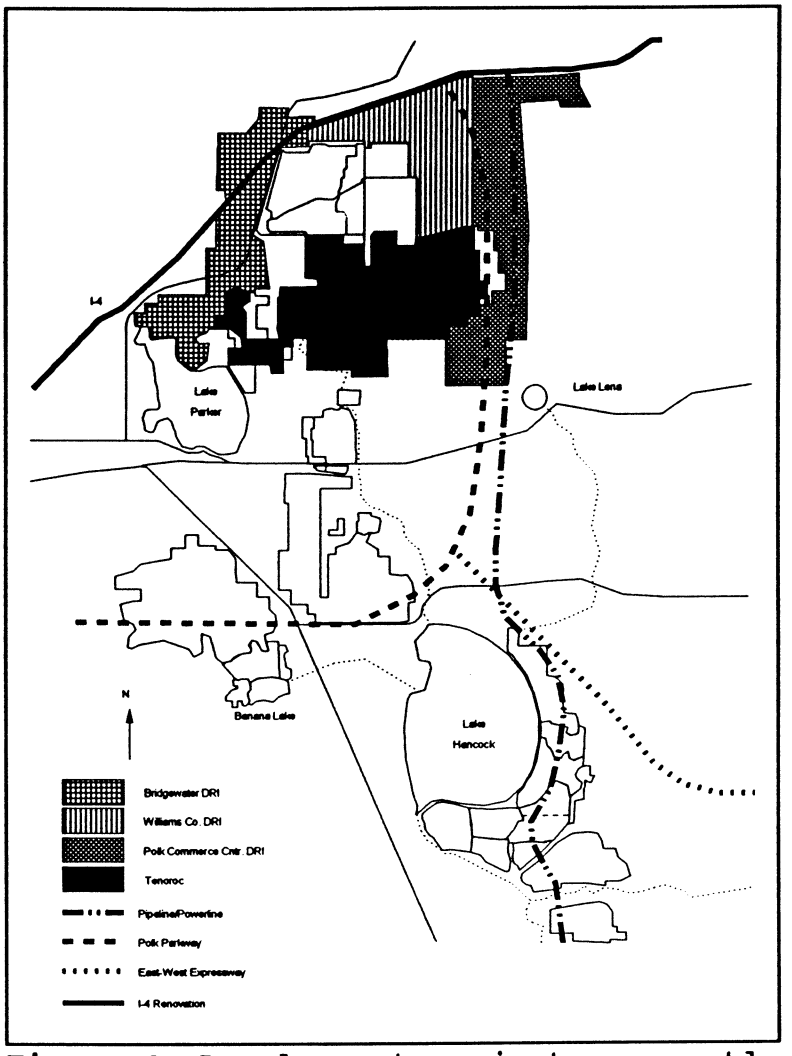

Figure 6. Development projects presently being planned in the upper Saddle Creek basin.

Conventional mitigation for the above linear development projects will provide little or no ecologic/hydrologic restoration benefit for Saddle Creek. In order to develop a comprehensive ecological restoration plan for the entire watershed, cooperation needed to be established between the

regulatory agencies involved in overseeing permitting and mitigation requirements. To achieve this interagency cooperation, a Memorandum of Understanding (MOU) was signed by FDEP, Florida Game and Freshwater Fish Commission (GFC), Southwest Florida Water Management District (SWFWMD), U.S. Army Corps of Engineers (ACOE), Florida Department of Transportation (FDOT), and the National Biological Survey (NBS).

Also instrumental in developing the restoration plan is the participation of the Williams Acquisition Holding Co. The Williams Co. agreed to set aside land reclaimed and funded through the Nonmandatory Land Reclamation Trust Fund to be used as wildlife habitat for the corridor system through the property.

Thus the Upper Peace River Ecosystem Planning Committee (UPREPC) was formed and includes representatives from each regulatory agency and the Williams Co. It is the responsibility of UPREPC to coordinate project goals and mitigation requirements, and develop a Saddle Creek ecosystem restoration plan for reclamation and mitigation.

The first priority in restoring the upper Saddle Creek watershed was to determine how to design and reclaim the landform to improve quantity and quality of flow into Tenoroc, and eventually down Saddle Creek and into Lake Hancock. As noted, the watershed north of, and including Tenoroc, has been impounded by the old dams of clay settling areas. Also former streams and floodplains have been replaced by diversion ditches. A study was necessary to determine present hydrologic conditions and drainage patterns. The Florida Institute of Phosphate Research (FIPR) has awarded a research grant to the University of South Florida to model the present hydrology, and evaluate existing and proposed interconnected wetland systems to aid in the wetland design process on the Williams Co. property and Tenoroc. Once the current system is modeled, several options appear to be feasible to unimpound the clay ponds, recreate sub-basins, and route flow through reclaimed wetlands.

The next task was to evaluate the 9 nonmandatory program parcels north

of Tenoroc funded or eligible for funding through the FDEP Nonmandatory Reclamation Trust Fund. Because the reclamation revegetation plans were submitted to the FDEP Bureau of Mine Reclamation prior to the formation of UPREPC, the plans did not incorporate a wildlife corridor into the overall design. Most of the revegetation plans were designed on a program unit basis resulting in a fragmented post-reclamation landscape. It was evident that in order to form a contiguous corridor or greenway through the entire Williams Co. property, it would be necessary to modify the existing plans.

FDEP biologists and engineers from the Bureau of Mine Reclamation together with GFC biologists from Tenoroc met and designed plans for establishing a wildlife corridor through the Williams property. This was accomplished mainly by moving the proposed revegetation within each individual program to strategic areas where it would form a contiguous strip with adjacent programs as long as certain criteria were met. The first criteria was to enhance the retention and filtration capabilities of the region. Therefore the general corridor route follows the proposed drainage patterns of the reclaimed areas where possible.

Secondly, the plant species must be appropriate for the elevation i.e. wetlands, uplands,...etc. Once the modified plans were approved by UPREPC and the landowner, FDEP biologists met with the reclamation engineer regarding redesigning the revegetation plans to form the corridor, as well as adding additional plantings and species to enhance it.

Interstate 4 serves as an effective ecological and hydrological barrier between Saddle Creek and the Green Swamp. The planned renovation and significant widening of the road corridor will exacerbate the problem. The widening of I-4 will impact several wetlands within the upper reaches of the Saddle Creek watershed resulting in mitigation requirements. In order to fulfill these requirements, FDOT has agreed to the construction of a wildlife underpass during the renovation. The underpass will connect the northern belt of the wildlife corridor with a stand of cypress north of I-4. This underpass along with future land acquisitions will reconnect the

Green Swamp to the Saddle Creek watershed.

Also, construction of the Polk Parkway will impact approximately 200 acres of wetlands within the Saddle Creek watershed resulting in mitigation requirements. As mitigation funds from this and other linear and development projects described above become available, further enhancement of hydrology and wildlife habitat within the watershed can be accomplished through reclamation.

RESULTS

The restoration of the Upper Saddle Creek watershed is ongoing at the time of this writing. Most of the work to date has been concentrated on the Williams property were the 9 nonmandatory programs are currently undergoing various stages of reclamation. Two programs will be ready for revegetation in September 1996 when the first segments of the corridor will be planted. The work will continue over the next several years as reclamation is completed and mitigation funds become available.

CONCLUSIONS

The Saddle Creek project incorporates the concepts of ecosystem management, greenways, and mitigation alternatives. It represents an innovative approach to reclamation planning through interagency cooperation. It is hoped that this pilot project will become a model for other restoration projects in the future.

The anticipated benefits of this project will include the reconnection of the Green Swamp and the Peace River for the first time since 1962, and an ecological and recreational greenway from Green Swamp to Lake Hancock (Fig. 7). Additional benefits include improved drainage and water quality for Saddle Creek and the Peace River, reduced potential for flooding impacts downstream, improved fish and wildlife habitat for the entire

system, and improved recreational opportunities within the Tenoroc Fish Management and Environmental Area.

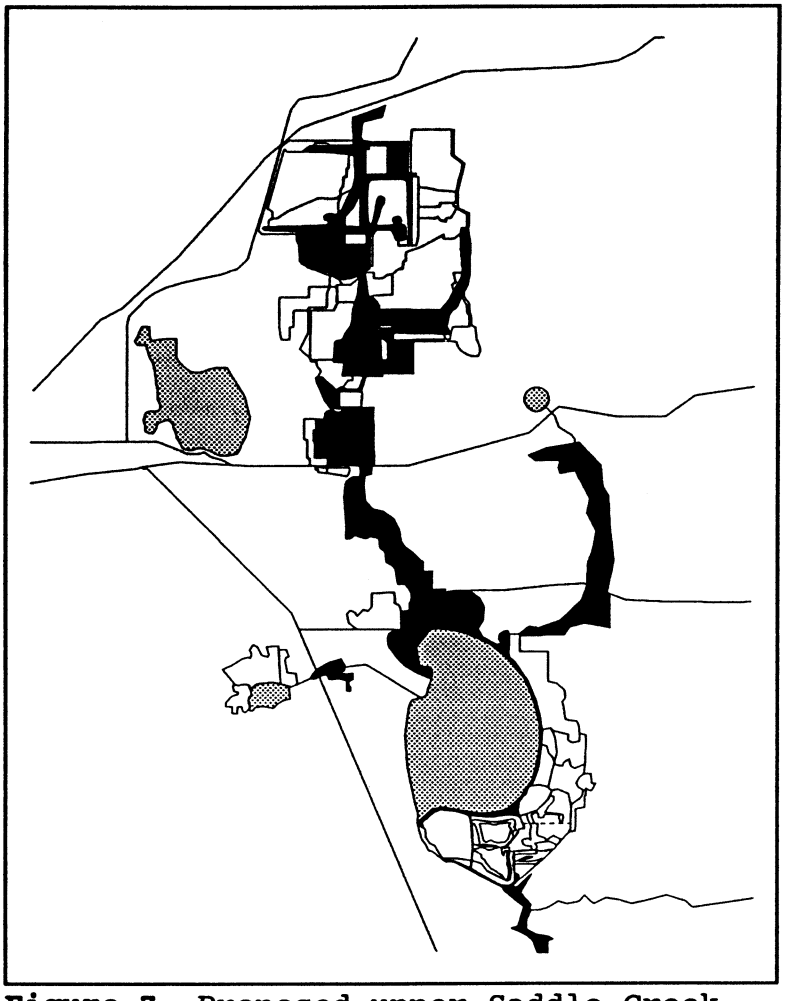

Figure 7. Proposed upper Saddle Creek habitat system.

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WETLANDS RESTORATION ALONG THE VETERANS EXPRESSWAY IN HILLSBOROUGH COUNTY

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ABSTRACT

The Veterans Expressway Wetlands Restoration Project was undertaken to restore a wetland area impacted by a discharge of sandy sediment during roadway construction. The project area is located in Hillsborough County adjacent to the Veterans Expressway, north of Linebaugh Avenue in Section 24, Township 28 South, Range 17 East. Approximately 0.4 hectares of the wetland was impacted. Following the discharge, severe tree mortality in the affected area was observed.

Restoration efforts included removal of the sediment and replanting the most heavily impacted areas of the wetland with native trees and herbaceous vegetation. Nuisance species such as primrose willow (*Ludwigia peruviana*) were removed from the site. Tree snags were left standing to provide habitat for cavity nesting birds.

Monitoring data indicate that the wetland is doing well. However, planted fern species have been replaced by soft rush (*Juncus effusus*) as the dominant herbaceous species. This appears to be the result of higher light availability in the restored area. The standing tree snags were successful in attracting woodpeckers.

INTRODUCTION

This paper is a presentation of BCI's successful efforts to restore an

impacted wetland located in central Florida. Data presented in this paper was extracted from a monitoring report for monitoring activities conducted in December 1995, approximately one year after the initiation of the restoration efforts.

In October 1994 BCI was requested by the FDOT Turnpike District and HNTB (Howard, Needles, Tammen and Bergendoff, Inc.), the general engineering consultant for the Veterans Expressway Project, to assist with the restoration of a wetland located adjacent to the project. BCI was instructed to develop a plan to restore the wetland, oversee the restoration efforts and monitor the restored wetland for one year.

Study Site

The site is located in Section 24, Township 28 South, Range 17 East and is positioned to the west of the Veterans Expressway approximately 0.5 kilometers south of the junction between the Veterans Expressway and Linebaugh Avenue from centerline stations 381 to 382. Approximately 0.4 hectares of the wetland had received fill washed in from the roadway embankment and from a water control structure upstream. Following this event hardwood trees located in an approximately 0.1 hectare area died.

Restoration Activities

Restoration activities at the site took place from November 1994 through May 1995. Mechanical and manual removal of sediment was performed within the restoration boundary. Sediment was removed from designated areas based on sediment thickness, drainage patterns, and vegetative communities. Sediment was not removed from areas which would have resulted in the disturbance of mature trees or significant amounts of herbaceous plant species. Sediment removal was accomplished using tracked and rubber-tire earth moving equipment in areas were such equipment could be used without significant disturbance to existing

vegetation and soils. Hand removal methods were employed in areas where there was a potential for disturbance of the existing plant community. All excavated sediment was removed from the restoration area and disposed of off-site.

Once sediment removal was completed, the site was graded to restore topography and hydroperiod appropriate for the existing communities. The dead tree snags in the restoration area were left standing to provide nesting and foraging habitat for woodpeckers and other cavity nesting birds. In addition, exotic/nuisance plant species that had become established in the restoration area were removed by hand.

Revegetation

The most heavily impacted areas of the wetland were replanted with native trees and herbaceous vegetation. This portion of the restoration site consisted of areas adjacent to the right-of-way, extending 17 to 50 meters into the wetland. Trees were planted on 3-meter centers and consisted of the following species:

50 Red Maple (*Acer rubrum*)
80 Bald Cypress (*Taxodium distichum*)
30 Laurel Oak (*Quercus laurifolia*)

The following herbaceous species were planted on 1 meter centers:

850 Pickerelweed (*Pontederia cordata*)
2000 Royal Fern (*Osmunda regalis*)
2000 Swamp Fern (*Blechnum serrulatum*)

Maintenance Activities

Maintenance activities were conducted on a monthly basis by BCI personnel. These activities were restricted to hand removal of nuisance species including primrose willow (*Ludwigia peruviana, L. octovalis*), cattail (*Typha latifolia* and *T. domingensis*), seedbox (*Sesbania* spp.) and Chinese tallow tree (*Sapium sebiferum*). Once pulled, nuisance species were disposed of in a landfill off site.

MONITORING METHODOLOGY

Field and data analysis techniques were obtained from Mueller-Dombois and Ellenberg (1974). Plant identifications were made from Clewell (1985) and Long and Lakela (1971). Monitoring data presented in this report was collected on December 4, 1995. Seven transects were established in the restoration area. Herbaceous vegetation was sampled in 77 one square meter quadrats established at 8 meter intervals along each transect. One transect was 107 meters long with 15 quadrat locations, three transects were 91 meters long with 13 quadrat locations, two transects were 61 meters long with nine quadrat locations, and one transect was 31 meters long with five quadrat locations. Two of the 91 meter long transects were belt transects (3 meters wide) used for sampling tree data in the restoration area. One transect was established in the replanted zone of the restoration area. The other transect was established in the naturally forested portion of the restoration area. In total, the tree transects covered approximately 15% of the site.

Trees

Tree survival rates were determined by direct counts in two belt transects. Percent survival of trees was determined by the number of living trees within the transect area divided by number of living trees counted during the initial monitoring event.

For the planted trees in transect 1, height, diameter at breast height (DBH), and crown cover measurements were taken. Diameter at breast height was only measured on those trees which have reached the 1.4 meter height requirement. Crown cover of planted trees was determined using the formula:

$$\left(\frac{D1 + D2}{4}\right)^2 \pi$$

D1 and D2 represent crown diameter measurements at right angles for a given tree. Percent canopy cover is determined by the total crown cover for all trees divided by the transect area. Canopy coverage of natural growing trees in transect 7 was estimated by visually quantifying the canopy cover at 15 meter intervals along a 91 meter long transect located in the naturally forested portion of the restoration area.

Herbaceous and Shrub Vegetation

Percent coverage was determined by direct visual esti-mation of herbaceous plant species within individual quadrats along the seven transects running the length of the restora-tion area. Percent herbaceous coverage was determined by applying the formula:

$$\frac{Species\ 1\ area + Species\ 2\ area + etc}{Quadrat\ Area} \times 100$$

Sampling parameters included percent coverage by desir-able wetland species (planted and recruited), upland species, nuisance/exotic species, relative abundance and species compo-sition.

RESULTS AND DISCUSSION

Hydrologic conditions at the site were good. Much of the site was inundated with water or contained saturated soils. The creek was flowing

at the time of the monitoring event.

Tree growth data, represented in Table 1, depicts the status of the planted trees. Bald cypress (*Taxodium distichum*) trees had an average height of 2.4 meters, an average DBH of 2.0 centimeters, and an average canopy cover of 0.6 square meters. Red maple (*Acer rubrum*) trees averaged 1.8 meters in height, 1.0 centimeters in DBH, and 0.3 square meters in canopy cover. One live oak (*Quercus virginiana*) was found with a height of 1.3 meters and an average canopy cover of 0.1 square meters. Planted trees were providing 3.3% canopy cover in the replanting zone. Most of the trees were in good condition and were entering into the dormant stage for the winter season. Tree survival in the replanting zone is 95%.

Table 1
Tree Growth Data (Transect 1 Only)

Species	Avg. Height	Avg. Diameter at Breast Height (DBH)	Avg. Canopy Cover
Bald Cypress (*Taxodium distichum*)	2.4 m	2.0 cm	0.6 m^2
Red Maple (*Ace rubrum*)	1.8 m	1.0 cm	0.3 m^2
Live Oak (*Quercus virginiana*)	1.3 m	-	0.1 m^2

NOTE:
21 Trees Sampled
11 Bald Cypress
8 Red Maple
1 Live Oak
1 Laurel Oak (dead)

Percent Canopy Cover = 3.3%
(Total canopy cover = 9.1m^2/transect area = 279m^2)
Tree Survival = 20 living trees counted/21 live trees expected = 95% survival

Pond cypress (*Taxodium ascendens*), bald cypress, red maple and wax myrtle (*Myrica cerifera*) were found in the transect running through the established forest. Fifty-five trees were observed with most being in good condition and going dormant for the winter season (see Table 2). Only one pond cypress tree appeared to be stressed. Canopy cover in the established forest was 50%.

Table 2
Tree Condition Data (Transect 7 Only)

Common Name	Scientific Name	Trees Counted	Tree Condition
Pond Cypress	*Taxodium ascendens*	11	10 Good 1 Stressed
Bald Cypress*	*Taxodium distichum*	2	2 Good
Red Maple	*Ace rubrum*	32	32 Good
Wax Myrtle	*Myrica cerifera*	10	10 Good

NOTE:
* = Planted trees, not trees from the original natural forest

Percent Canopy Cover = 50% (Canopy cover was determined by visually quantifying the canopy cover at 15 meter intervals along the transect and taking the average of the values).

Fifty-one (51) species of herbaceous taxa were counted in the restoration area (see Table 3). Herbaceous vegetation was providing approximately

61.8% of the cover in the restoration site. The dominant vegetation was soft rush (*Juncus effusus*), providing 16.3% cover. Subdominant species included pickerelweed (*Pontederia cordata*), red ludwigia (*Ludwigia repens*), flatsedge (*Cyperus* spp.), dayflower (*Commelina* spp.), water pennywort (*Hydrocotyle umbellata*), Virginia chainfern (*Woodwardia virginica*), and royal fern (*Osmunda regalis*). The cover provided by these subdominant species was collectively 34.3%. Pickerelweed was predominantly found in the areas inundated with water, while soft rush was dominant in the higher, less wet areas.

Coverage by nuisance and/or exotic species was low at the restoration site, approximately 3.9%. Nuisance and/or exotic vegetation at the site included primrose willow (*Ludwigia peruviana*), Chinese tallow tree (*Sapium sebiferum*), hempvine (*Mikania scandens*), sesbania (*Sesbania* spp.), Mexican seedbox (*Ludwigia octovalis*), alligator weed (*Alternanthera philoxeroides*), and seedbox (*Ludwigia* spp.).

CONCLUSIONS

Hydrological conditions at the site were good and were favorable for supporting a functional wetland. Most of the restoration site was inundated with water and the creek was flowing.

Planted trees in the restoration area were in good condition and appeared to be going dormant for the winter season. Numerous red maple seedlings had germinated within the mitigation site during the 1995 growing season. Some of these recruited trees were growing and may contribute to the reforestation of the project site. Naturally occurring trees in the established forest were also in good condition. A total of 55 pond cypress, bald cypress, red maple and wax myrtle trees were observed in the naturally forested portion of the restoration area. Only one tree was stressed. There were, however, numerous stressed and dying trees outside of the transects. This may be the result of a pathogenic fungus and the damage due to fallen limbs and tree tops.

Table 3
Percent Herbaceous Cover - Quadrat Sampling

Common Name	Scientific Name	Total Relative Cover	Common Name	Scientific Name	Total Relative Cover
Soft Rush	*Juncus effusus*	16.1	Lance-leaf Loosestrife	*Lysimachia lanceolata*	0.6
Pickerelweed	*Pontederia cordata*	13.3	Maidencane	*Panicum hemitomon*	0.5
Royal Fern	*Osmunda regalis*	6.0	Beakrush	*Rhynchospora* spp.	0.4
Dayflower	*Commelina* spp.	5.1	Dog Fennel	*Eupatorium capillifolium*	0.3
Red Ludwigia	*Ludwigia repens*	3.6	Swamp Fern	*Blechnum serrulatum*	0.3
Water Pennywort	*Hydrocotyle umbellata*	2.8	Clover	*Trifolium* spp.	0.3
Primrose Willow*	*Ludwigia peruviana*	2.4	Arrowhead	*Sagittaria lancifolia*	0.3
Virginia Chainfern	*Woodwardia virginica*	1.9	Mistflower	*Conoclinium coelestinum*	0.3

Common Name	Scientific Name	Total Relative Cover	Common Name	Scientific Name	Total Relative Cover
Flatsedge	*Cyperus* spp.	1.1	Groundsel Tree	*Baccharis glomeruliflora*	0.3
Hempvine*	*Mikania scandens*	0.8	Seedbox*	*Ludwigia* spp.	0.3
Water Fern	*Salvinia rotundifolia*	0.8	Smartweed	*Polygonum punctatum*	0.2
Capeweed	*Lippia* spp.	0.2	Bedstraw	*Galium* spp.	0.1
Blackberry	*Rubus* spp.	0.2	Black Willow	*Salix nigra*	0.1
Grass	*Poaceae* spp.	0.2	Bahia Grass	*Paspalum notatum*	0.1
Red Maple	*Ace rubrum*	0.2	Water Hyssops	*Bacopa monneiri*	0.1
Sedge	*Carex* spp.	0.1	Pepper Vine	*Ampelopsis arborea*	0.1
Swamp Willow	*Salix caroliniana*	0.1	Sourgrass	*Oxalis* spp.	0.1

Asian Coinleaf	*Centella asiatica*	0.1	Elderberry	*Sambucus canadensis*	0.1
Joint Paspalum	*Paspalum distichum*	0.1	Ragweed	*Ambrosia* spp.	0.1
Camphor Weed	*Pluchea* spp.	0.1	Wax Myrtle	*Myrica cerifera*	0.1
Buttonweed	*Diodia* spp.	0.1	Poison Ivy	*Toxicodendron radicans*	0.1
Foxtail	*Setaria geniculata*	0.1	Mock Bishop's Weed	*Ptilimnium capillaceum*	0.1

Table 3
Percent Herbaceous Cover - Quadrat Sampling

Common Name	Scientific Name	Total Relative Cover
Cypress Knee	*Taxodium ascendens*	0.1
Unknown 1	-	0.1
Unknown 2	-	0.1
Alligator Weed*	*Alternanthera philoxeroides*	0.1
Chinese Tallow Tree*	*Sapium sebiferum*	0.1
Sesbania*	*Sesbania* spp.	0.1
Mexican Seedbox*	*Ludwigia octovalis*	0.1

Common Name	Scientific Name	Total Relative Cover
Wood	-	2.9
Bare Ground	-	26.4
Water	-	9.9
	TOTAL	100.4

*Nuisance and/or exotic species

Herbaceous vegetation was providing 61.8% coverage in the restoration area. Planted and recruited species were healthy and growing. Considering that the vegetation was measured at the end of the growing after the first frost, coverage by herbaceous vegetation was good. Soft rush and pickerelweed were providing the greatest cover of the herbaceous species. Nuisance and/or exotic vegetation was providing only 3.9% coverage in the restoration area.

Much wildlife was observed utilizing the restoration area. Woodpecker marks were present on many of the dead trees in the forest. Many small fish were observed in the creek and numerous green treefrogs (*Hyla cinerea*) and bullfrogs (*Rana catesbeiana*) were observed in the restoration site. A great egret (*Casmerodius albus*) was frequently seen in the restoration area and a woodstork (*Mycteria americana*) was observed flying over head.

Maintenance activities were performed in December 1995, after the monitoring event. Nuisance and/or exotic vegetation was removed from the restoration area. With continued regular maintenance the site should succeed into a healthy wetland forest.

ACKNOWLEDGEMENTS

We wish to thank all the staff at BCI, HNTB and the FDOT Turnpike District associated with the project for their support. The authors of this paper are especially indebted to Muriel Berry for her clerical and editorial services. We are also indebted to Allan Schreiber (BCI Environmental Division Chief) and Wayne Ericson (BCI's Vice President) for their review and comments concerning the paper.

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WETLANDS MITIGATION MONITORING - A CRITICAL COMPONENT OF WETLAND IMPACT REGULATION AND MITIGATION DESIGN

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ABSTRACT

Broward County's Department of Natural Resource Protection (DNRP) has been regulating dredge and fill activities and wetlands impacts since 1984. The field of wetlands regulation, mitigation design, and compliance is evolving rapidly and requires responsive administration. DNRP, through its wetlands mitigation monitoring program, has identified critical permitting issues relating to mitigation success.

Success of a wetland mitigation area can be viewed in two ways: fulfilment of permitting conditions and/or creation of a viable, functioning wetland system that compensates for unavoidable impacts in wetlands. DNRP's success criteria accounts for both views, however, the ecological success carries the greater weight.

The dynamic nature of the wetland field provides opportunities for DNRP to continuously re-evaluate conditions or design features placed on wetlands impact approvals. DNRP can then determine, through the

wetlands mitigation monitoring program, which conditions are contributing to the ultimate success of the area. The program also facilitates review of mitigation design specifications, which can then be incorporated into future impact and mitigation reviews.

1. Introduction

In the rapidly evolving field of wetlands regulation, mitigation, and design, it is important to have a regulatory program that is flexible enough to identify deviations and adapt. The philosophy of the program in Broward County, executed by the Department of Natural Resource Protection (DNRP), is that monitoring wetland mitigation areas is critical to the ultimate success of these areas.

The local ordinance that DNRP implements is the Aquatic and Wetland Resource Protection Ordinance (Broward County Board of County Commissioners, 1994). The ordinance sets forth activities that are regulated, impacts that are allowed with or without compensation, and impacts that are not allowed. Clear mitigation design and success criteria are also detailed in the ordinance.

When DNRP first entered the realm of wetlands mitigation (around 1984), littoral shelf creation and planting were the standard in the science, with a rare attempt at preservation or enhancement in forested systems. Today, a full range of mitigation design options exists. Areas are sometimes preserved or restored if sufficient enhancement and long-term protection can be provided. Other times areas are scraped and lowered, at the same time the site construction is proceeding, and re-mucked and planted, essentially resulting in a new wetland. Consolidation of small mitigation projects in a larger off-site location, or mitigation banking, is also becoming more and more popular and seems to offer benefits. Many of the old mitigation techniques have fallen by the wayside, due, in part, to the knowledge gained through mitigation monitoring programs.

DNRP requires monitoring plans for mitigation projects, as do many other agencies. However, DNRP believes that its niche in the scheme of regulatory agencies is to provide a strong local presence to verify monitoring reports, look for pitfalls to ultimate success and provide timely intervention.

This paper will examine the need for strong yet responsive monitoring oversight, discuss the products of a mitigation monitoring program and address future concerns.

2. Study Site

Broward County is a densely populated area of southeast Florida containing the major cities, among others, of Fort Lauderdale, Hollywood, and Pompano Beach (Figure 1). DNRP's jurisdiction over dredge and fill activities encompasses all of the County and its variety of aquatic and wetland resources. The ocean with its fragile coral reefs and the inland waterways with mangrove and seagrass communities converge with freshwater sawgrass marshes, red maple forests and cypress swamps. Development pressures have been confined to the eastern one-third of the county while the western two-thirds is protected as Water Conservation Areas (Everglades).

3. Materials and Methods

DNRP's Wetlands Resources staff, responsible for implementing all aspects of the local regulatory program, has grown from two technical positions in the 80's to the present count of eight. Currently, two staff members are dedicated to the mitigation monitoring program, with support from other positions as needed.

At this time, DNRP tracks the status of nearly 300 mitigation projects. Table 1 provides a summary of the number, status, and size of projects.

Computer programs, most recently Lotus, have proven to be essential in the tracking of mitigation projects. Spreadsheet programs, like Lotus, can be set-up to provide trigger dates that can be easily monitored, such as report due dates. Geographical Information Systems (GIS) coverages have also been developed for Broward County wetlands and mitigation areas.

Generally, a quarterly monitoring plan for a duration of five years is required of an applicant at the time of licensing of the wetland impact, granted in an Environmental Resource License (Broward County Board of County Commissioners, 1994). The five year duration allows two years for the area to reach success criteria and requires three subsequent years demonstrating persistence of success factors. Occasionally, semi-annual or annual monitoring is allowed or the duration is lengthened or shortened, depending on the type of mitigation proposed.

While DNRP requires mitigation to be constructed concurrently with licensed impacts (Broward County Board of County Commissioners, 1994), the actual mitigation area may not be completed for some time after license issuance. Therefore, projects must be routinely inspected to determine if the mitigation area is progressing as prescribed by license conditions.

Once the mitigation is constructed, as-built drawings and vegetation data, with photographic documentation, are submitted in the form of a time-zero report which starts the clock for the monitoring period. The time-zero report and all subsequent quarterly reports are reviewed for compliance and field verified by DNRP. The state of compliance is noted in two ways, adherence to license conditions and production of a functioning wetland system. If one form of compliance is present and the other is not, DNRP re-evaluates the license conditions and examines ways to bring the project into full compliance. If necessary, administrative enforcement procedures are employed to ensure compliance with the license and conditions in a timely manner (Broward County Board of County Commissioners, 1994).

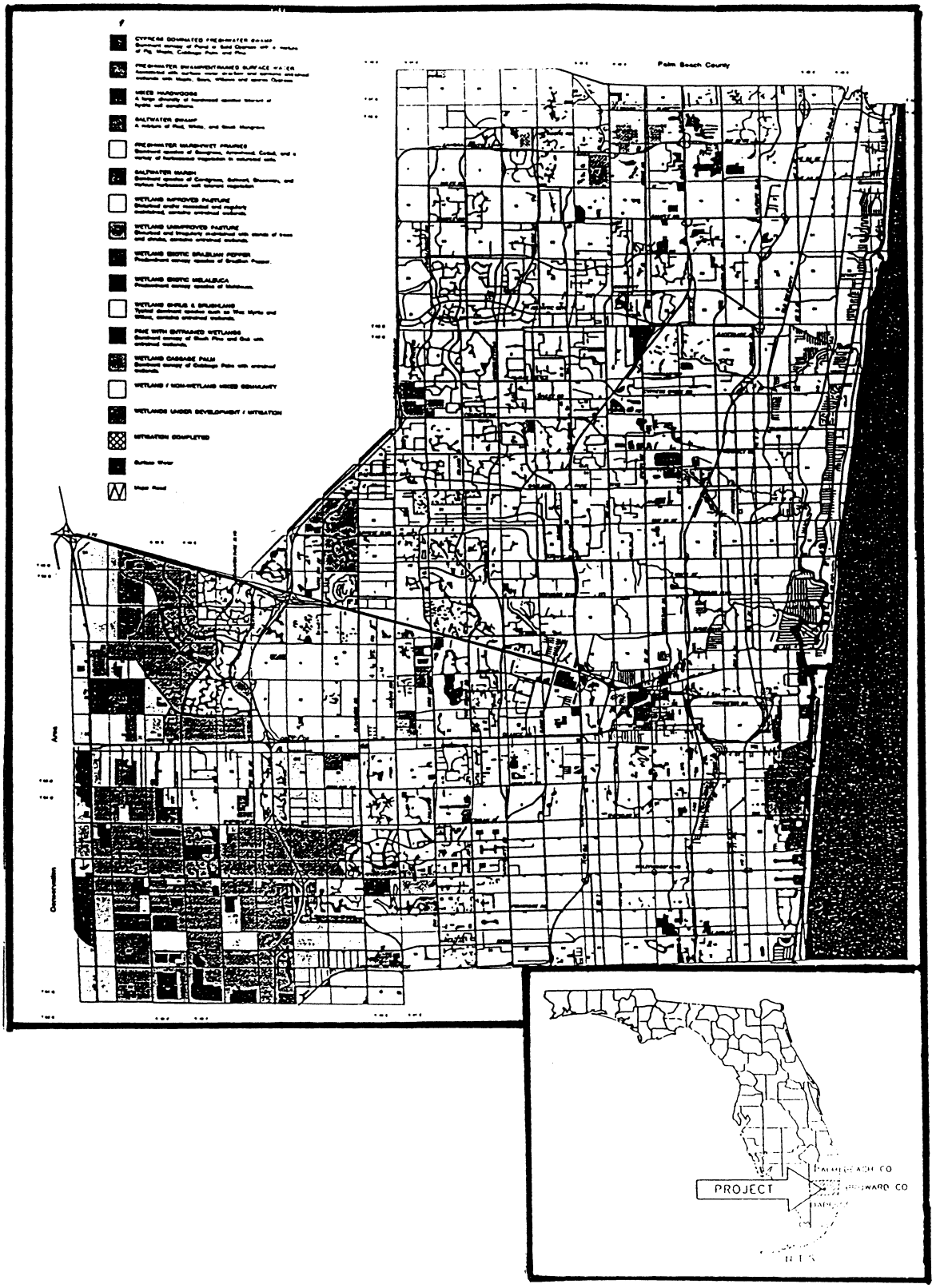

Figure 1. Location map of Broward County, Florida

PROJECT STATUS	NUMBER OF PROJECTS	% OF TOTAL	HECTARES OF MITIGATION	% PROJECTS WITH ACTIVE LICENSES
Close-out Pending	2	1	0.41	0.00
Impacts Started	30	11	177.28	29.00
In Monitoring	91	32	855.55	41.76
Installation Pending	114	40	979.72	77.19
Installation Started	6	2	41.35	100.00
In for Relicensing	4	1	29.59	0.00
Successful	35	12	48.26	0.00
Unsuccessful	3	1	0.26	0.00
TOTAL	**285**	**100**	**2,132.42**	**51.95**

Table 1. Wetland mitigation projects in Broward County, Florida

4. Results

Through the mitigation monitoring program, DNRP has recognized several factors that contribute to the success of mitigation areas. Overlooking these factors may result in failure, or at least delayed success, of the project. The following is a list of the majority of these factors.

A. Size

In general, the belief is, and has been, that bigger is better. However, small is not necessarily indicative of a loss of wetland functions. Many of the early projects impacted small areas of wetlands and therefore required small areas of mitigation, which may in fact be providing as much or more benefit than the original wetland. In addition, some areas may be small in size but may provide huge benefits to an existing system due to the location or nature of the mitigation. For example, mangrove fringes provide tremendous habitat value to the marine/estuarine environment.

Figure 2 provides a graphical summary of the mitigation projects DNRP is currently tracking. As seen in Figure 2, the greatest overall number of projects are small in size (< 5 ha). This information is deceiving in that the majority of actual acres of mitigation are, in fact, in larger sized mitigation projects (> 50 Ha). Figure 3 provides a graphical representation of project distribution as a function of size. This apparent discrepancy could be due in part to the fact that DNRP also tracks mitigation provided in a bank or joint project individually, so the raw data does not show it as a part of a larger wetland.

B. Required plantings

Results noted through the mitigation monitoring program indicate that more is better in the case of wetland planting. No projects that have been allowed to naturally recruit have become successful without supplemental

planting. In addition, leaving an area bare, awaiting natural propagation of desirable plants, often leads to invasion of the area by undesirable exotic and nuisance species. This in turn leads to a greater maintenance problem throughout the life of the project.

C. Consolidated

As can be expected, mitigation monitoring results have shown that mitigation consolidated within a project has a greater degree of success than mitigation scattered among many stormwater lakes. Scattered areas tend to be smaller and more susceptible to exotic and nuisance invasion.

D. Buffered

Early mitigation designs often took advantage of available linear open space into which mitigation areas were crammed, without much thought of buffering from adjacent land uses. Littoral shelves on stormwater management lakes were often provided as mitigation even though they were in an individual homeowner's backyard. A variety of problems result from this type of juxtaposition. Adjacent land owners/users mowed, trimmed, planted sod, built playground areas, and dumped vegetative debris into mitigation areas. Even if the area was physically tolerated, it was often not tolerated aesthetically, prompting numerous complaints to public offices, media, and others. Mitigation areas are now planned with sufficient buffers, such as open water barriers, to minimize the effects of adjacent uses.

E. Sufficient substrate

Adequate substrate, primarily muck in Broward County, is essential to the growth and success of mitigation areas. Through the mitigation monitoring program, DNRP has noted that a minimum of one foot of

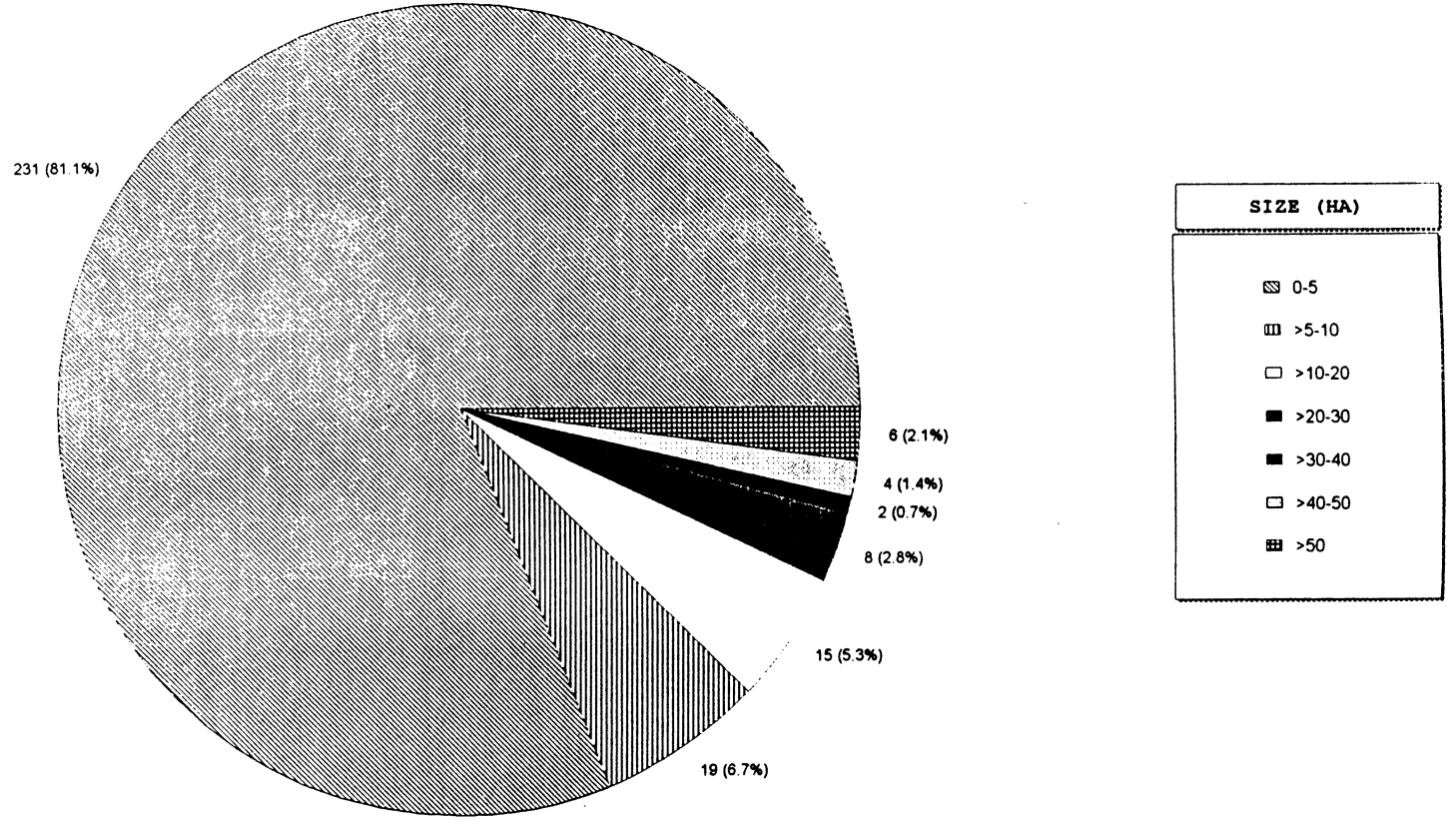

Figure 2. Variation of mitigation projects by size

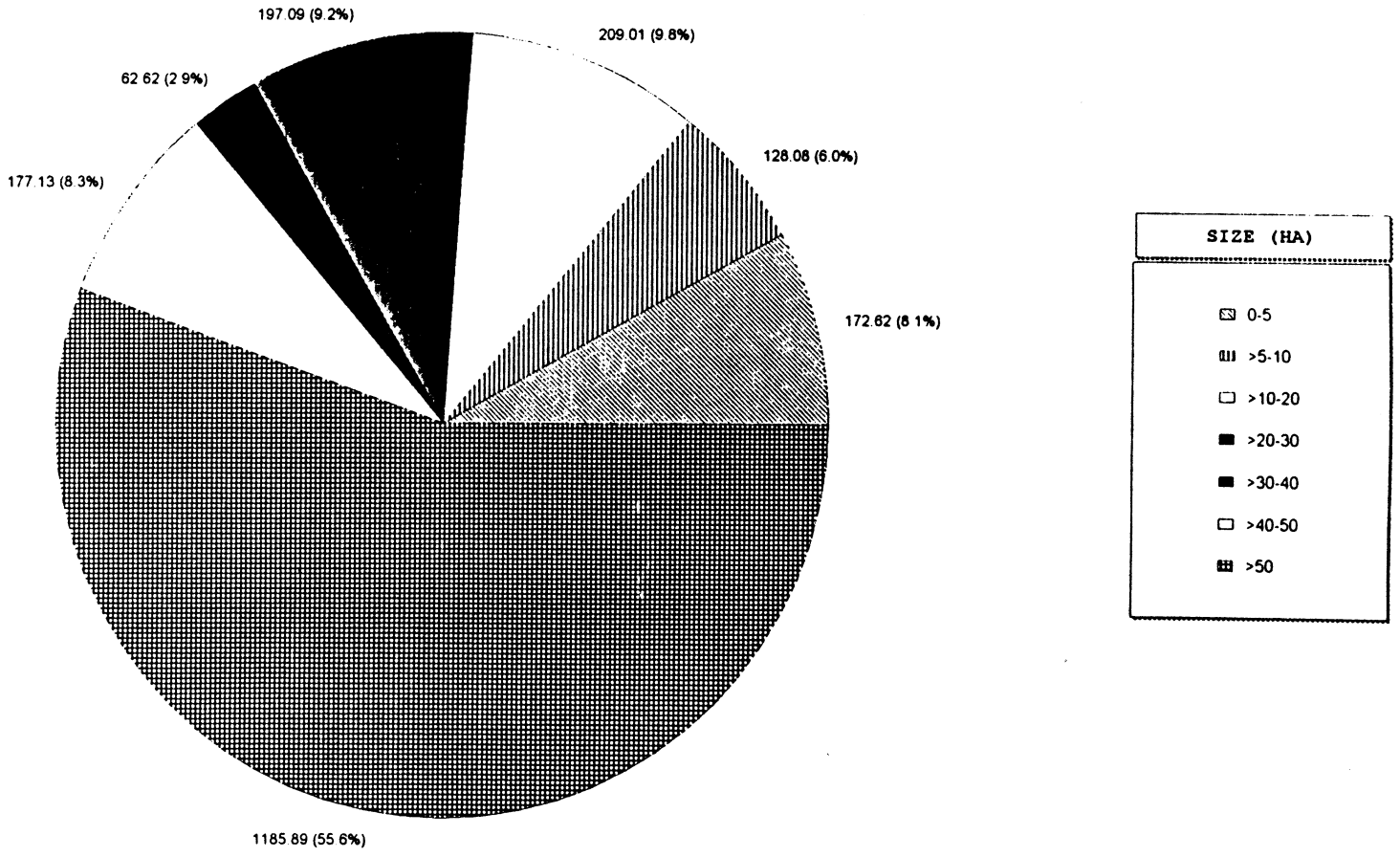

Figure 3. Coverage of wetland mitigation projects by size class (ha)

muck provides the best start for created wetland areas. Although areas with less muck may ultimately become successful, the rate of maturation is slower and the health and vigor of the plants may suffer.

F. Preclusion of non-native herbivores

A somewhat unusual problem for a wetland mitigation area has proven to be predation by grass carp. Whether or not they are introduced with proper permits and approvals, grass carp can decimate a mitigation area. They are particularly detrimental to spikerush (Eleocharis spp.), wiping out entire stands and cropping new growth at a rate that inhibits re-establishment. Grass carp must be prevented from entering mitigation zones.

G. Sufficient hydrologic inputs

Many mitigation areas can be designed for the best hydrologic conditions when being incorporated into a stormwater management system. However, preservation and/or enhancement projects often lack sufficient hydrologic inputs to ensure that the health, viability, and wetland conditions remain intact or improve. Adjacent drainage effects and surface water flow alterations must be counteracted by providing hydrologic improvements.

5. Discussion

Being aware of all of the factors that contribute to mitigation success is important in DNRP's review of projects during the application process. Mitigation proposals and designs can be modified to reflect the best possible factors for success, including those addressed in this paper and many others, such as both short term and long term stewardship and financial responsibility. Many of these concerns can be dealt with prior to

license issuance.

In addition, the constraints and requirements placed on a licensee serve as an essential tool on a mitigation area's path to success. DNRP places requirements on licensees by granting an Environmental Resource License with Special Conditions. Following is a list of the pertinent conditions that govern mitigation success, excerpted from the Broward County Code of Ordinances, Chapter 27.

Section 27-339(3)(a) states, in part:

> **Construction and installation of the area shall be in accordance with plans dated (date) by DNRP (attached) and associated information. The area shall be installed concurrently with licensed construction.**

This condition serves to minimize the loss of wetland values over time and lead a mitigation area more rapidly to success.

Section 27-339(3)(b) states:

> **Upon completion of the area, the following documentation shall be submitted to DNRP:**
> **1. Certification of elevations in relation to design;**
> **2. Verification of actual acreage; and**
> **3. The time-zero monitoring report.**
> **This documentation is required within thirty (30) days of completion of the area and prior to any certificate of occupancy's being issued for any structure on the site/phase.**

This condition serves many purposes; to ensure correct construction, size, and timing. Comparison to certificates of occupancy provides DNRP with a "time certain" should enforcement action be necessary. One possible improvement in this condition would be to require elevations and acreage verifications prior to time-zero, thereby allowing DNRP to confirm that target substrate and hydrologic conditions have been met, before earth-moving equipment has been removed from the site and plants have been

installed.

Section 27-339(3)(c) states:

> **A viable wetland system shall be established that replicates a natural reference wetland in basic structure and function. In order to assure that the area becomes self-sustaining, the following criteria shall be met:**
>
> 1. **A minimum or eighty (80) percent coverage by desirable wetland species after a two-year period and demonstration of persistence for three (3) additional years.**
> 2. **Less than five (5) percent coverage by invasive exotic and undesirable species. Exotic and undesirable species include, but are not limited to, melaleuca, Australian pine, Brazilian pepper, bischofia, torpedo grass, primrose-willow, and cattail.**
> 3. **A minimum of eighty (80) percent survival of each planted species. This rate shall be maintained each quarter except where species composition, density of planted and recruitment species and overall wetland condition, growth rates and viability of the area are of greater quality, as determined by DNRP.**
> 4. **Hydrologic conditions and soil characteristics are in general conformity to those specified in plans.**
>
> **The area shall be monitored and reports submitted quarterly for five (5) years describing in detail the condition of the area relative to the reference wetland and the criteria listed above.**

This condition sets up the basic criteria that the mitigation area must achieve. These criteria were formulated by reviewing successes and failures in the mitigation monitoring program. It is important to note in item 3 that DNRP has reserved the flexibility to look at the overall viability of an area, not just one factor.

Section 27-339(3)(d) states:

> **should DNRP determine that the area is not achieving the listed criteria during some portion of the monitoring period, the licensee shall prepare contingency plans that demonstrate clearly how the problems(s) will be corrected and submit such plans immediately to DNRP for approval.**

This condition provides DNRP with the ability to force a licensee to remedy a problem, if necessary.

Other conditions are used to ensure that the licensee follows through on early proposals for long term health and preservation, such as granting conservation easements and posting performance bonds.

In conclusion, DNRP acknowledges a need to respond to the everyday realities of wetland mitigation engineering, construction, and maintenance. The result of monitoring on-going projects is an irreplaceable knowledge that can be translated into future requirements at both the design and construction stages.

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THE ROLE OF FIRE IN THE MANAGEMENT OF THE PINE BARRENS: SUFFOLK COUNTY, N.Y.

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ABSTRACT

Frequently the techniques used in the management of woodlands are based on an inadequate knowledge of the history and pre-history of a given area. Often the assumption is made that the present day forest composition reflects that of the woodland prior to European contact. As a result, the management strategies employed may, in fact, perpetuate decades, if not centuries of mismanagement.

This is particularly true on Long Island where woodlands, dominated by the fire dependent pitch pine *(Pinus rigida)*, are assumed to reflect the species composition of the pre-European forests. As a result, fire is considered to be necessary for the maintenance of these woodlands.

Pollen analysis, early historical records and comparisons of burned vs. unburned plots indicate that fire may be a post-European influence in the Long Island forests. Moreover, the analysis of maps and historical aerial photographs indicate that the clear-cutting and/or clearing of earlier hardwood forests may have enabled the shade intolerant pine to enter and dominate the Long Island forests within the past 400 years. Thus, the pine barrens of Long Island may actually be a post-European artifact brought about and maintained by the mismanagement of the hardwood forests.

INTRODUCTION

The role of fire in the management and maintenance of the woodlands of Long

Island is questionable. Although the pine barrens, which comprise a 52,000 acre New York State Preservation Area, are purported to be a fire dependent ecosystem, it is probable that the pre-contact woodlands were dominated by hardwoods and that the pines are the result of mismanagement during the Colonial and post-Colonial periods. Conversely, studies by Conard (1935) and, more recently, by Boerner, et. al. (1987) indicate that fire may actually hasten the succession to oak *(Quercus sp.)*.

That the pre-contact woodlands may have been dominated by hardwoods is based on the time that Long Island has been icefree, thereby allowing biological succession to proceed, as well as early historical records and various pollen studies. In addition, surveys of cleared areas show that pines rapidly colonize these mineralized sediments. Moreover, studies of recently burned areas confirm those of Conard (1935) and Boerner, et al (1987).

Long Island was formed by sediments deposited during the last advance of the Wisconsinian Glacier which retreated some 20,000 YBP. Immediately following the retreat of the glacier the climate was considerably colder and moister than at present. Such a climatic regime would have resulted in vegetational communities akin to the arctic tundra. As the glacier continued its retreat the climate moderated and forests similar to the spruce-fir communities of present-day Newfoundland would have predominated. Somewhat later, again in response to further climatic moderation, a northern transitional hardwood forest such as that of New England would have dominated the landscape.

Approximately 8,000 YBP, during the Hypsithermal, the climate became considerably warmer and drier enabling more southerly species to establish. This could have led to the decline of the northern hardwood transitional forest. It is probable that the pitch pine *(Pinus rigida)* became established during this time. The climate then moderated, and for the past 4,000 years the temperature and precipitation patterns have been fairly constant and similar to those of today.

During this time a complex soil structure was developing. As noted, pines are presumed to have colonized the area during the Hypsithermal. The slower growing oaks (Quercus spp.), hickories (*Carya* spp.), red maple (*Acer rubrum)*

and the chestnut *(Castanea dentata)* were present in lower numbers.

Over time, and in the absence of frequent disturbance that would open the canopy, the hardwoods would tend to eliminate the shade intolerant pines. Thus, the species composition would have shifted to one in which the hardwoods predominated. The pines would persist and be confined to areas subject to a higher frequency of disturbance which would adversely affect the hardwoods and favor the shade intolerant species.

Such vegetational sequences since the end of the Pleistocene appear to be confirmed by palynological studies. Sirkin (1989) has found pine pollen at higher percentages than oak in sediments dating from approximately 10,000 to 5,000 YBP, while those f rom 5, 000 YBP to the present show the oak pollen to be at a significantly higher percentage than that of the pine.

Two thousand years would be a sufficient time to develop an old growth stage of hardwoods provided that severe, extensive and rather continual disturbance did not bring a return to an earlier vegetational stage.

Thus, extensive pine forests could only have persisted by frequent disturbance and, on Long Island, the only natural disturbance would be fire. The only natural cause of fire, however, is by lightning strikes. Lightning, on Long Island, is generally preceded by heavy rainfall. It is generally agreed that fires set by lightning in a wet woodland are rare (Patterson, 1988). If the pre-European forests were, in fact subject to frequent disturbance, either by clearing or fire, the cause would be anthropogenic rather than natural.

There is considerable controversy as to the extent of burning by pre-contact Native Americans (Russell,1983; Patterson, 1988). At the time of European contact the Native Americans inhabiting Long Island had developed a lifestyle based on shell and fin fishing as well as on agriculture. Patterson (1988) notes that burning was proportional to the population density, while Ceci (1977) estimates that the Native American population at that time was less than 4,000. This is a population density of less than three individuals per square mile and argues for a minimum of human disturbance, both before and at the time of

European contact.

Based on the pollen record for Long Island, the presumed succession during the Holocene and the low population density of the Native Americans at the time of European contact, as well as the low, pre-contact fire frequency, it is logical to assume that the forests of Long Island in the early 1600s were dominated by oak, hickory and chestnut. Pines were confined to those areas experiencing frequent disturbance. Thus, the role of the early settlers needs to be considered since it is probable that their activities had a profound effect on the species composition of the Long Island forests.

It is known that the early settlers considered the forest as a vast and exploitable resource. The land was cleared for farming and wood was cut for home construction and for cordwood, the only source of home heating fuel. Somewhat later, wood was cut to supply Long Island's shipbuilding industry.

Pitch pine is generally not preferred for home or ship building. It is also considered to be a dangerous heating fuel since its resins coat chimneys and cause house fires. However, the Long Island forests were the major source of wood for New York City's heating needs. By 1812, Brookhaven Town, located in central Long Island and the area presently containing the majority of the pine forests, was sending 100,000 cords of wood per year to the city (Prime, 1845). Wood cutting on this scale, combined with that cut for local needs would have virtually eliminated the hardwood forests and allowed for the reestablishment of the shade-intolerant pines in the absence of fire.

The presence of an earlier hardwood forest is confirmed by the records of Camp Upton, now the site of Brookhaven National Laboratory. Although the surrounding forest is now dominated by pine, it was necessary when clearing the area in 1917 to remove closely spaced hardwood stumps that were commonly six feet in diameter (Coyne, 1919).

Fire did become a major factor in the Long Island woodlands in 1844 when the New York to Boston link of the Long Island Railroad, which traversed the present central pine barrens, was completed. Sparks emitted from the coal

burning locomotives caused widespread, well documented forest fires. One fire burned an area 10 miles long and four miles wide. Historians noted that portions of the forest were burned almost daily by the rail road (Treadwell, 1912). Fires of this magnitude were obviously sufficient to perpetuate a fire dependent woodland.

STUDY SITES

The major areas considered in this study are the 5,247 acre Rocky Point Natural Resources Area and the 1,775 acre Dwarf Pine Plain. The Rocky Point Natural Resources Area was previously owned by the Radio Corporation of America and was the site of a trans-Atlantic transmitting station from 1921 to the 1960s. Much of the area was cleared for antennas and fire was suppressed. The Dwarf Pine Plains are purported to require the highest fire frequency in order to maintain the stunted, contorted growth form. Both of these sites are located in the 52,000 acre New York State Pine Barrens Preservation Area. In addition, several smaller sites outside of the preservation area were considered in order to assess the effects of fire and/or clearing (Figure 1).

MATERIALS AND METHODS

A number of methods were used in this study. Quantitative studies involving the establishment of 400 square meter circular plots in which the heights of all of the pines were measured and the degree of tree bole distortion determined and categorized as straight, moderately or extremely contorted. In addition, representative trees in each plot were cored in order to determine their age. Semi-quantitative methods involved line transacts and the point quarter technique, as well as Rapid Environmental Assessments as devised by Welch (1994).

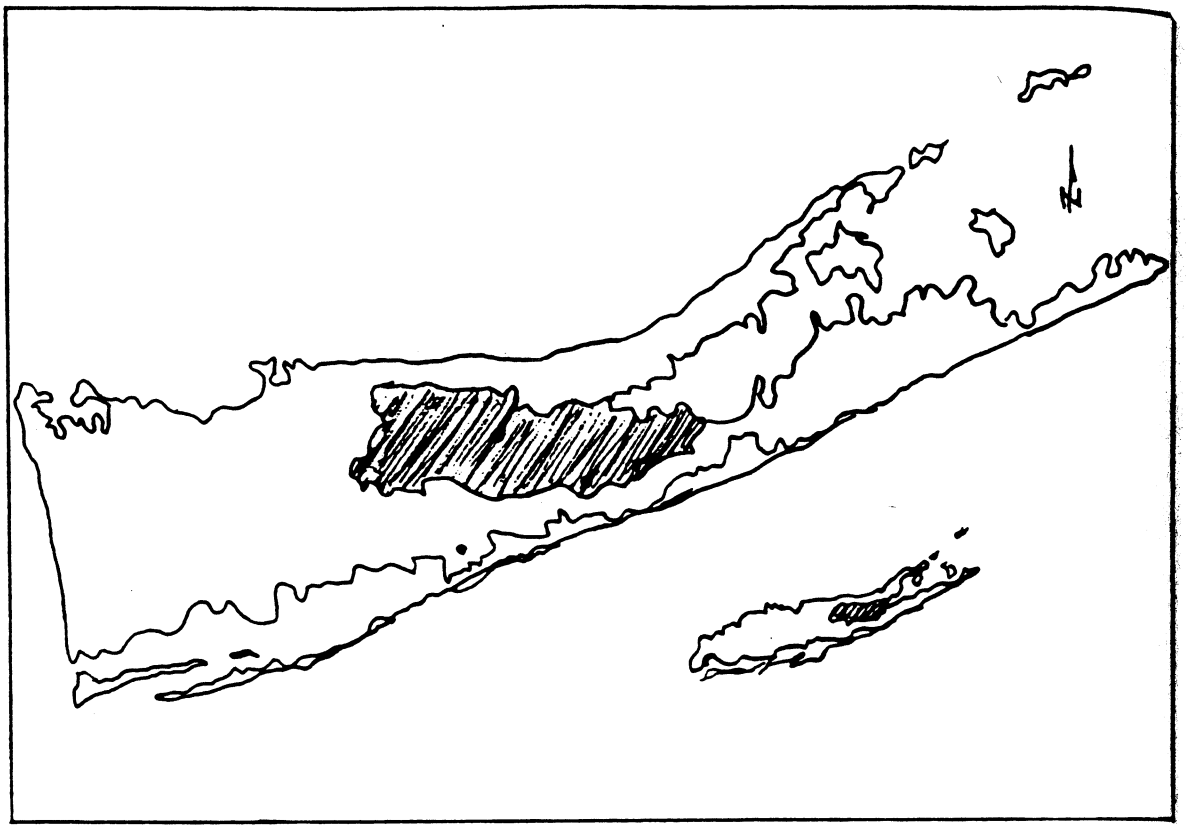

Figure 1: Shaded area indicates the central pine barrens of Suffolk County. Dots indicate study sites.

According to this method soils and topographic data are determined prior to the site visit. On site, the area is traversed in several directions and the herb, shrub and canopy layers are described, as well as the general structure of the community, approximate vegetational density, percent cover and dominance of the various floral components.

In addition, various sets of aerial photographs, dating from 1932 to 1990 were used in order to determine the time frame in which certain areas had burned, were cleared, etc.

RESULTS

General observations, semiquantitative and quantitative studies consistently indicate that pitch pines tend to dominate recently cleared areas such as fire breaks, old fields, etc. For example, in virtually all of the fire breaks surveyed, pitch pine was the first to invade and appears to maintain its dominance for decades. Indeed, in many fire breaks it is the only tree present regardless of the time elapsed since the last clearing.

In addition, the analysis of the forest types on the 5,247 acre Rocky Point Natural Resources Management Area indicates that pitch pine dominates those areas that were cleared frequently between the early 1920s and 1960s. The adjacent areas, on the other hand, have been subject to little, if any clearing and are dominated by scarlet oak *(Q. coccinea)* with black *(Q. velutina)*, and white oak (Q. alba) present in lower numbers. This indicates that the pitch pine invades and dominates on cleared, highly mineralized substrate in the absence of fire.

Fire, including prescribed burning, may, in fact have a deleterious effect on pine communities. Conard (1935) notes that fire eliminates the pitch pine while favoring the growth of scrub oak in the Long Island pine barrens. Boerner, et al (1987) in a study of the New Jersey Pine Barrens notes that "prescribed burning may accelerate, not retard, succession toward oak dominance".

This appears to be confirmed by semiquantitative surveys carried out on several areas on Long Island. This data clearly shows that, in pine barrens that have burned approximately 15 to 20 years ago, there is now a predominance of oak. Moreover, in the majority of these areas there is a total absence of pine recruitment.

In addition, sample plots in the dwarf pine forest, purported to require fire on the order of every seven years (Olsvig, et al, 1979) show just the opposite. Plots established in an area documented to have burned in 1945 show a lower density of pines than a control plot with no documented burn history. Moreover, the number of individuals exhibiting the growth form indicative of the dwarf pine

phenotype is significantly higher than in the control plot.

Very early data obtained in the wake of the wildfires of August 1995 indicate that in the Dwarf Pine Plains the scrub oak *(Quercus ilicifolia)* is rapidly resprouting in the burned area. Within four weeks of the fire the oaks had numerous root sprouts which had attained a height of 0.5-1 meter. Within four months only approximately 50% of the pines had begun to resprout; however, by late April, 1996 many of the pine root sprouts had "browned" and may have died over the winter. The oaks, on the other hand, were robust and continued to dominate the area.

By early summer in the Rocky Point Natural Resources Area, however, epicormic sprouting was common on the pines and a number of new pine seedlings were also noted. Again, however, oak was dominant.

SUMMARY

The forests of Long Island were most probably composed of an oak-hickory-chestnut assemblage at the time of European contact. Over-cutting of this hardwood forest by the early settlers eliminated the hardwoods and enabled the reestablishment of the shade intolerant pitch pine. Although the early railroad is known to have caused extensive wildfires, the impact of such fires is unclear.

Semiquantitative surveys of several areas known to have experienced recent wildfires seem to confirm Conard's (1935) data. Moreover, qualitative surveys indicate that pines rapidly invade and prosper on cleared areas in which the subsoils are exposed. Such areas include fire breaks, antenna fields, etc. The dwarf pines are purported to require the most frequent fire regimes in order to maintain the "dwarf" growth form. Quantative studies, however, indicate that the dwarf phenotype is much more common in areas with no fire history. It is believed that competition with scrub oak, which dominates these areas, rather than fire, forces the pines into the contorted growth form. Moreover, these studies also show that there is a higher population density of pine in the unburned areas.

The value of prescribed burning in the management of the pine barrens remains in doubt. As noted Boerner (1987) holds that prescribed burns may accelerate succession toward oak. Baker (1995) also believes that small prescribed burns will further alter the forest, while larger fires may hasten the restoration of the landscape. It appears obvious that prescribed burning as a management strategy in the Long Island pine barrens is questionable.

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RESTORATION OF IMPROVED PASTURES IN CENTRAL FLORIDA PINE FLATWOODS COMMUNITIES.

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ABSTRACT

In 1995 The Nature Conservancy (TNC) initiated a pilot project in upland pasture restoration aimed at determining cost effective methods for restoring 1500 acres of pasture to pine flatwoods at the Disney Wilderness Preserve (DWP). The preserve is located in central Florida in Osceola and Polk Counties and totals 11,500 acres. The pilot project will examine methods for removal of exotic pasture grasses, primarily bahia *(Paspalum notatum)*, and methods for re-introducing native plants to the site. Methods to be tested for removing bahia grass include multiple disking and herbiciding, single herbiciding and disking, and a combination of treatments. It is anticipated that revegetation of the restoration sites will be accomplished by direct seeding native seed collected on DWP.

INTRODUCTION

As more land is set aside for conservation in Florida, more land is being acquired by private, state, and federal agencies that is currently or was previously agricultural land. Land managers are challenged with the possibility of restoring these agricultural lands, or allowing natural succession to occur. It is becoming apparent that in Central Florida succession of old fields does not follow the predictable stages to a wooded climax community in the classical manner (KBN Engineering, 1988). Instead, succession in Central Florida agricultural lands may follow one of many paths depending on hydrology and fire frequency (Cattelino, 1979).

Other important factors to consider are degree of disturbance and persistence (or competitive ability) of the introduced species.

Land managers and researchers throughout Florida have experienced problems with leaving improved pastures to natural succession. There are documented instances where pastures improved to bahia grass *(Paspalum notatum)* have been abandoned for 20 to 30 years yet are not being colonized by herbaceous native plants. For example, the abandoned pastures at San Felasco State Preserve have had no management of any kind for 21 years, and the only species present with the bahia are blackberry *(Rubus* spp.*)* and loblolly pine *(Pinus taeda)* (V. Doig, pers. comm.). Neither of these species is a dominant in the desired or anticipated community. Other abandoned pastures at the same state park have been completely colonized by woody species, but bahia grass persists as the herbaceous layer. The lack of pyrogenic fuels in the bahia dominated understory prevents naturally occurring fires from burning through, and allows woody species to become dominant in a system traditionally dominated by the herbs (V. Doig, 1994). Other documented sites that continue to persist with bahia grass remaining dominant include Audubon's Kissimmee Prairie Sanctuary, Archbold Biological Research Station, and Apalachicola Bluffs and Ravines Preserve.

Because natural succession does not appear to proceed to a desired or recognized community type even in long abandoned pastures, many land managers and researchers in Florida and the Southeast are experimenting with options for restoring the herbaceous groundcover. The most common habitats converted to pasture were sandhills and flatwoods (KBN, 1988). These vegetation types are dominated by fire adapted species, predominately wiregrass *(Aristida beyrichiana*, formerly *A. stricta)*. Much of the research has focused on how to re-introduce this species into abandoned agricultural lands. Seeding wiregrass into bahia pastures is difficult, since bahia grass is more competitive and wiregrass grows more slowly and has low seed viability. The alternative, planting seedling wiregrass, is very labor intensive.

To date, most of the research in bahia grass pasture restoration has focused on planting plugs of wiregrass and other native species, and eliminating the bahia grass through herbicides and/or disking (Seamon, 1992; Hatcher, 1994; Uridel, 1994). These methods have proved effective on a small scale. Uridel's (1994) herbicide, disking, and planting in $10m^2$ plots was successful but costly to undertake on a larger scale. A current project on South Florida Water Management District land on the Kissimmee River north of Lake Okeechobee focuses on removing the bahia grass, but not planting or seeding any native plants. Preliminary results are showing bahia grass re-invades or other exotic species quickly gain dominance (Hatcher, 1994). Direct seeding of wiregrass has been tried with limited success, but this treatment is very new (Bissett, 1994; Seamon, 1994).

STUDY SITES

Pasture restoration at The Disney Wilderness Preserve began in the fall of 1995 with the establishment of a pilot project. Six sites (Figure 1) were randomly chosen in pastures that are currently classified as improved bahia pastures. Each site contains 5 plots 30x30 meters in size (Figure 2). There is an 8 meter buffer between plots, and the actual area of treatment is 32x32 meters to minimize edge effects. The pastures are currently grazed, although the treatment areas have been fenced from cows with hog fencing and barbed wire. The sites are aligned perpendicular to any elevational gradient to include representative elevations within the pastures.

MATERIALS AND METHODS

The pilot project will test 5 options for removal of bahia grass:

1. Disking
2. Herbiciding
3. Disking and herbiciding
4. Multiple disking

Figure 1. Pasture restoration study sites on the Disney Wilderness Preserve.

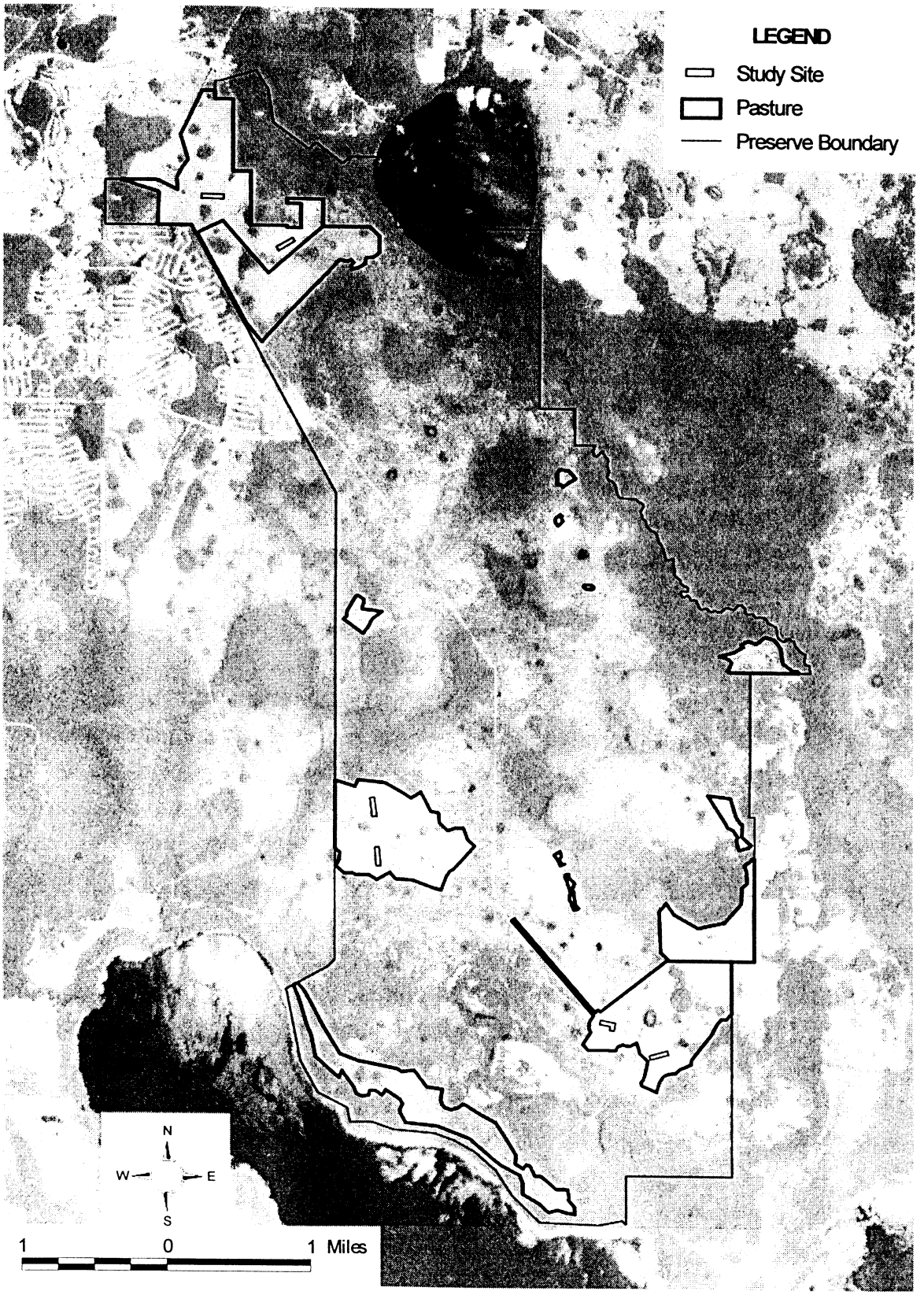
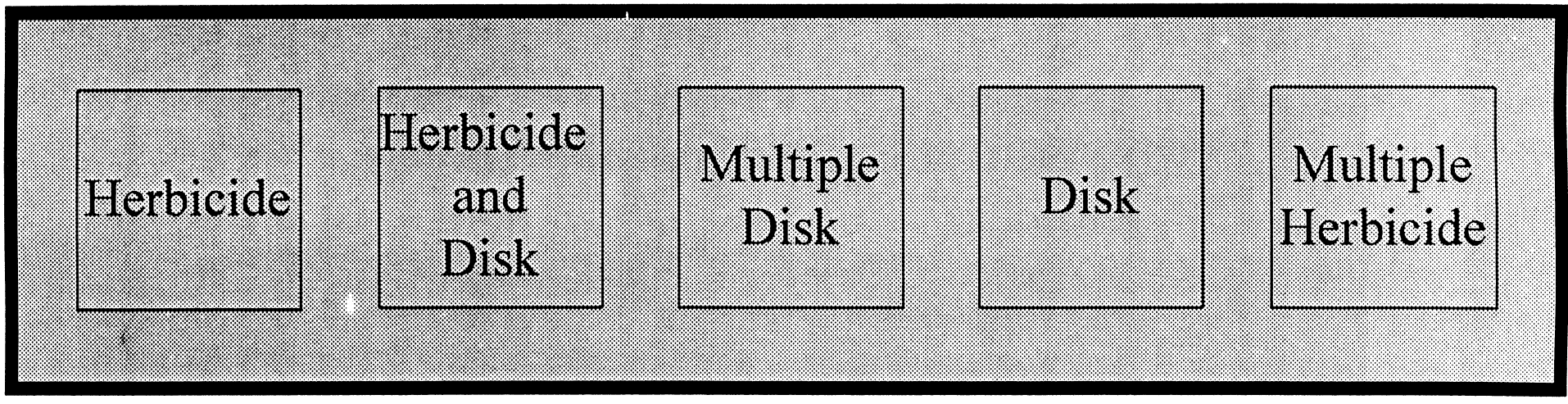


Figure 2



Plots 30 x30 meters

10 meter buffer between plots

5. Multiple herbiciding

Single disking and herbiciding treatments are incorporated to provide comparative information regarding the additional benefits of multiple efforts to remove bahia grass. Multiple treatments are anticipated to provide better weed control, although the additional costs may be substantial. The combination treatment of disk and herbicide was included to see if combining the treatments reduced cost or provided better bahia control for the same cost as one of the other treatments.

Following the treatments for removal of bahia grass, the sites will be both direct seeded with native plant species and also planted with a small number of potted plants of native species. (Native species will be naturally occurring plants in Florida communities). Pre-treatment monitoring began prior to any weed treatment to quantify the current condition of the pastures based on cover and plant composition. Following the monitoring, bahia removal treatments will begin in the spring of 1996, and continue until fall 1996. Seed will be collected in the fall of 1995 for the potted plant portion of the pilot project, to be planted in the treatments in September 1996. Seed will be collected in the spring, summer, and fall of 1996, to be spread on the plots in late November 1996. All seed for direct seeding and potted plants will be collected from DWP.

The pre-treatment monitoring used the point-intercept method. For each 30x30 meter plot a point intercept transect was randomly located within each of six 5x30 meter belts. Then measurements were made at 30 equidistant points along the transect. The initial point was randomly selected within the first meter of the transect. A camera tripod with a level attachment allowed the pin to be dropped vertically at each point. A botanist identified plants touched by the tip of the pin as it descended. Data were recorded on a Hewlett Packard palm-top computer in Lotus format. The transects are not permanent. These data will permit us to evaluate whether we have been able to reduce exotic vegetation to a percent cover of less than 10%. The sites will be monitored annually for a period of three years following completion of treatments to determine

TASK	1995						1996											
	Jul	Aug	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
PLANNING																		
complete restoration and implementation plan																		
additional planning/analyze data/reports																		
PLANT MATERIAL COLLECTION																		
most grasses and forbs (mechanical collection)																		
longleaf pine seeds																		
Indian grass and other species (hand collect)																		
SITE PREPARATION																		
selecting sites																		
building fences																		
*herbiciding																		
**mowing																		
*disking																		
PLANTING																		
planting seedlings																		
sowing seeds (including rolling them in)																		
MONITORING																		
pre-treatment																		
survivorship/cover																		
photo-point																		

*Herbiciding and disking will be performed on an as needed basis during these months.

**Mowing will be performed on an as needed basis between the plots.

Figure 3

TASK	1997												1998											
	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
PLANNING																								
Site plans for restoration sites																								
additional planning/analyze data/reports																								
PLANT MATERIAL COLLECTION (new sites)																								
most grasses and forbs (mechanical collection)																								
longleaf pine seeds																								
Indian grass and other species (hand collect)																								
SITE PREPARATION (Additional Sites)																								
selecting sites (for next year)																								
building fences (where necessary)																								
*herbiciding (only new sites)																								
**mowing (as needed for all sites)																								
*disking (only new sites)																								
PLANTING																								
planting seedlings																								
sowing seeds (including rolling them in)																								
MONITORING																								
pre-treatment (new sites)																								
survivorship/cover																								
photo-point																								

*Herbiciding and disking will be performed on an as needed basis during these months.

**Mowing will be performed on an as needed basis between the plots.

which bahia removal method is most effective at diminishing the bahia cover and facilitating establishment of the native groundcover. Photo-points were established to track changes on the sites. A time line for the next three years is presented in Figure 3.

RESULTS

The results of the pilot study will not be known for a few years. What information is learned about pasture restoration from the pilot study will be applied to large scale pasture restoration on DWP in the coming years.

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EFFECTS OF MECHANICAL HARVESTING OF FLOATING PLANT COMMUNITIES ON WATER QUALITY IN LAKE ISTOKPOGA, FLORIDA

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ABSTRACT

The effects of harvesting floating plant communities (tussocks) by mechanical methods on selected water quality parameters in Lake Istokpoga, Florida were examined. Dissolved oxygen, temperature, pH, conductivity, turbidity, chlorophyll *a*, and nutrients (nitrogen and phosphorus) were compared under pre-harvest, harvest and post-harvest conditions. Water quality data were compared between one tussock harvest site and three reference sites (30 m, 61 m, and 91 m away from the harvested plot). Very minor, although statistically detectable ($p < 0.05$) DO differences occurred at the harvest site. Chlorophyll *a*, total nitrogen and total phosphorus concentrations decreased significantly during the harvest. Loadings of total nitrogen concentrations in Lake Istokpoga could be reduced by harvesting of floating plant communities in the littoral areas. A significant increase in turbidity and dissolved solids occurred during

mechanical harvesting in both the harvested plot and the reference site 30 m away.

INTRODUCTION

Any lake in which there is an accumulation of littoral sediments has the potential for being plagued with a nuisance growth of aquatic macrophytes. Nutrients, along with dissolved and particulate organic matter, are released in the water column as plants decompose. Oxidation of plant detritus may bring about an oxygen deficit and cause release of nutrients from chemically reduced sediments. The release of nutrients can stimulate algae growth and increase macrophyte production. Thick beds of macrophytes will also accelerate accretion of the littoral zone as plant material accumulates at the sediment surface after senescence (Wetzel 1975; Rich and Wetzel 1978). The regulated water level regime on Lake Istokpoga is relatively narrow with a maximum high level of 12.0 m NVGD (occurring in fall and winter) and a low of 11.5 m NVGD. Typically on water bodies with narrow water level regimes ($\leq$ 0.6 m), accumulation of organic detritus in the littoral zone is accelerated. Some aquatic plants root in the organic detritus and form extensive interconnected mats or tussocks. During low water periods tussocks may rest on the bottom, but during high water periods, the buoyancy of meshed aquatic plants and gas produced in the organic detritus allow tussocks to float. Water below tussocks is typically anaerobic and heavily laden with organic material; thereby, unsuitable for fish habitat. The accumulation of organic detritus associated with tussock communities on Lake Istokpoga eliminated large areas of high quality aquatic habitat. As tussock communities expand, sections occasionally break free. Floating tussocks destroy beneficial plant communities by blocking sunlight needed for photosynthesis.

The most common method of controlling nuisance aquatic plants is application of approved herbicides. Alternative methods of macrophyte control, e.g. mechanical harvesting, are being sought or refined. The effectiveness and limitations of aquatic vegetation removal by harvesters

is well documented (Dunst 1974; Neel et al. 1973; Nichols and Cottam 1972; Garrison 1979; Wile and Hitchin 1977; Bedrosian 1982; Storch and Winter 1983; Conyers and Cooke 1983). The immediate effect of harvesting is to improve navigation, increase angler access, and open shoreline areas for activities such as swimming. Indirectly, harvesting can reduce the potential for odor and aesthetic problems associated with decaying vegetation. Additionally, when all other controllable sources of nutrient inputs are reduced, removing vegetation could have a positive effect on the total nutrient pool available for vegetation production. The results of several studies on the latter theory have been variable (Neel et al. 1973; Bedrosian 1982). While the total reduction in the nutrient availability may be insignificant in many cases, plant biomass reduction during subsequent growth periods is significant and may be related to sediment/plant nutrient interaction (Aiken and Picard 1980; Bole and Allen 1978; Kimbel and Carpenter 1981; Carpenter and Adams 1977; Barko and Smart 1983; Barko 1983).

Diverse habitat provided by rooted aquatic plants is important for good quality fish spawning and nursery areas. Although conditions for long-term maintenance of desirable littoral plant communities on Lake Istokpoga are restricted by regulated water levels, desirable littoral areas (quality fishery habitat) could be restored and managed through a project directed toward physical removal of tussock communities. The objectives of the study were: (1) to investigate the effectiveness of mechanical techniques commonly available to control nuisance aquatic vegetation, and (2) to document the effects of harvesting on selected water quality parameters.

STUDY AREA

Lake Istokpoga, the fifth largest lake in Florida, is an 11,080 ha freshwater lake located in Highlands County. The circumference of the lake is about 50.7 km, and the lake has an adjacent littoral zone of about 881 ha. The lake is relatively shallow, with an average depth of 1.8 m. The north,

northeast and southwest shorelines of Lake Istokpoga are dominated by residential development. The littoral community along the northeast shore has been substantially altered by canals and access channels for private residential development.

Milleson (1978) documented an increase in littoral vegetation from 881 to 1,294 ha in Lake Istokpoga between 1944 and 1975 caused by persistently lower lake levels and subsequent reduction of the natural frequency of seasonal drying and inundation. Annual maximum water level fluctuation was reduced from 2.1 m prior to 1962, to 1.0 m because of S-68 operation (Milleson 1978). In 1991, fluctuation was further reduced to 0.6 m by a water level regulation schedule change (38.0 msl to 39.5 msl) (SFWMD unpublished data). The amount of quality fish habitat had been reduced by the formation of extensive floating tussocks and dense cattail (*Typha* sp.) communities. Without dewatering events, germination of diverse aquatic plant seeds was reduced, consolidation and compaction of organic sediments did not occur, and formation and expansion of floating mats of water hyacinths (*Eichhornia crassipes*) and other species common to tussock communities was promoted. These mats reduced overall productivity and diversity of the marsh and littoral zone. In 1991, floating tussock communities occupied 415 ha, excluding island areas. Cattail coverage had increased by 87% since 1977.

Hydrilla (*Hydrilla verticillata*), water hyacinth, and water lettuce (*Pistia stratiotes*) are exotic plants introduced into Florida's waters that have subsequently spread at alarming rates. Hydrilla expanded in Lake Istokpoga from 1,441 ha in 1987 to more than 5,261 ha in 1988 and impedes navigation. Common aquatic plants present in Lake Istokpoga include cattail (*Typha latifolia*), giant bulrush (*Scirpus californicus*), duck potato (*Sagittaria lancifolia*), pickerelweed (*Pontederia cordata*), spatterdock (*Nuphar luteum*), Illinois pondweed (*Potamogeton illinoensis*), eelgrass (*Vallisneria americana*) and bladderwort (*Utricularia* sp). Problematic species include water hyacinth, water lettuce, bulrush (*Scirpus cubensis*), cattail and hydrilla. Reduced water level fluctuation stimulated growth of dense cattail monocultures along portions of the west, north, and

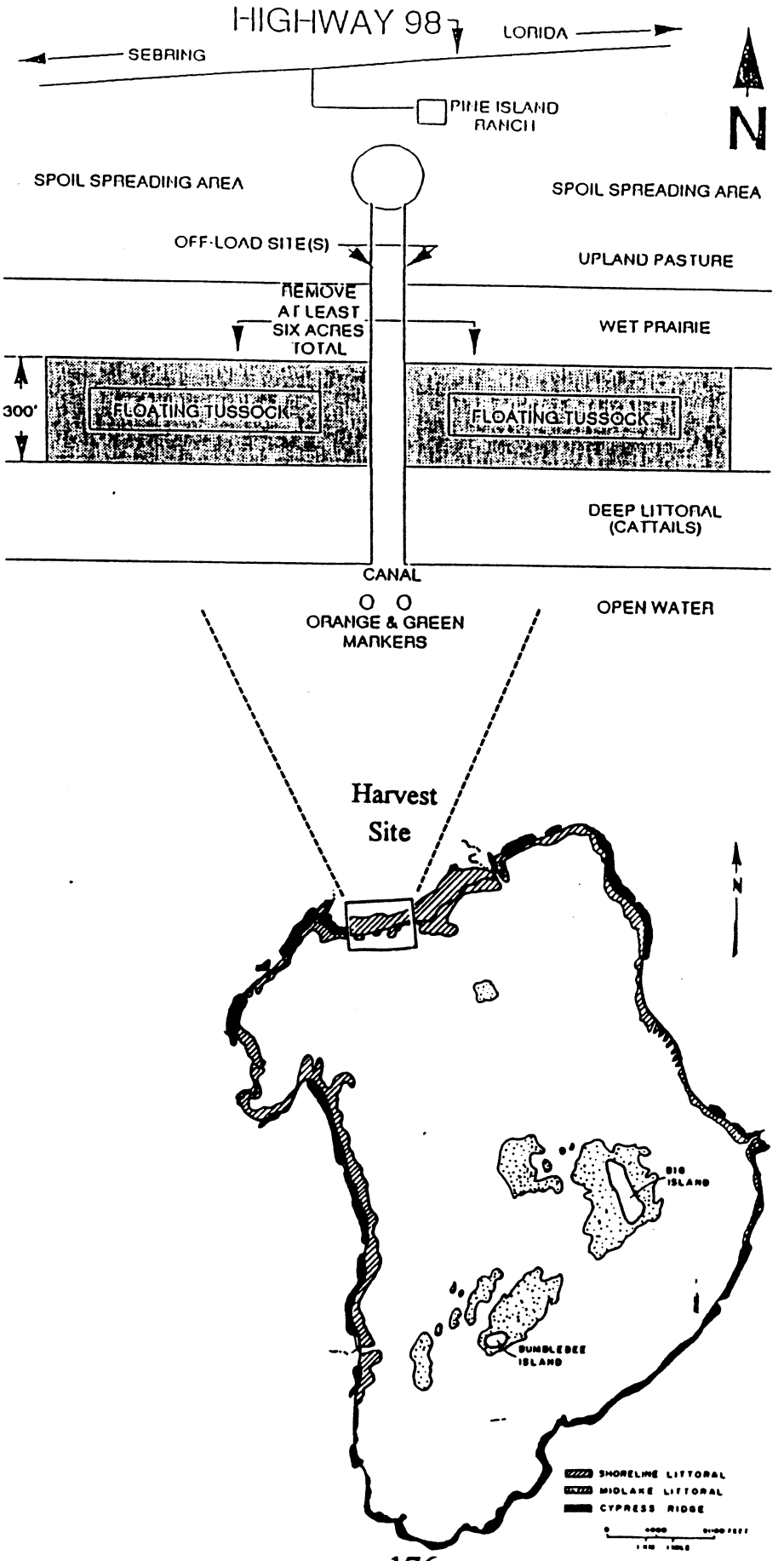
east shorelines, and enhanced mat formation of bulrush, water hyacinth and water lettuce.

METHODS

Beginning 6 February 1995, tussocks within two one-hectare plots separated by a shallow 10 m wide canal and adjacent to a large area of tussocks were harvested (Figure 1). The material was deposited on upland canal banks. The harvest duration was 25 days. Harvest equipment consisted of a "cookie cutter" and a five foot sickle-bar harvester. The cookie cutter chopped the tussocks into small pieces and the harvester transported the material to the deposition sites along the canal bank. Two subsequent cuttings were made with the cookie cutter before the harvester could function efficiently.

Data from seven stations established along four water quality transects were collected to monitor off-site perturbation. Transects were spaced 30 m apart. Pre-harvest (December 5, 1994 and February 1, 1995), harvest (February 8, 1995), and post-harvest (March 20, 1995) water quality samples were collected. Field measurements of pH, specific conductivity, dissolved oxygen (DO), and temperature were made at the surface. All samples were stored on ice, transported to the Florida Game and Fresh Water Fish Commission (GFC) chemistry laboratory, and analyzed within the appropriate holding times as outlined in the Quality Assurance/Quality Control methods of the American Public Health Association (APHA 1989). Chlorophyll *a* samples were filtered in the lab within 24 hours of collection, and the filters were then frozen and analyzed within 30 days. Water samples for turbidity, chlorophyll *a*, total nitrogen (TN), and total phosphorus (TP) were determined by use of standard methods (APHA 1989). Relationships between water quality samples collected during pre-harvest, harvest, and post-harvest and transects were examined using one-way analysis of variance (ANOVA).

Figure 1. Mechanical harvesting sites of floating plant tussock communities, and littoral area distribution in Lake Istokpoga.



RESULTS AND DISCUSSION

A total of 2.0 hectares of floating tussocks were removed in two 1.0 ha plots. Tussock thickness was a uniform 30 cm from the rooted pickerelweed marsh near shore to the rooted cattail band bordering the lakeward edge. Beneath the floating tussock mat, a 30 cm thick layer of flocculent organic material overlaid a sand bottom. The water depth in the plots was approximately 0.5 m shoreward increasing to 1.5 m lakeward. Plant species composition was similar in each plot. Approximately 85% of the plots had dense plant coverage and the remaining 15% was sparsely covered or void of vegetation. The common tussock plant species observed in the test plots were alligator weed (*Alternanthera philoxeroides*), azola (*Azolla caroliniana*), bladderwort (*Utricularia sp.*), broom sedge (*Andropogon sp.*), cattail, bulrush, pickerleweed, water lettuce, water hyacinth, smartweed (*Polygonum sp.*), common salvinia (*Salvinia rotundifolia*), water spider orchid (*Habenaria repens*), water primrose (*Ludwigia sp.*), water pennywort (*Hydrocotyle umbellata*) and wax myrtle (Table 1).

Giant bulrush, cattail, hydrilla, maidencane and spatter dock were the most common plants in the littoral zone (Table 2). The lakeward extent of littoral zone varies with location and plant species. Bulrush extend into the lake to bottom elevation of about 10.8 m msl, and cattail to a bottom of 10.9 m msl. Along the south and southeast shore, the average lakeward extent of the littoral zone is to the 10.6 m msl contour and cypress (*Taxodium distichum*) grow as deep as 10.0 m msl (Milleson 1978).

Mean values of selected water quality parameters at each transect are presented in Table 3. Post-harvest DO differences in the harvest site were minor but statistically different ($p < 0.05$). The pH and electrical conductivity values increased after harvesting of tussocks. This may be due to the mixing of the phytoplankton biomass and high concentrations of dissolved solids released during the harvesting process.

Table 1. Common tussock plant species observed in the test plots.

Common Name	Scientific Name
Alligator weed	*Alternanthera philoxeroides*
Azola	*Azolla caroliniana*
Bladderwort	*Utricularia sp.*
Broom sedge	*Andropogon sp.*
Bulrush	*Scirpus cubensis*
Cattail	*Typha latifolia*
Common salvinia	*Salvinia rotundifolia*
Dog-fennel	*Eupatorium capillifolium*
Pickerelweed	*Pontenderia cordata*
Smartweed	*Polygonum sp.*
Umbrella sedge	*Cyperus articulatus*
Water spider orchid	*Habenaria repens*
Water primroses	*Ludwigia sp.*
Water pennywort	*Hydrocotyle umbellata*
Water lettuce	*Pistia stratiotes*
Water hyacinth	*Eichornia crassipes*
Wax myrtle	*Myrica cerifera*

Table 2. Common plant species in Lake Istokpoga littoral area. Occurrence of plants is: A = Abundant, C = Common, and P = Present (modified from Milleson 1978).

Common name	Scientific name	Distribution
Bald cypress	*Taxodium distichum*	A
Cattail	*Typha latifolia*	A
Giant bulrush	*Scirpus californicus*	A
Hydrilla	*Hydrilla verticillata*	A
Maidencane	*Panicum hemitomon*	A
Spatter dock	*Nuphar luteum*	A
Duck potato	*Sagittaria lancifolia*	C
Pickerelweed	*Pontederia cordata*	C
Primrose willow	*Ludwigia peruviana*	C
Torpedo grass	*Panicum repens*	C
Umbrella grass	*Fuirena scirpoidea*	C
Water hyacinth	*Eichhornia crassipes*	C
Bladderwort	*Utricularia sp.*	P
Common three square bulrush	*Scirpus americanus*	P
Cordgrass	*Spartina bakeri*	P
Eel grass	*Vallisneria americana*	P
Smartweed	*Polygonum sp.*	P
Southern naiad	*Najas guadalupensis*	P
Spike rush	*Eleocharis cellulosa*	P
Water pennywort	*Hydrocotyle umbellata*	P
Water fern	*Salvinia rotundifolia*	P
Watergrass	*Hydrochloa caroliniensis*	P
Willow	*Salix caroliniana*	P
Yellow eyed grass	*Xyris sp.*	P

Table 3. Mean values of selected water quality parameters for samples taken from four transects during pre-harvest (Dec. 5, 94 and Feb. 1, 95), harvest (Feb. 8, 95) and post-harvest (Mar. 20, 95) periods. Values with a letter not in common are significantly different at $p < 0.05$.

Parameter	Pre	Harvest	Post	Transect
Temperature (°C)	17.9^{a}	13.5^{b}	23.4^{c}	Harvest
	22.3^{a}	13.9^{b}	25.4^{ac}	30 m
	19.3^{a}	13.4^{b}	26.4^{c}	61 m
	17.6^{a}	13.8^{b}	26.9^{c}	91 m
DO (mg/l)	4.39^{ab}	5.49^{a}	3.61^{b}	Harvest
	5.9	5.97	6.0	30 m
	5.43^{a}	4.07^{b}	7.69^{c}	61 m
	3.08^{a}	3.02^{a}	5.52^{b}	91 m
pH	5.33^{a}	5.24^{ab}	6.54^{c}	Harvest
	6.24^{a}	5.17^{b}	6.73^{ac}	30 m
	5.33^{a}	5.0^{b}	6.97^{c}	61 m
	5.08^{a}	4.92^{b}	7.2^{c}	91 m
Conductivity	111.1^{a}	99.3^{b}	136.9^{c}	Harvest
(μm/cm)	130.8^{a}	101.3^{b}	140.3^{c}	30 m
	113.1^{a}	100.4^{b}	143.9^{c}	61 m
	112.0^{a}	101.8^{b}	146.8^{c}	91 m
Turbidity (NTU)	0.771^{a}	1.971^{b}	0.786^{ac}	Harvest
	1.023^{a}	1.986^{b}	0.657^{ac}	30 m
	0.957	0.971	0.700	61 m
	0.617	0.883	0.583	91 m
TN (mg/l)	1.245^{a}	0.979^{b}	0.970^{bc}	
TP (mg/l)	0.049^{a}	0.031^{b}	0.053^{c}	
Chlorophyll *a* (mg/m^3)	16.04^{a}	3.89^{b}	7.79^{bc}	

Chlorophyll *a*

A post-harvest decrease in chlorophyll *a* concentration was observed during our study (Table 3). Similar trends in algal biomass were observed following commencement of the harvesting program in Chemung Lake (Wile 1978). Canfield et al. (1984) found in Lake Pearl and Lake Wales, Florida that after the loss of aquatic macrophytes there was a functional and structural shift in phytoplankton and chlorophyll *a*. The change in chlorophyll *a* concentrations over 10 μg/l occurred before the abundance of aquatic macrophytes was reduced below 50% areal coverage. They were not certain of the causative mechanisms for relations between aquatic macrophytes and chlorophyll *a* concentrations. Several factors may be involved in reducing chlorophyll *a* concentrations during harvest period both separately, in combination, and include: off site mixing, reaeration, a reduction in nutrient cycling, increased turbidity, and settling of plant material to the aerobic portion of the water column.

Nutrients

Mean total phosphorus concentrations were significantly different in pre-harvest, harvest, and post-harvest samples (Table 3). Following harvest, TP concentrations increased indicating phosphorus was most likely tied up in microbial or algal biomass or in detritus prior to harvest and was released during harvesting of tussocks. The average concentrations of ortho phosphorus (OP) in the pre-harvest, harvest, and post-harvest periods were 0.01 mg/l, 0.009 mg/l, and 0.008 mg/l, respectively. Since aquatic plants are generally believed to obtain their phosphorus both from the water and sediments (Bannerman 1973, Brooker 1976), it is likely that harvesting may have interrupted the process of obtaining phosphorus from the sediments as well as from the water.

Total organic nitrogen (TON) concentrations during pre-harvest, harvest, and post-harvest periods were 1.03 mg/l, 0.86 mg/l, and 0.95 mg/l, respectively. Total nitrogen concentrations decreased significantly during

harvest and post-harvest periods (Table 3) and pre-harvest TN concentrations were significantly higher than previously reported for the lake (Alam et al. in press, Milleson 1978). The mean concentrations of NH_3-N during pre-harvest, harvest and post-harvest samples were 0.21 mg/l, 0.11 mg/l, and 0.01 mg/l respectively. Milleson (1978) reported an average of 0.03 mg/l NH_3-N concentration for Lake Istokpoga. The concentrations of NH_3-N during pre-harvest and harvest were 600% and 250%, respectively, higher than the findings of Milleson (1978) and were 110% and 10% higher respectively, than the existing concentrations in the lake (Alam et al. in press). The higher concentrations of NH_3-N at pre-harvest show anoxic conditions because the concentrations of NH_3-N greatly increase under anoxic conditions. The concentrations of NO_3-N at pre-harvest, harvest, and post-harvest period were 0.008 mg/l, 0.016 mg/l, and 0.007 mg/l respectively. An increase in NO_3-N during harvest may be due to an increase in DO concentrations. The NO_3-N is the most highly oxidized form of nitrogen and is usually most abundant form of combined inorganic nitrogen in lakes.

Turbidity

During harvesting a significant increase in turbidity occurred up to 30 m from the plot (Table 3). The post-harvest turbidity decreased to pre-harvest level indicating that turbidity is a short-term problem that occurs during harvesting of tussocks. Harvesting caused mixing of soluble and insoluble substances in the water column. Also, with an increase in turbidity, levels of dissolved solids increased in the water resulting in an increase in electrical conductivity. Water movements are not only important in the physical movement of organic matter, but they are critical in the vertical transport of organic matter, mineralized water, and nutrients from the bottom of the water column and from shallower depths and littoral regions to the deeper open water areas.

The effects of harvesting on temperature, dissolved oxygen, and pH were negligible. The only appreciable effects were a temporary increase in

turbidity and a decrease in TP, TN, and chlorophyll *a* concentration which are believed to be indicative of algal density. This was expected given the well documented effects of aquatic vegetation on water quality (Edwards 1968, Fitzgerald 1969, Nichols 1973, Dale and Gillespie 1977).

It is not possible to draw widespread conclusions on water chemistry effects based on this single test. However, this test was conducted under operational conditions and there is no reason to believe that results would not apply to larger tussock removal projects. Given the plant density harvested, it seems likely that the turbidity effects observed up to 30 m from harvesting are likely to result from such harvesting for other commonly performed small area harvesting operations. The harvesting need not be a weapon of destruction, but a tool to build more useful and diverse aquatic systems.

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BIRD COMMUNITIES AT WASTEWATER PONDS IN SOUTHEASTERN IDAHO

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ABSTRACT

Disposal of aqueous waste is often accomplished through discharge into constructed ponds. Because some of these ponds are heavily used by wildlife, assessments of their dangers or benefits begins with quantitative analysis of the wildlife communities which inhabit them. Constructed ponds at the Idaho National Engineering Laboratory (INEL) vary in size and shape and contain diluted sanitary waste (n=8), industrial waste (n=4), and radioactive waste (n=3). We documented bird use of wastewater ponds from August 1989 through July 1991. Birds were surveyed 8 times per month from March through October, and 4 times per month from November through February. One hundred thirty-one bird species were observed within 20 m of the ponds. Species richness was greatest in May and September. Species diversity was greatest in May and August. The most common guilds were dabblers and surface dippers, upland ground gleaners on animal food, aerial foragers, upland ground gleaners on plant food, aquatic ground gleaners, and divers. The most species-rich and diverse guild was upland ground gleaners on animal food. This guild was positively associated with grass and forb cover within 20 m of a pond.

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Dabblers and surface dippers were positively associated with pond surface area. Aquatic ground gleaners were positively associated with bare shoreline. Bird communities at wastewater ponds were diverse, and ponds were heavily used, especially during migrations.

INTRODUCTION

Water bodies created for wastewater treatment vary in appearance and function from artificial appearing settling ponds to ponds with abundant macrophytes and irregular shorelines designed for tertiary sewage treatment; these latter wastewater treatment sites may appear similar to palustrine emergent wetlands. Wastewater ponds often attract large numbers of birds and other wildlife species. With the rapid rate at which wastewater ponds are being created throughout many nations, quantitative information on wildlife use of these ponds becomes important in evaluating potential positive as well as negative influences.

The Idaho National Engineering Laboratory (INEL) is one of 5 National Environmental Research Parks, which was designated, in part, for study of the interaction between energy production and wildlife (United States Department of Energy 1985). Research at the INEL produces sanitary, industrial, and radioactive waste, much of which is discharged into wastewater ponds. In the semiarid desert of the upper Snake River Plain, where the INEL lies, many water sources are ephemeral or temporary, and permanent sources are commonly diverted for agricultural purposes. Other than game watering cisterns, waste disposal ponds are the only permanent surface water at the INEL.

Birds may become contaminated through association with some of the ponds, and may transport pond contaminants during migration and daily movements (Craig et al. 1979, Markham and Halford 1982, Connelly and Markham 1983). Although studies have shown no radiation hazard is posed to birds which use the ponds (Millard et al. 1990, Halford et al. 1982) or to secondary consumers (Craig et al. 1979, Markham and Halford

1982, Connelly and Markham 1983, Browers 1983, Howe and Flake 1988), it is not known to what extent most bird species use INEL ponds. Species which feed similarly (i.e. guilds) should have the same potential to be exposed to toxins and the same potential to benefit from the ponds. Our objectives were to (1) document bird species richness and diversity at wastewater ponds in a high desert, (2) determine the composition of feeding guilds at ponds, and (3) determine pond characteristics which are associated with species richness, and use level by the most common guilds.

STUDY SITE

The INEL encompasses 231,600 ha on the western edge of the Snake River Plain in southeastern Idaho. For further description and a map of the INEL, see Flake and Cieminski (1996) in these proceedings.

Wastewater ponds at the INEL contained sanitary waste (8 ponds), industrial waste (4), radioactive waste (1), or radioactive and industrial waste (2). The ponds which contained radioactive and industrial waste were considered radioactive waste ponds. Three ponds had 2 basins, which were pumped into alternately, on approximately a 6-month rotation. Because only 1 of the 2 basins contained water at a time, we considered the 2 basins as one pond so they could be compared to ponds which contained water year-round. The dual basins of each pair lay side by side and were identical in shape, water chemistry, surrounding vegetation, and size, with one exception; the surface area of one basin of the Chemical Processing Plant industrial pond when full was 2.3 ha, whereas the other basin was 1.6 ha.

METHODS

Counts.--Diurnal and nocturnal counts were conducted monthly on each pond, August 1989 through July 1991. The length of each count was determined by how long it took to observe the entire pond and record all

birds within 20 m of the pond. Ponds were circled by truck or by walking to flush birds on the shore which might otherwise have been overlooked. Because pond areas were sparsely vegetated, counts were thorough. Therefore, we do not believe the length of the count affected the number of individuals or species observed. Counts were not conducted during inclement weather.

From March through October, 5 diurnal and 3 nocturnal counts were conducted per month on each pond. Due to the shortened daylight hours and decrease in bird activity around the frozen ponds, monthly surveys were reduced to 3 diurnal and 1 nocturnal counts per month from November through February. Due to low (60 observations) pond use in January and February 1990, the ponds were not surveyed during January and February 1991.

Five diurnal (D) and 3 nocturnal (N) time periods were established during which counts were conducted: sunrise (D1), midmorning (D2), midday (D3), midafternoon (D4), sunset (D5), dusk (N1), mid-night (N2), and pre-dawn (N3). Each pond was sampled once per month during each time period. November through February, diurnal counts were only conducted during D1, D3, and D5, and nocturnal counts only during N1.

Ponds were scheduled to be counted only once per day because observer presence at the ponds could influence later counts at the pond.Counts were not spaced throughout the month because surveys were also used to assess diel use in a larger study (Cieminski 1993), and seasonal variation within the month might have interfered with detection of diel variation.

For analysis, counts D1-D5 were used for all birds except owls and caprimulgids. For owls and the common poor-will (Phalaenoptilis nuttallii) counts D5, N1, N2, N3, and D1 were used. Because common nighthawks (Chordeiles minor) were active throughout the day, observations in all 8 time periods were totaled and multiplied by 5/8 to make the number of observations comparable to the remaining species, for which 5 counts were used.

Pond characteristics.--Water surface area, pond shoreline length, and maximum depth were taken from blueprints, aerial photos, and on-site tape measurements. Percent surface area covered by emergents and percent bare shoreline (no vegetation > 15 cm tall within 1 m of the water) were visually estimated. Percent of the shoreline that was cobble (as opposed to gravel and dirt) was also estimated, as was percent of the shoreline that was plastic. The aforementioned characteristics were recorded on a monthly basis and the 22-month mean used for analysis.

Six 20-m line-intercepts were run at each pond (Canfield 1941). Each intercept line was placed 1 m above the ground to estimate percent coverage of shrubs and grasses/forbs. The 6 intercept lines were evenly spaced around the pond, perpendicular to the shoreline. Percent coverage of vegetation inside the fences around radioactive ponds was visually estimated.

Water column invertebrates were sampled from June 1990 through May 1991 as described in Cieminski and Flake (1995). Each pond was ranked 1-5 (lowest to highest) according to the number of microinvertebrates (excluding Rotifera) collected per sample, and in the same manner for macroinvertebrates.

Guilds.--Bird species were grouped into guilds according to their primary foraging technique (Ehrlich et al. 1988), with the following exceptions. Dabblers and surface dippers were combined. Upland ground gleaners were separated into those which feed primarily on plant material and those which feed primarily on animal material. Species which would have been the only species, or one of only 2 species, in their guild were included with the most similar guild: American avocet (Recurvirostra americana) (sweeper) with probers, black tern (Chlidonias niger) (aquatic hawker) with aerial foragers, belted kingfisher (Ceryle alcyon) (high diver) with stalkers, California gull (Larus californicus) (low diver) with dabblers and surface dippers, and sage grouse (Centrocercus urophasianus) (foliage browser) with upland ground gleaners on plant material, mountain

bluebirds (Sialia currucoides) (swooper) with hawkers, and shrikes (swoopers) with raors. All raptors were grouped together regardless of feeding technique.

Statistical analysis.--Bird species diversity (months combined), species diversity within each guild (ponds and months combined), and guild diversity by pond (months combined) were computed with the Shannon Wiener index ($\log_2$) (Brewer 1994). Species richness, guild diversity, and guild size were compared with selected pond characteristics (Table 3). For each comparison, the 15 ponds were divided into 3 groups of 5 ponds each - low, medium, and high richness, diversity, or use. Logistic regression (SAS Institute Inc. 1988) was used to compare pond characteristics between groups. Variables entering the logistic regression at $P \leq 0.05$ were considered significant.

RESULTS

Species richness and diversity.--During 22 months of observations, 131 bird species were observed at study ponds. For 21 species, only 1 individual was observed: in the diver guild, common loon (Gavia immer); in the stalker guild, black-crowned night-heron (Nycticorax nycticorax); in the aquatic ground gleaner guild, cattle egret (Bubulcus ibis); in the dabbler and surface dipper guild, trumpeter swan (Cygnus buccinator) and

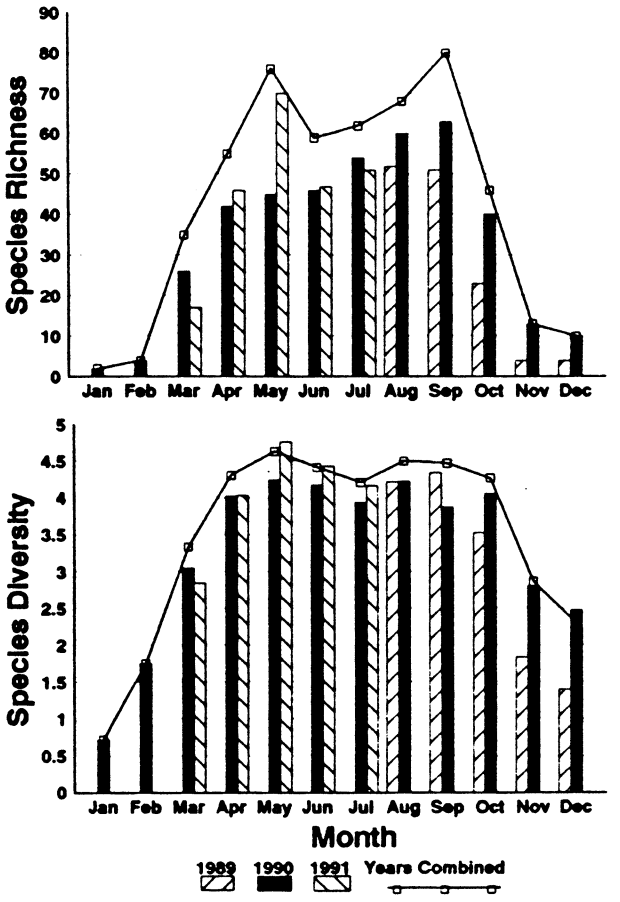

Figure 1. Seasonal change in avian species richness and diversity at 15 wastewater ponds at the Idaho National Engineering Laboratory, August 1989 - July 1991.

Bonaparte's gull (Larus philadelphia); in the prober guild, Virginia rail (Rallus limicola); in the raptor guild, golden eagle (Aquila chrysaetos), Swainson's hawk (Buteo swainsoni), and long-eared owl (Asio otus); in the upland ground gleaner on animal food guild, ring-necked pheasant (Phasianus colchicus), rock wren (Salpinctes obsoletus), American tree sparrow (Spizella arborea), and lapland longspur (Calcarius lapponicus); in the hawker guild, Lewis' woodpecker (Melanerpes lewis), western flycatcher (Empidonax difficilis), and Townsend's solitaire (Myadestes townsendi); in the foliage gleaner guild, ruby-crowned kinglet (Regulus calendula), yellow-breasted chat (Icteria virens), western tanager (Piranga ludoviciana), American goldfinch (Carduelis tristis), and lesser goldfinch (Carduelis psaltria).

Overall, species richness peaked in May and September, and species diversity peaked in May and August (Fig. 1). But some species were not observed during any peak month (Table 1). Species richness was not associated (logistic regression) with any characteristic measured.

Table

1. Total observations, residence time, and guild affiliation of bird species observed at Idaho National Engineering Laboratory wastewater ponds, August 1989 - July 1991.

Species[a] [total obs][b]	Residence time and (months absent)[c]	Guild[d]
Pied-billed grebe *(Podilymbus podiceps)* [6]	16 Apr-30 Oct (Jul-Aug)	Diver
Eared grebe *(Podiceps nigricollis)* [273]	8 Apr-27 Oct	Diver
Western grebe *(Aechmophorus occidentalis)* [9]	9 May-27 Oct (Jun-Aug)	Diver
Great blue heron *(Ardea herodias)* [17]	23 Mar- 3 Sep (May-Jul)	Stalker
Canada goose (*Branta canadensis)* [31]	18 Mar- 8 Oct (Jul)	Dabl.&dip.
Wood duck (*Aix sponsa)* [17]	8 Apr-27 Oct (Jul-Sep)	Dabl.&dip.
Green-winged teal *(Anas crecca)*[e] [2893]	13 Mar- 7 Nov	Dabl.&dip.
Mallard *(Anas platyrhynchos)* [1994]	16 Jan-21 Dec	Dabl.&dip.
Northern pintail *(Anas acuta)*[e] [289]	7 Mar-27 Oct	Dabl.&dip.
Blue-winged teal *(Anas discors)* [328]	16 Apr-20 Oct	Dabl.&dip.
Cinnamon teal *(Anas cyanoptera)* [252]	19 Mar- 4 Sep	Dabl.&dip.
Northern shoveler *(Anas clypeata)*[e] [1469]	27 Mar-30 Oct	Dabl.&dip.
Gadwall (*Anas strepera)* [132]	30 Mar- 2 Nov	Dabl.&dip.
American wigeon *(Anas americana)*[e] [787]	18 Mar- 2 Nov	Dabl.&dip.
Canvasback *(Aythya valisineria)* [19]	23 Mar-15 Jul	Diver
Redhead *(Aythya americana)*[e] [506]	19 Feb-26 Oct	Diver
Ring-necked duck *(Aythya collaris)* [151]	20 Mar-29 Oct (Jul)	Diver
Lesser scaup *(Aythya affinis)* [289]	25 Mar-30 Oct (Aug)	Diver
Surf scoter *(Melanitta perspicillata)* [5]	12 Oct-27 Nov	Diver
Common goldeneye *(Bucephala clangula)* [142]	5 Mar-10 Dec (Jun-Sep)	Diver

Table 1. Continued.

Species[a] [total obs][b]	Residence time and months absent ()[c]	Guild[d]
Barrow's goldeneye *(Bucephala islandica)* [23]	4 Apr-13 May	Diver
Bufflehead *(Bucephala albeola)* [100]	26 Mar-28 Nov (Sep)	Diver
Hooded merganser *(Lophodytes cucullatus)* [8]	20 Mar-18 Apr	Diver
Common merganser *(Mergus merganser)*[8]	19 Mar-29 Nov (Jun,Aug-Oct)	Diver
Red-breasted merganser *(Mergus serrator)*[5]	16 Apr-15 Jun	Diver
Ruddy duck[e] *(Oxyura jamaicensis)* [725]	4 Apr-29 Nov	Diver
Northern harrier *(Circus cyaneus)* [166]	11 Mar-27 Dec	Raptor
Red-tailed hawk *(Buteo jamaicensis)* [2]	4 May- 8 Aug (Jul)	Raptor
Ferruginous hawk *(Buteo regalis)* [3]	8 Jul-11 Jul	Raptor
Rough-legged hawk *(Buteo lagopus)* [2]	29 Nov- 5 Mar (Dec)	Raptor
American kestrel *(Falco sparverius)* [3]	12 Mar- 3 Nov (Jun, Aug)	Raptor
Prairie falcon *(Falco mexicanus)* [4]	17 Apr-26 Dec (May, Oct)	Raptor
Gray partridge *(Perdix perdix)*[6]	7 Mar-12 Nov (Apr-Oct)	Up.glean.(p)
Sage grouse *(Centrocercus urophasianus)* [154]	6 Mar-12 Dec (Apr)	Up.glean.(p)
Sora *(Porzana carolina)*[8]	9 May- 3 Sep (Jun-Aug)	Aq.glean.
American coot[e] *(Fulica americana)* [452]	19 Mar-29 Nov	Dabl.&dip.
Killdeer[e] *(Charadrius vociferus)* [1338]	3 Mar-19 Dec (Nov)	Aq.glean.
American avocet *(Recurvirostra americana)* [322]	18 Apr- 7 Sep	Prober
Greater yellowlegs *(Tringa melanoleuca)* [11]	22 Mar-24 Aug (May)	Prober
Lesser yellowlegs *(Tringa flavipes)*[5]	27 Apr-16 Aug (Jul)	Prober
Solitary sandpiper *(Tringa solitaria)*[12]	10 May-11 Sep (Jun)	Prober
Willet *(Catoptrophorus semipalmatus)*[40]	16 Apr-22 Jul	Prober
Spotted sandpiper[e] *(Actitis macularia)* [626]	4 May-24 Sep	Aq.glean.
Western sandpiper *(Calidris mauri)* [779]	9 May-15 Sep (Jun)	Aq.glean.
Least sandpiper *(Calidris minutilla)* [61]	16 Apr- 4 Sep (Jun)	Aq.glean.
Baird's sandpiper *(Calidris bairdii)* [58]	2 Apr- 6 Sep (Jun-Jul)	Aq.glean.

Table 1. Continued.

Species[a] [total obs][b]	Residence time and months absent ()[c]	Guild[d]
Long-billed dowitcher *(Limnodromus scolopaceus)*[7]	8 May-31 Jul (Jun)	Prober
Common snipe *(Gallinago gallinago)* [9]	19 Mar-24 Aug (May-Jun)	Prober
Wilson's phalarope *(Phalaropus tricolor)* [889]	27 Apr- 5 Sep	Dabl.&dip.
Red-necked phalarope *(Phalaropus lobatus)* [165]	10 May-29 Sep (Jun)	Dabl.&dip.
Franklin's gull *(Larus pipixcan)* [4]	8 May-25 Jul	Aq.glean
Ring-billed gull *(Larus delawarensis)* [34]	22 Mar- 2 Nov (Jul)	Aq.glean.
California gull *(Larus californicus)* [26]	7 May-12 Oct (Aug)	Dabl.&dip.
Black tern *(Chlidonias niger)* [7]	9 May-31 Jul	Aer.forager
Rock dove *(Columba livia)* [46]	29 Mar-29 Oct	Up.glean.(p)
Mourning dove *(Zenaida macroura)* [1759]	16 Apr-26 Sep	Up.glean.(p)
Burrowing owl *(Athene cunicularia)* [9]	17 Aug-11 Dec	Raptor
Short-eared owl *(Asio flammeus)* [27]	23 Apr-11 Nov (May, Oct)	Raptor
Common nighthawk *(Chordeiles minor)* [388]	10 Jun-13 Sep	Aer.forager
Common poorwill *(Phalaenoptilus nuttallii)* [11]	26 Jun-27 Jul	Aer.forager
Belted kingfisher *(Ceryle alcyon)* [2]	24 Apr- 8 Oct (Jun-Sep)	Stalker
Northern flicker *(Colaptes auratus)* [2]	26 Sep-27 Sep	Up.glean.(a)
Western wood-peewee *(Contopus sordidulus)* [2]	6 Apr-15 Jun (May)	Hawker
Gray flycatcher *(Empidonax wrightii)* [4]	3 Sep- 7 Sep	Hawker
Say's phoebe *(Sayornis saya)* [20]	18 Apr-27 Sep	Hawker
Western kingbird *(Tyrannus verticalis)* [14]	15 May-21 Aug	Hawker
Eastern kingbird *(Tyrannus tyrannus)* [4]	22 May-31 Jul	Hawker
Horned lark *(Eremophila alpestris)* [1012]	16 Jan-28 Dec	Up.glean.(p)
Tree swallow *(Tachycineta bicolor)* [181]	21 Mar-25 Aug	Aer.forager
Violet-green swallow *(Tachycineta thalassina)* [356]	3 Apr- 5 Sep	Aer.forager
Northern rough-winged swallow *(Stelgidopteryx serripennis)* [396]	8 Apr- 5 Sep	Aer.forager
Bank swallow *(Riparia riparia)* [591]	6 May-11 Sep	Aer.forager

Table 1. Continued.

Species[a] [total obs][b]	Residence time and months absent ()[c]	Guild[d]
Cliff swallow *(Hirundo pyrrhonota)* [74]	6 May- 6 Sep	Aer.forager
Barn swallow[e] *(Hirundo rustica)* [1794]	8 Apr-10 Oct	Aer.forager
Black-billed magpie[e] *(Pica pica)* [368]	30 Jan-21 Dec	Up.glean.(a)
Common raven *(Corvus corax)* [18]	6 Mar-28 Dec	Up.glean.(a)
House wren *(Troglodytes aedon)* [2]	21 Aug- 4 Sep	Up.glean.(a)
Marsh wren *(Cistothorus palustris)* [41]	8 Apr-11 Oct (Jun-Jul)	Up.glean.(a)
Mountain bluebird *(Sialia currucoides)* [3]	7 Mar-19 Jul (Apr-Jun)	Hawker
American robin *(Turdus migratorius)* [17]	8 Mar- 1 Nov (Jul)	Up.glean.(a)
Sage thrasher[e] *(Oreoscoptes montanus)* [76]	7 May- 4 Sep	Up.glean.(a)
Water pipit *(Anthus spinoletta)* [146]	12 Mar-12 Oct (Jun-Aug)	Up.glean.(a)
Northern shrike *(Lanius excubitor)* [3]	12 Dec-18 Apr (Jan-Feb)	Raptor
Loggerhead shrike *(Lanius ludovicianus)* [10]	9 May-29 Sep (Jun)	Raptor
Bohemian waxwing *(Bombycilla garrulus)* [10]	14 Nov-27 Dec	Folia.glean.
European starling *(Sturnus vulgaris)* [731]	5 Mar-13 Nov	Up.glean.(a)
Solitary vireo *(Vireo solitarius)* [3]	5 Sep- 7 Sep	Folia.glean.
Yellow warbler *(Dendroica petechia)* [7]	9 May-10 Jun	Folia.glean.
Yellow-rumped warbler *(Dendroica coronata)* [135]	26 Apr-12 Oct (Jun-Aug)	Folia.glean.
MacGillivray's warbler *(Oporornis tolmiei)* [14]	21 May-25 Sep (Jun-Jul)	Folia.glean.
Common yellowthroat *(Geothlypis trichas)* [6]	8 May- 5 Oct (Jun-Aug)	Folia.glean.
Wilson's warbler *(Wilsonia pusilla)* [34]	13 May-26 Sep (Jun-Jul)	Folia.glean.
Lazuli bunting *(Passerina amoena)* [4]	3 Sep-4 Sep	Up.glean.(a)
Chipping sparrow *(Spizella passerina)* [61]	13 May-27 Sep (Jun-Jul)	Up.glean.(a)
Brewer's sparrow *(Spizella breweri)* [495]	6 May-29 Sep (Jun)	Up.glean.(a)
Vesper sparrow *(Pooecetes gramineus)* [88]	9 Apr-26 Sep	Up.glean.(a)
Sage sparrow[e] *(Amphispiza belli)* [202]	7 Mar-29 Sep (May-Jun)	Up.glean.(a)
Lark bunting *(Calamospiza melanocorys)* [3]	7 Aug- 8 Aug	Up.glean.(a)
Savannah sparrow *(Passerculus sandwichensis)* [30]	3 Apr-12 Oct (Jun-Aug)	Up.glean.(a)

Table 1. Continued.

Species[a] [total obs][b]	Residence time and months absent ()[c]	Guild[d]
Song sparrow *(Melospiza melodia)* [25]	7 Mar- 3 Nov (Jun, Aug)	Up.glean.(a)
Lincoln's sparrow *(Melospiza lincolnii)* [12]	8 Apr-15 May	Up.glean.(a)
White-crowned sparrow *(Zonotrichia leucophrys)* [261]	6 May-11 Oct (Jun-Aug)	Up.glean.(a)
Dark-eyed junco *(Junco hyemalis)* [37]	19 Mar-10 Oct (Jun-Aug)	Up.glean.(p)
Red-winged blackbird[e] *(Agelaius phoeniceus)* [276]	7 Mar-12 Oct	Up.gelan.(a)
Western meadowlark[e] *(Sturnella neglecta)* [153]	14 Mar- 1 Nov 1	Up.glean.(a)
Yellow-headed blackbird[e] *(Xanthocephalus xanthocephalus)* [2674]	5 Apr- 6 Oct	Up.glean.(a)
Brewer's blackbird *(Euphagus cyanocephalus)* [761]	13 Apr- 6 Oct	Up.glean.(a)
Brown-headed cowbird *(Molothrus ater)* [831]	20 Apr-27 Oct	Up.glean.(a)
House finch *(Carpodacus mexicanus)* [115]	24 Apr-11 Oct (Jul)	Up.glean.(p)
House sparrow *(Passer domesticus)* [9]	9 Apr- 7 Aug (May-Jun)	Up.glean.(p)

[a] To conserve space, species observed on only 1 occasion are not included in this table, but are given in the text. Nomenclature follows American Ornithologists' Union (1983).

[b] Total observed at 15 ponds during 5 counts per month March-October and 3 counts per month November-February: diurnal and crepuscular counts were summed for diurnal species, crepuscular and nocturnal counts were summed for nocturnal species.

[c] Taken from Cieminski (1993). Earliest arrival and latest departure date observed July 1989 and July 1991, and months not observed between arrival and departure months.

[d] Abbreviated guild names are: Dabl.&dip. = dabblers and surface dippers; Up.glean.(p) = upland ground gleaners on plant food; Aq.glean. = aquatic ground gleaners; Aer.forager = aerial forager; Up.glean.(a) = upland ground gleaners on animal food; Folia.glean. = foliage gleaners.

[e] Observed to nest on the INEL during the study.

Guild composition.--Eleven feeding guilds were seen at study ponds. Six guilds were large, in terms of numbers of birds observed, and 5 were small (Fig. 2). Upland ground gleaners on animal food, dabblers and surface dippers, and divers (3 of the large guilds) had the highest species richness and species diversity.

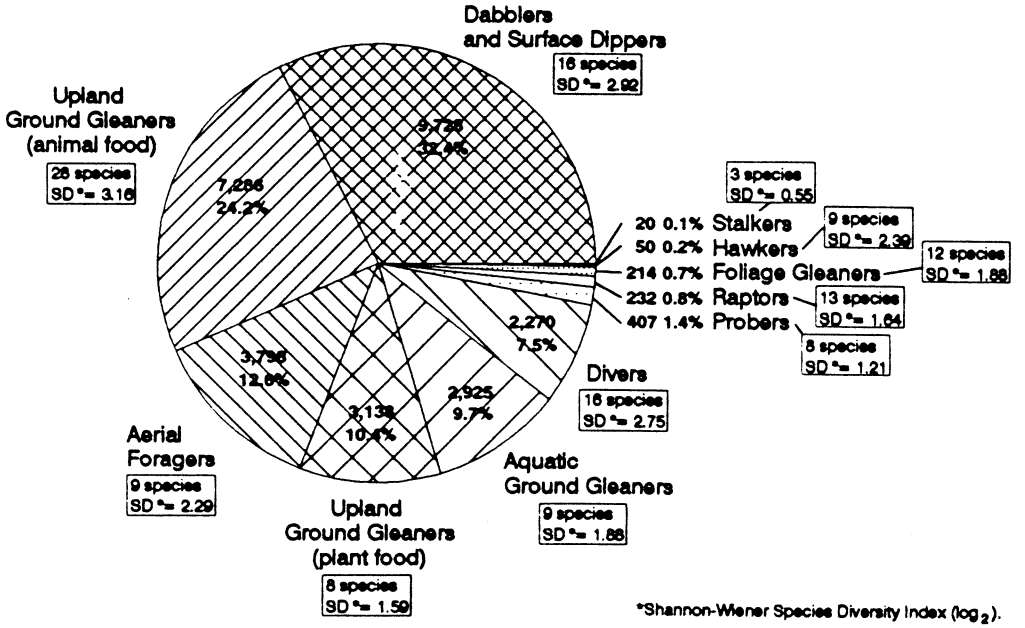

Figure 2. Size, in number of birds observed, of feeding guilds at 15 wastewater ponds at the Idaho National Engineering Laboratory, August 1989 - July 1991. Within-guild species richness and diversity (Shannon-Wiener index $\log_2$) are presented in boxes.

However, we were unable to detect differences in species richness (P=0.75) or species diversity (P=0.33) between the 6 large guilds and 5 smaller guilds, using a median test. Therefore, the larger guilds contained more individuals, but did not comprise more species.

The number of individuals observed peaked in May and August. This was due mostly to dabblers and surface dippers and upland ground gleaners on animal food (the 2 largest guilds) peaking these months (Fig. 3). However, smaller guilds also had peaks in one of these months: aerial foragers in May (1991) and upland ground gleaners on plant food in August. Pond use by aquatic ground

gleaners, however, peaked in July.

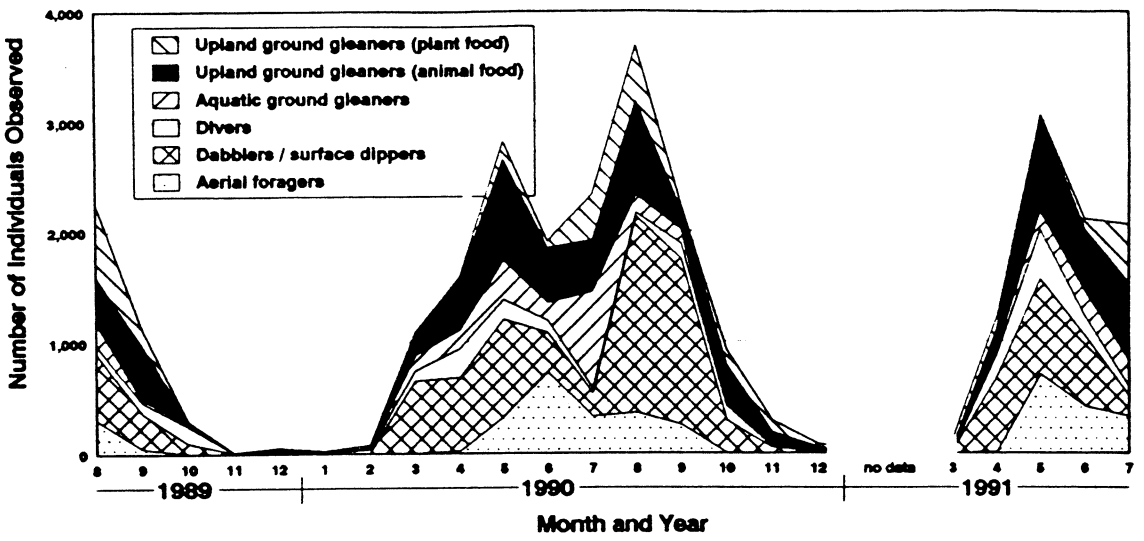

Figure 3. Seasonal change in size of avian feeding guilds observed at the Idaho National Engineering Laboratory, August 1989 - July 1990. Because fewer counts were conducted per pond per month November through February, the number of individuals has been adjusted upward by 5/3 for these months to equalize them to the other months.

Relative distribution of individuals among guilds was similar between sanitary and radioactive ponds, although total numbers differed (Fig. 4). Industrial ponds, however, had higher ($P = 0.02$ and 0.03) proportional use by both guilds of upland ground gleaners than sanitary or radioactive ponds (Fig. 4).

Logistic regression showed upland ground gleaners on animal food were positively associated ($P < 0.05$) with grass and forb cover at the pond (standardized estimate = 0.81). Aquatic ground gleaners were positively associated with percent bare shoreline (standardized estimate = 0.83). Dabbler and surface dipper use was positively associated with surface area (standardized estimate = 1.46). Guild diversity was not associated with any pond characteristic we measured.

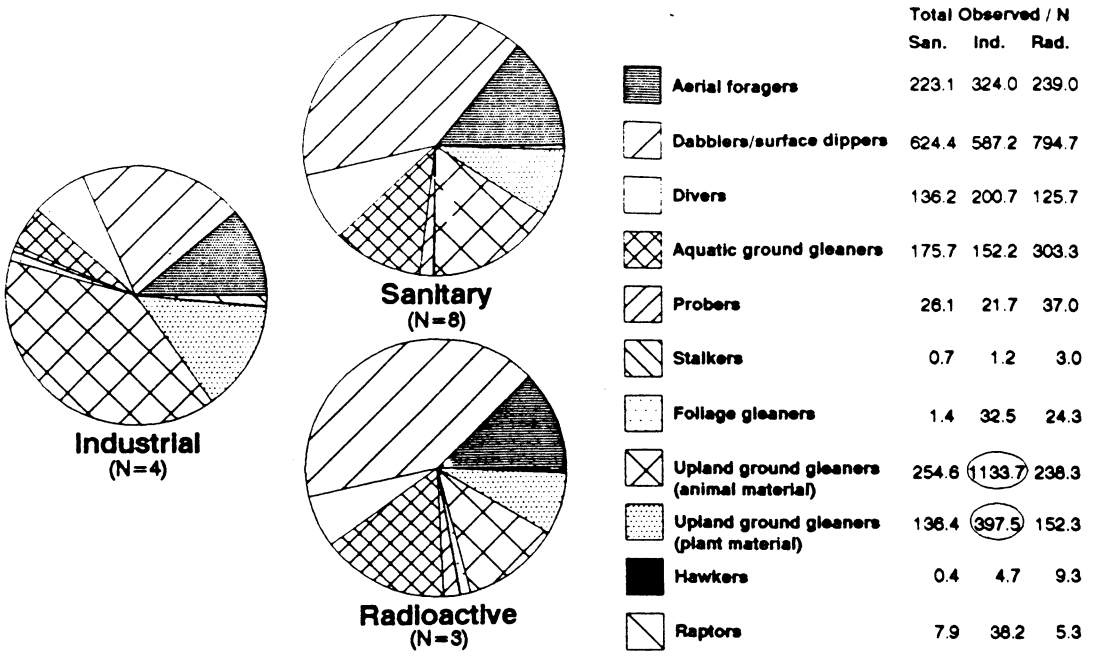

Figure 4. Feeding guild composition of avian communities at sanitary, industrial, and radioactive waste ponds at the Idaho National Engineering Laboratory, August 1989 - July 1991, and comparison of guild size among the 3 pond types. Encircled numbers are significantly higher than other numbers in the same row.

DISCUSSION

Species richness and diversity.--Species richness and diversity from November through February were not directly comparable to other months due to the unequal number of counts conducted. However, it is apparent that these months had the lowest richness and diversity. The most species-rich and diverse months were May, August, and September. This information is important in designing survey schedules for wastewater ponds. For such surveys, time is usually limited, and it is best to sample during months when the most species could be indexed.

Sampling during months of greatest species richness does not assure all species will be observed, however. Twenty-three species were never observed during May or September. Half of these species were accidental or winter residents on the INEL. In either case, they do not figure prominently in transport of pond

contaminants due to their extremely low numbers (accidentals) and the fact that ponds are mostly frozen in winter (winter residents).

Guild composition.--Raptors, being carnivorous, generally do not need to drink (Bartholomew and Cade 1963), which may account for their low use of ponds. It seems evident that there were few stalkers on the INEL because fish, their main prey, were absent from wastewater ponds. Habitat components were also absent for other guilds: few perches for hawkers, no trees for foliage gleaners, and little or no mudflat-like shoreline at most ponds for probers. Although adding these missing habitat aspects might increase use by some guilds, it could decrease use by other guilds.

Dabblers and surface dippers were dominant when analyzing positive and negative influences of wastewater. Although upland ground gleaners dominated in numbers, dabblers and surface dippers are also a large guild, and in addition most are aquatic. Because most are also larger in body size and are game species, future studies of the movements of contaminants through this system should concentrate on dabblers and surface dippers first.

Second in importance in analyzing pond effects are upland ground gleaners on animal food and aquatic ground gleaners. Upland ground gleaners on animal food were often seen feeding at shorelines. This plus being more numerous than aquatic ground gleaners probably makes these 2 guilds equal in benefits and hazards derived from the ponds. Following them would be divers, a smaller guild, but important because their feeding habits bring them into constant contact with pond water and sediment. Individual divers may accumulate more contaminants than individuals in other guilds, but because there are fewer of them, the effect of the guild as a whole is reduced. Of the 6 large guilds, aerial foragers and upland ground gleaners on plant food are probably last in effects sustained from wastewater ponds. However, species in both guilds are dependent on the ponds. Aerial foragers obtain the bulk of their food and water from ponds (Cieminski 1993), and upland ground gleaners on plant food use the ponds daily for drinking water(Bartholomew and Cade 1963).

Wastewater ponds at the INEL were most heavily used by most guilds during

May, August, and September. This information can be used when scheduling maintenance activities. To maximize pond usefulness to most guilds, lengthy maintenance should be avoided during May, August, and September. Conversely, if there is concern about contaminants, these are the months to make the greatest effort to keep birds away from ponds.

Considering seasonal use by guilds and pond characteristic associations together, we discovered that partial drawdowns (where water level manipulation is possible) to benefit aquatic ground gleaners would be most useful in July. Other pond characteristic associations may be useful when designing new ponds. Dabblers and surface dippers should respond better to larger ponds. Upland ground gleaners on animal food should be more numerous at ponds with well vegetated shores. Although we measured grass and forb cover within 20 m of the pond shoreline, most between-pond difference in percent cover was on the immediate shoreline. A narrow band of vegetation appeared sufficient to attract birds, and we do not recommend artificially increasing grass and forb density in surrounding naturally-occurring vegetation.

Difference in guild composition between pond types appeared due to characteristics other than waste type. Industrial ponds may have had greater use by upland ground gleaners on animal food because they were surrounded by greater ($P = 0.052$) mean coverage of grass and forbs than radioactive or sewage ponds.

Most of the species common at INEL wastewater ponds were reported by Merriam (1891) as inhabitants or transients along Birch Creek, the Little Lost River, and the Big Lost River, which flowed seasonally onto the INEL. The ponds may in part replace wetland habitat lost since these sources are now largely diverted for irrigation. It is unquestionable that for better or worse, INEL ponds are heavily used. Suggested topics for further research include assessment of the ponds influence on the species using them - i.e., are wastewater ponds sources or sinks? We suggest that dabblers and surface dippers would be the best guild for further research.

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WATERFOWL USE OF WASTEWATER PONDS ON THE IDAHO NATIONAL ENGINEERING LABORATORY

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ABSTRACT

Wastewater ponds attract a variety of waterfowl to the Idaho National Engineering Laboratory (INEL) in southeastern Idaho. We censused waterfowl on INEL ponds monthly from August 1989 through July 1991. Twenty-four species of waterfowl were observed at these ponds but only 14 were abundant enough to allow evaluation of seasonal use. Peak use of ponds by dabbling ducks (tribe Anatini) occurred in spring (April and May) and in late summer and fall (August and September). Divers, including Aythyini, Mergini, and Oxyurini (tribes), were most abundant on INEL ponds in April and May. Seven species including ruddy ducks (Oxyura jamaicensis), redheads (Aythya americana), northern shoveler (Anas clypeata), northern pintail (A. acuta), American wigeon (A. americana), mallard (A. platyrhynchos), and green-winged teal (A. crecca) nested and reared or attempted to rear broods on the INEL ponds. Logistic regression indicated that most species were positively associated with larger ponds. Other positively associated variables in the various regression models included natural shoreline (not plastic or cobble) and microinvertebrate abundance while negative variables included shallowness (< 15 cm) and distance from the water to the top of the

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surrounding berm or bank (relief). Mallard and green-winged teal broods selected sewage ponds over industrial-radioactive ponds but American wigeon, northern pintail, and northern shoveler broods showed no selection between these pond types.

INTRODUCTION

Constructed ponds are becoming increasingly important for use in industrial and municipal wastewater treatment and represent one of the most common types of created wetlands (Moshiri 1993). These ponds vary in characteristics from settling ponds to ponds with abundant aquatic macrophytes that enhance removal of nutrients and break down organic materials (Brix 1993). Many of these ponds, particularly sewage ponds, may be used by waterfowl (Anseriformes) (Dornbush and Anderson 1964, Uhler 1964, Dodge and Low 1972, Maxson 1981, Piest and Sowls 1985). Documentation of wildlife use of wastewater ponds is important in evaluating both benefits as well as negative effects of these created wetlands. Waterfowl (Anseriformes) are highly mobile and can readily occupy created wetlands even if these are isolated from more traditional wetland habitat.

Wastewater ponds on the Idaho National Engineering Laboratory (INEL) in southeastern Idaho were created for sanitary, industrial, and radioactive wastewater disposal. INEL ponds receiving radioactive wastewater have been monitored in terms of wildlife uptake of radionuclides and potential human exposure through harvested wildlife. Radionuclide contamination observed in waterfowl on the INEL (Halford et al. 1982, Halford and Markham 1984, Markham et al. 1988) has been linked to their association with the radionuclide leaching ponds; however, to date, no radiation hazard has been found for waterfowl using INEL ponds (Halford et al. 1982) nor to waterfowl hunters potentially harvesting these ducks (Halford et al. 1981). The potential for waterfowl disease problems from sewage effluent (Moulton et al. 1976, Goldberg and Yuill 1990) as well as potential toxic chemicals (Pounder 1976) in wastewater ponds has also been a concern.

Our objectives in this study were as follows: (1) to assess seasonal use of wastewater ponds on the INEL by waterfowl, (2) to identify potential factors influencing pond use by adult or other fully feathered and fledged waterfowl, and (3) to evaluate brood production and brood use of ponds in relation to sewage versus industrial-radioactive status.

STUDY SITE

The 231,600 ha INEL lies on the western edge of the Snake River Plain in southeastern Idaho (Fig. 1). The topography of the Plain is generally flat to rolling, averaging 1485 m above sea level. The area of the Plain in which the INEL lies is a semiarid cold desert. Temperatures range from -42°C to 39°C. Average annual precipitation is 21.6 cm, 40% of which falls from April through June. Precipitation levels are lowest in July (Clawson et al. 1989). Most vegetation communities found on the INEL are dominated by sagebrush (Artemisia spp.) (McBride et al. 1978).

Wastewater ponds on the INEL contained sanitary waste (9 ponds), industrial waste (4 ponds), radioactive waste (1 pond), or a combination of industrial plus radioactive waste (3 ponds). Two industrial radioactive ponds, 2 industrial ponds, and 2 sewage ponds were located in pairs with only one of each pair containing water at any one time. Thus, at any one time, 14 ponds contained water, including 8 sewage, 3 industrial, 2 industrial-radioactive, and 1 radioactive. Pond surface areas ranged from 0.04 to 2.24 ha. Ponds were 0.3 to 4.5 m deep, but most were 1-2 m deep. Only 3 ponds contained emergent vegetation, and most shorelines were sparsely vegetated. Bottom and shoreline materials were natural (subsoil, soil, and gravel), plastic-lined, or large cobble over plastic lined. A detailed description of the ponds can be found in Cieminski (1993).

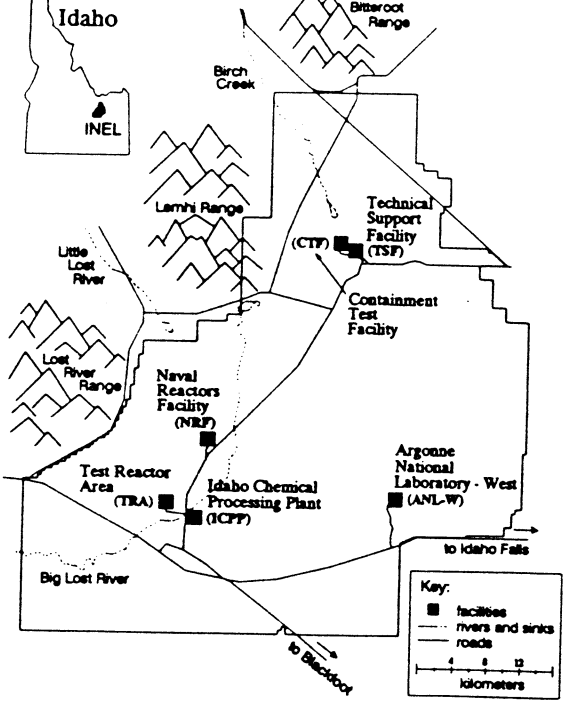

Figure 1. Map of the Idaho National Engineering Laboratory showing location of facilities near which wastewater ponds were located.

MATERIALS AND METHODS

To assess seasonal use by waterfowl, counts were conducted monthly on each pond, August 1989 through July 1991. From March through October, 5 counts were conducted per pond per month. Each of the 5 counts at a pond was conducted at a different time of day, ideally on consecutive days. Counts were not conducted during inclement weather. Counts lasted as long as it took to observe the entire pond and record all waterfowl using the pond. Counts were reduced to 3 per pond per month November through February. Ponds were not surveyed during January and February 1991 because of general lack of use of these ponds when frozen.

Characteristics of ponds or peripheral habitats that we considered potentially important to pond use by waterfowl were recorded. These characteristics included pond surface area (ha), shoreline length (m), percent of pond < 15-cm deep, percent of pond with bare shoreline (no vegetation or vegetation < 15-cm tall within 1 m of land-water interface), percent emergent vegetation, distance to INEL facilities (m) (buildings and

structures associated with energy research sites), microinvertebrate rating (1-5), macroinvertebrate rating (1-5), percent natural shoreline (percent of land-water interface comprised of mud or gravel), and pond relief. Surface area of ponds, shoreline length, and distance to facilities were taken from blueprints of ponds when possible. When blueprints were not available, maps, aerial photos, pacing, and a tape measure were used to estimate these characteristics. The percent of a pond less than 15 cm in depth was estimated from measurements of pond depth taken at 10 evenly spaced transects projecting perpendicularly into the pond; this parameter was estimated visually in radioactive and radioactive-industrial ponds because of restrictions on wading these ponds. Percent emergent vegetation was visually estimated. Ponds were either lined with plastic or cobble or were of natural subsoil or gravel materials; subsoil or gravel bottoms and shorelines at the land-water interface were classified as natural shorelines. Pond relief was the depth which the pond was sunk in the immediate terrain (usually below the surrounding berms) and was estimated using a meter stick. Characteristics that sometimes changed with changing water levels, such as percent bare shoreline, were measured monthly. Microinvertebrate and macroinvertebrate ratings were based on abundance categories developed from data collected in Cieminski and Flake (1995). Rotifers were not included in the microinvertebrate rating because they were too small for waterfowl food items.

Statistical analysis--Since individual waterfowl were not marked, it was not feasible to determine accurately how many of each species were using the ponds. Therefore, "observations" were recorded as an indication of use. An observation is defined herein as 1 individual, seen once. Therefore, a flock of 5 mallards seen once would be 5 observations. However, if a mallard stayed at a pond and was seen during 5 different counts, this would also be 5 observations. Only species with more than 25 total observations per year during regular survey counts were analyzed for seasonal use patterns. For seasonal use analysis, months with 5 counts per pond were made comparable to months with 3 counts per pond by dividing the number of observations by the number of counts conducted that month to get a ratio of number of observations per count.

Nonparametric statistics were used because several of the habitat variables lacked a normal distribution. A Spearman's Rank Correlation was run to evaluate possible correlations between habitat variables (Hoshmand 1988). Possible associations between waterfowl use of ponds and habitat characteristics were explored using logistic regression analysis (LOGISTIC) (SAS Inst. Inc. 1989a). We used a log-linear analysis under Categorical Data Modeling procedures (CATMOD) to evaluate differences in brood use between sewage and industrial-radioactive ponds (SAS Inst. Inc. 1989b).

RESULTS

Seasonal trends and species composition--Twenty four species of waterfowl were observed using wastewater ponds on the INEL; waterfowl tribes represented included Cygnini, Anserini, Anatini, Cairinini, Aythyini, Mergini, and Oxyurini. We combined years in presenting seasonal patterns of abundance because of yearly similarities. Waterfowl use on the INEL was highest in spring from March through June while the fall peak was more was more concentrated in August and September (Fig. 2). Because pond use by Anatini and Cairinini (referred to as dabblers herein) was greater than use by combined Aythyini, Mergini, and Oxyurini (referred to as divers herein), the timing of use peaks is heavily weighted by dabbler migration. Pond use by dabblers peaked about 1 month earlier than divers during spring; dabblers showed a strong August-September peak in pond use while divers had almost no fall peak associated with migration (Fig. 2).

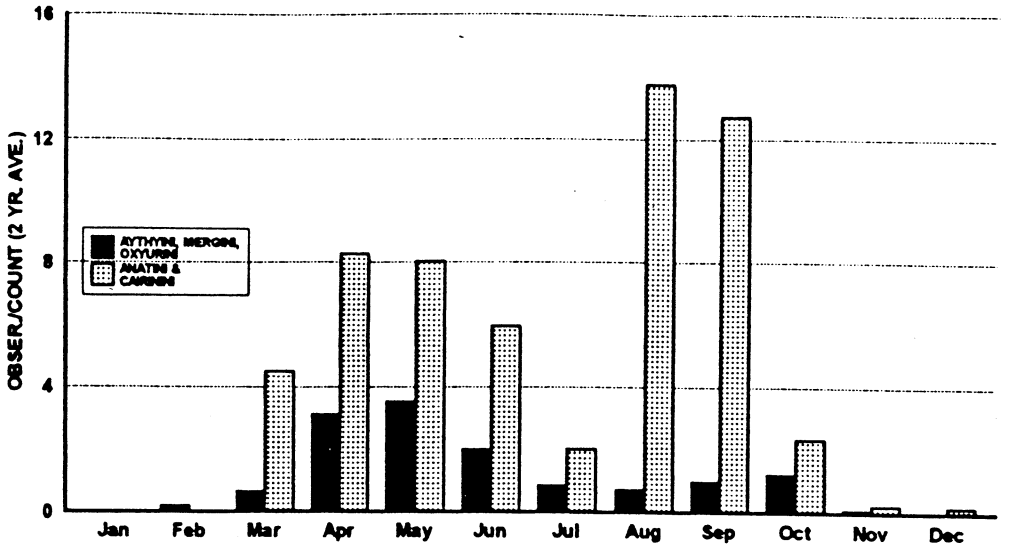

Figure 2. Seasonal use of 17 wastewater ponds on the INEL by divers (Tribes Aythyini, Mergini, and Oxyurini) and dabblers(Anatini and Cairinini) calculated as the 2-year average of the mean number of ducks(observations) seen per visit to a pond(count), August 1989-July 1991.

Seasonal composition of waterfowl in relation to the 8 most common dabbler and 6 most common diver species is illustrated in Fig. 3. Blue-winged teal (Anas discors) and cinnamon teal (A. cyanoptera) were combined (C&B teal) due to the difficulty in separating these species during mid summer to early fall when males were in basic plumage. Cinnamon teal were more abundant than blue-winged teal during April, May, and June, when cinnamon teal made up 61% of the combined cinnamon plus blue-winged teal count (Cieminski 1993). Mallards and green-winged teal were the most abundant dabblers on the INEL while ruddy ducks and redheads were the most abundant divers. Very few species (and individuals) remained on the mostly frozen INEL ponds in November and December--those remaining were primarily mallards, common goldeneyes (Bucephala clangula), and buffleheads (B. albeola).

Of the most common 15 waterfowl species using the INEL, bufflehead, common goldeneye, lesser scaup (Aythya affinis), and ring-necked ducks

(A. collaris) were transients, i.e., the INEL is not within their nesting range or appropriate nesting habitat is absent. However, the ring-necked duck was seen at the INEL during the summer (Fig. 3). The INEL is within the nesting range of gadwall (Anas strepera), cinnamon teal, and blue-winged teal and at least marginal habitat appeared available; however, these three species were considered nonnesting residents because no broods were observed. The remaining 7 species, mallard, green-winged teal, northern shoveler, American wigeon, northern pintail, ruddy duck, and redhead (only 1 brood) nested and reared broods on the INEL during the study and will be referred to as resident species. However, we recognize that many individuals of resident species are actually transient as indicated by the influx of species such as mallards and green-winged teal during migration.

Other waterfowl that occurred on the INEL in small numbers and generally for short periods of time, usually during migration, included tundra swans (Cygnus columbianus), trumpeter swans (C. buccinator), Canada geese (Branta canadensis), wood ducks (Aix sponsa), canvasback (Aythya valisineria) surf scoters (Melanitta perspicillata), Barrow's goldeneye (Bucephala islandica), common merganser (Mergus merganser), red-breasted merganser (M. serrator), and hooded merganser (Lophodytes cucullatus).

Pond characteristics and waterfowl use--We evaluated pond use in relation to habitat characteristics for adult or other fully flighted ducks on the INEL. The species included American wigeon, green-winged teal, mallards, northern pintails, northern shovelers, redheads, lesser scaup, and ruddy ducks. Surface area of ponds was the
primary variable entering logistic regression models predicting the probability of observing a target species on a pond; this variable was positively associated with occurrence of most species (Table 1). Unlike the other target species, ruddy duck occurrence (May-July) was more strongly associated with emergent vegetation than surface area although the latter was also important.

American wigeon and Green-winged teal both showed a positive

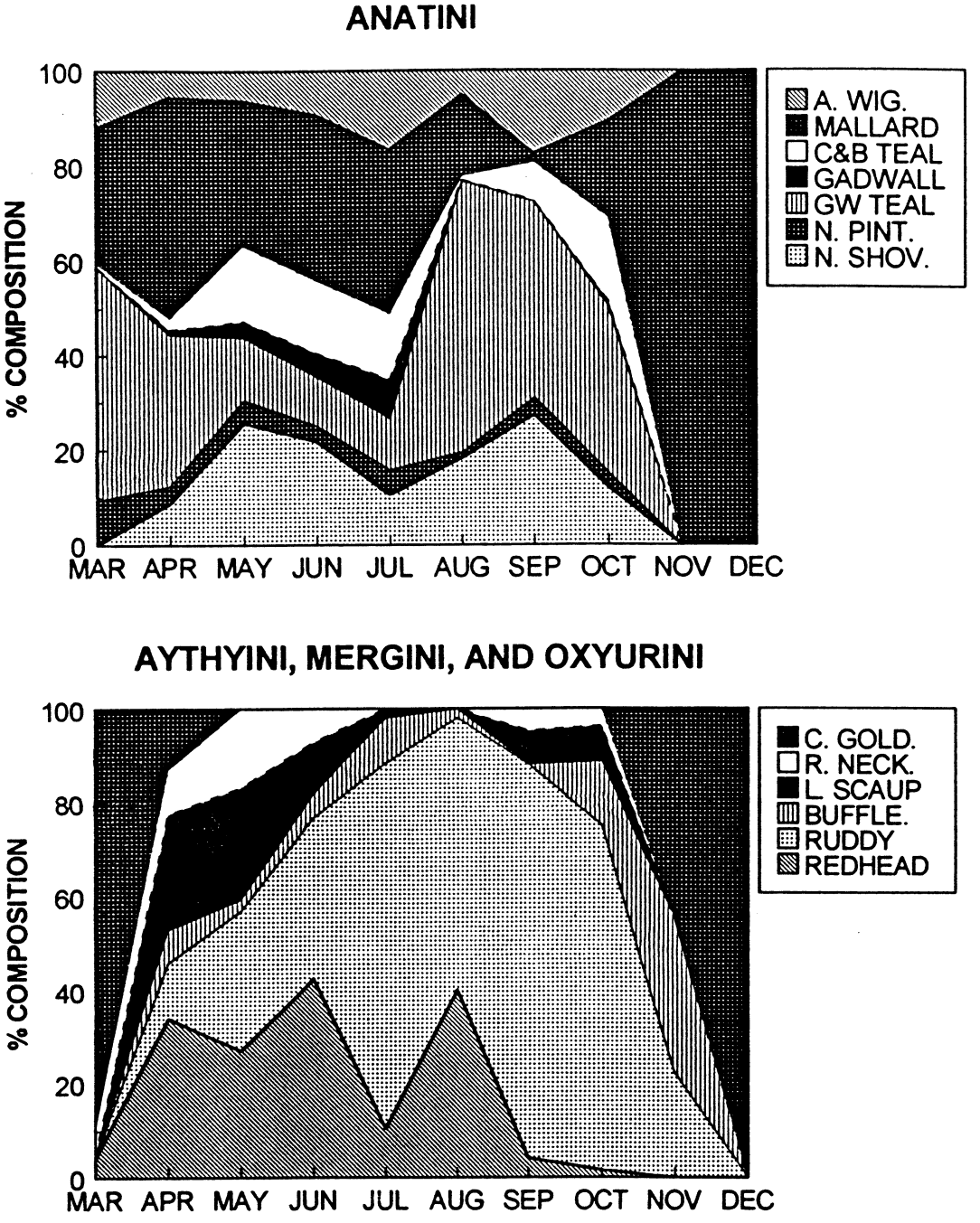

Figure 3. Seasonal (March-December) waterfowl composition on the INEL for the 7 most common dabbler species and the 6 most common diver species.

Table 1. Variables associated (logist.; stepwise P<0.05) with pond use in seven duck species commonly using wastewater ponds on the Idaho National Engineering Laboratory.

Species, Season[a], Group Size (N,Y)[b]	Model[c]	Conc. Pairs (%)[d]	Secondary variables (P=)[e]
Am. wigeon			
Mar.-Apr. (44,10)	= -4.55 + 0.86 (surf. area) + 0.03 (% nat. shore.)	86.6	+ Dist. to facil. (0.01), - Relief (0.02)
May-Jun. (36,18)	= -2.47 + 1.08 (surf. area)	86.4	+ Dist. to facil. (0.01), + macroinv. rating (0.09), - Relief (0.07), + % nat. shore. (0.05)
Aug.-Sep. (35,21)	= -2.44 + 0.63 (surf. area) + 0.02 (% nat. shore.)	81.9	- % pond < 15 cm (0.04), + % bare shore. (0.07)
G.W. teal			
Mar.-Apr. (31,23)	= -2.08 + 0.71 (surf. area) + 0.02 (% nat. shore.)	84.3	None
May-Jun. (21,23)	Not signif.		- % pond < 15 cm (0.03)
Aug.-Sep. (21,35)	= -1.34 + 2.15 (Surf. area)	83.3	- % pond < 15 cm (0.01)
Mallard			
Mar.-Apr. (18,36)	Not signif.		None
May-Jun. (12,42)	Not signif.		None
Aug.-Sep. (34,22)	+ -1.14 + 0.43 (Surf. area)	73.3	None
N. pint.			
Mar.-Apr. (40,15)	-5.60 + 0.79 (Surf. area) + 0.85 (microinv. rating)	84.6	+ % nat. shore. (0.07), + % emerg. veg. (0.02)
May-Jun. (36,18)	= 0.74 - 0.21 (% pond < 15 cm)	80.6	+ microinv. rating (0.06)
Aug.-Sep. (42,14)	= -2.00 = 0.47 (Surf. area)	80.4	- % pond < 15 cm (0.07)
N. Shov.			
Mar.-Apr. (45,9)	= -2.46 + 0.45 (Surf. area)	75.9	+ % Nat. shore. (0.06)
May-Jun. (29,25)	= 0.33 + 0.56 (Surf. area) - 0.15 (% pond < 15 cm)	83.7	- relief (0.06)
Aug.-Sep. (33,23)	+ -3.44 + 2.59 (Surf. area)	95.8	- % pond < 15 cm (0.01)
Redhead			
Apr.-Jun. (57,24)	= 0.33 + 1.10 (Surf. area) - 1.66 (Relief)	92.2	- % pond <15 cm (0.02), + % nat. shore. (0.05), + % emerg. veg. (0.01)

Species, Season[a], Group Size (N,Y)[b]	Model[c]	Conc. Pairs (%)[d]	Secondary variables (P=)[e]
Ruddy duck			
May-Jul. (64,17)	= -8.93 + 0.33 (% emerg. veg.) + 1.09 (Surf. area) + 1.24 (microinv. rating)	92.1	- % pond < 15 cm (0.07), - % bare shore. (0.01), + macroinv. rating (0.10), - relief (0.04), + % nat. shore. (0.01)
Aug.-Sep. (21,35)	= -2.19 + 0.55 (Surf. area)	83.6	- % pond < 15 cm deep (0.01)

[a] Season was broken into two migrant periods (March-April and August-September) and one resident period (May-June) for Anatini (dabblers). Due to low numbers of redheads after June only the April-June period was analyzed. Because of the later arrivals and population peaks in ruddy ducks, May-July and August-October periods were used in the analysis.

[b] Each month ponds were grouped as those with target species and those without. Months were combined for each seasonal period. For example, if green-winged teal were observed in one out of 2 months, March and April for example, on a particular pond, then that pond and its habitat characteristics would be entered once in the not observed (N) category and once in the observed (Y) category for that seasonal period.

[c] The variables in the stepwise logistic regression equation ($P<0.05$) are presented here as our best predictors of presence of the target species on a pond. The equation is of the form logit (p) = intercept $\pm$ coefficients x significant habitat variables. Interpretation of predicted probabilities (p) for evaluating the probability or target species occurring on a particular pond are explained in SAS Institute Inc. (1989a:1076).

[d] Models with higher concordant pair readings had higher predictive capabilities.

[e] These represent other variables selected at $P<.10$ if variables were evaluated one at a time with logistic regression. Some of these variables could be used in developing alternate models.

association with percent natural shoreline and ruddy ducks and northern pintails with increased microinvertebrates. Variables sometimes entering models negatively included percent of pond < 15 cm deep (northern pintails, May-June) and pond relief (redhead). Several other potential variables that could be used to develop a logistic model or that were close to significant are presented in Table 1.

Waterfowl broods--Ducklings were seen May through September (Fig. 4); peak duckling use occurred June through August. Based on brood size, distance between ponds, and date of observation, we estimated the following number of different broods were observed over two years: ruddy duck (n=8), redhead (n=1), mallard (n=19), northern shoveler (n=9), northern pintail (n=8), American wigeon (n=10), and green-winged teal (n=6). Broods sometimes moved between adjacent ponds. We observed mallards moving from open water to upland escape cover in several cases as we approached ponds. For all broods combined, July was the month of peak abundance. Peak observation for ruddy duck broods was in August.

Log-linear analysis indicated that brood observations per visit to ponds were higher on sewage ponds than on industrial-radioactive ponds for mallards ($P<0.01$) and green-winged teal ($P<0.01$). Broods of northern shovelers ($P=0.10$) may have had some association with sewage ponds but it was weak; northern pintail ($P=0.42$) and American wigeon ($P=0.88$) showed no pattern in pond use. The ruddy duck broods were confined to 2 sewage and 1 industrial ponds so we did not analyze their selection on pond type statistically; there was no pattern evident (Fig. 4).

DISCUSSION

Seasonal trends and species composition--Sewage and industrial-radioactive ponds on the INEL attracted a variety of waterfowl species. However, dabblers (tribe Anatini plus some wood ducks) were the species most common in our surveys and the most likely to remain on INEL ponds

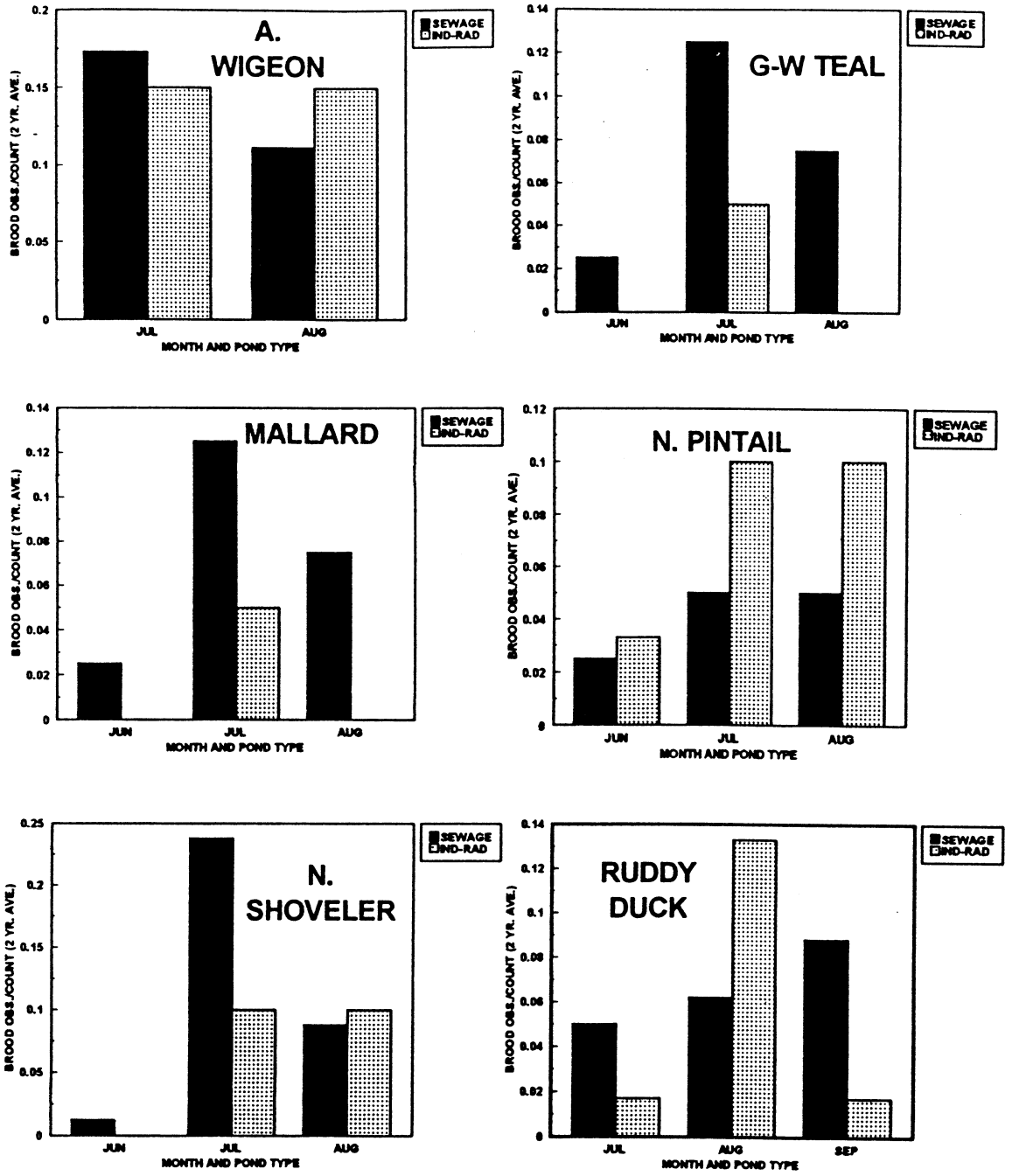

Figure 4. Occurrence of duck broods by month on sewage and industrial-radioactive ponds on the INEL for the combined period from August 1989-July 1991.

during the reproductive season. Belanger and Couture (1988), Maxson (1981), and Piest and Sowls (1985) also noted a predominance of dabbling duck species using sewage ponds. Waterfowl species commonly using wastewater ponds at the INEL were almost identical to those reported by Piest and Sowls (1985) in Arizona, and Maxson (1981) in North Dakota.

Most waterfowl species stopped only during migration but others remained on the INEL during the reproductive season. Waterfowl on the INEL reached peak numbers in April-May and August-September. At the Hanford Site in Washington, also a Department of Energy research site, waterfowl were in greatest abundance from February through May and October through November (Fitzner and Price 1973). Unlike the INEL, most ponds at the Hanford Site did not freeze from mid November through mid February. Arrival dates at ponds were about a month earlier than those at Grays Lake National Wildlife Refuge (Steel and Bizeau 1956) which is about 130 km southeast of the INEL and about 500 m higher in altitude. Dates of arrivals, peaks, and departures given for Rupert, Idaho, 135 km south-southwest of the INEL, were generally earlier than those we observed (Davis 1935).

Pond Characteristics And waterfowl use--The positive association of pond size with target species occurrence generally applied to each of the seasonal periods. Lokemoen (1973) and Flake et al. (1977) found that pond size was the primary factor associated with increased numbers of dabbling ducks on stock watering ponds in the northern Great Plains. The association of ruddy ducks with ponds in May-July was better predicted by increased amounts of emergent vegetation than increased surface area. This association was related to the need for appropriate emergent cover for building their overwater nests and was primarily due to use of a single wetland with abundant emergent vegetation of cattail (Typha spp.) and hard stemmed bulrush (Scirpus acutas). After the nesting season (August-September), occurrence of ruddy ducks was best predicted by surface area, not emergents.

Natural shorelines offer a substrate of soil, subsoil, or gravel in

comparison to ponds with cobble or plastic shorelines; the positive associations of green-winged teal and American wigeon (both in March-April) with natural shorelines may indicate that artificial shorelines of cobble or plastic reduced use by these two species. Percent natural shoreline was also one of the "other" important variables listed for northern pintails, northern shovelers, redheads, and ruddy ducks.

We anticipated that larger concentrations of microinvertebrates, as measured by our monthly surveys, might be related to observation of northern shovelers; our analysis did not detect this relationship. When monthly counts of Cladocera were included in the logistic regression analysis for northern shovelers, we still were unable to detect a relationship. Our monthly samples of invertebrates and our subsequent ratings of microinvertebrate abundance on a 1-5 scale may not have adequately reflected the daily fluctuations in invertebrates that could influence northern shoveler use of ponds. Microinvertebrates were positively associated with northern pintail as a secondary variable in the March-April model. Swanson (1977) observed heavy use of midge fly adults and pupae (Chironomidae) and Cladocera in gadwalls, blue-winged teal, northern shovelers, and mallards on sewage lagoons. Cladocera, especially Daphnia spp., were unusually abundant in most INEL sewage ponds while Chironomidae were more equally distributed between sewage and industrial-radioactive ponds (Cieminski and Flake 1995).

Macroinvertebrate abundance was potentially associated with ruddy duck occurrence on INEL ponds but as a secondary variable at P=0.10. Ruddy ducks strongly select insect matter over crustaceans during the breeding season (Krapu and Reinecke 1992), possibly explaining the inclusion of macroinvertebrates as a secondary variable . We had no information on ruddy duck food habits on the INEL and we cannot explain the inclusion of microinvertebrate rating in the model.

The lack of significant models for mallards in March-April and May-June seems reasonable considering the adaptability of mallards to a wide variety of habitats on both a landscape and local level (Bellrose 1976). Mallards

are the generalists of the Anatidae. The adaptation of mallards to a wide variety of water bodies including streams, marshes, constructed ponds, lakes, and urban water bodies is evident in the literature.

Northern pintails are perhaps the best adapted duck in the northern hemisphere for using shallow prairie (Hochbaum and Bossenmaier 1972) or tundra wetlands. However, extremely shallow ponds in our study area may have exposed birds to a greater threat from mammalian predators; the concentrations of chemical waste in the bottom sediments is also unknown and may influence use when some industrial waste ponds are at extremely low water levels. The height from water to the berm or surrounding upland was entered into the model for redheads as a negative association. This association may be due to the difficulty redheads may have taking flight on small bodies of water that are set too deeply into the ground. Their low trajectory during takeoff would likely cause some selection for ponds where they have the space to clear the surrounding berm or bank. Relief was also negatively identified as a secondary variable for the ruddy duck and northern shoveler; trajectory problems when taking flight could explain the negative association for ruddy ducks but not northern shovelers.

Of the target species, American wigeon flushed most quickly on our approach to ponds. Distance to facilities could be used to develop a model for American wigeon during March-April and May-June if surface area were removed from the equation. This species may be influenced by the presence of facilities, humans, and vehicles near ponds on the INEL; distance to facilities did not appear to influence pond use by any of the other species.

Waterfowl broods--Dornbush and Anderson (1964) observed much higher brood densities on stabilization ponds for wastewater treatment than on natural wetlands in South Dakota; these treatment ponds were associated with small municipalities. In Arizona, Piest and Sowls (1985) observed unusually high levels of waterfowl brood production on a 19.0-ha marsh used for disposal of secondarily treated sewage effluent. Belanger and

Couture (1988) observed exceptionally high use of sewage ponds by dabbling duck broods in Quebec. Similar observations of high brood use of sewage treatment lagoons were made by Dodge and Low (1972) in Utah, and Maxson (1981) in North Dakota.

Cladocera, primarily Daphnia magna, and midges (Chironomidae) dominated the diets of juvenile and adult dabbling ducks on waste-stabilization ponds in the northern plains (Swanson 1977). Cieminski and Flake (1995) observed unusually high levels of Daphnia spp. in sewage ponds in relation to industrial and radioactive ponds on the INEL. In our study, mallard and green-winged teal selected sewage ponds over industrial-radioactive ponds; however, northern shovelers showed only a weak and non significant (P=0.09) relationship to these pond categories despite the importance of Cladocera to this species. Cladocera are important to the diet of northern shovelers ducklings (Sugden 1973, Krapu and Reinecke 1992) but we could detect no selection for sewage effluent ponds. American wigeon ducklings are much less dependent on invertebrates and more dependent on plant matter than most other dabbling duck species (Sugden 1973) and showed no pattern in relation to sewage versus industrial-radioactive pond categories.

CONCLUSIONS

Twenty-four species of waterfowl were observed on INEL ponds during a 2-year study. Seven species remained on the INEL ponds throughout the reproductive period and reared broods. The most common waterfowl species were mallards, green-winged teal, northern shoveler, American wigeon, northern pintail, cinnamon teal plus blue-winged teal (combined), ruddy ducks, and redheads; all but the cinnamon and blue-winged teal were observed with ducklings. Pond surface area was the best predictor of waterfowl use for most of the more common species. Several other factors were also associated with waterfowl use of the ponds including some positive associations with the percentage of ponds with a natural mud or gravel shoreline (compared to plastic or cobblestone),

microinvertebrate rating, and percent cover of emergent vegetation; percentage of ponds < 15 cm deep (due to avoidance of nearly dewatered ponds) and relief (depth of pond surface below the surrounding landscape or berm) were negatively associated with pond use. Mallard and green-winged teal broods appeared to selectively use sewage ponds in comparison to industrial-radioactive ponds.

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Reptile and Amphibian Characterization of a Thermally Disturbed Braided StreamUndergoing Restoration Near Aiken, SC

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ABSTRACT

Pen Branch is a third order tributary of the Savannah River located in the upper Atlantic Coastal Plain of west-central South Carolina. From 1954 through 1988 the stream received thermal effluent which greatly increased flow and raised water temperatures in the study area to above 50°C (122°F). This resulted in the elimination of nearly all plant and animal life in the stream corridor. Restoration efforts began in the winter of 1992-93, and consist of plantings of cypress and indigenous bottomland hardwood species suitable to the soil conditions and hydrologic regime; additionally, early in the restoration process, experimental portions of the stream corridor were sprayed with herbicide and/or burned to facilitate access by the planting crews and reduce competition from pioneering vegetation. A herpetofaunal survey was initiated January 1, 1995 in order to evaluate the current status of reptile and amphibian populations utilizing the corridor and to establish baseline data to evaluate changes in the herpetofaunal community as reforestation progresses. Data gathered during the first year

of the project, in which approximately 10,000 reptiles and amphibians representing 67 species were captured, indicates that many reptile and amphibian species have already become established in the disturbed wetland. Although rich and diverse, the reptile and amphibian population at the site does not appear to be typical of a closed-canopy, bottomland forested wetland, and may be an indicator of the early transitional phase of the restoration project.

INTRODUCTION

Although wetlands are created and restored for various reasons such as recreation, aesthetics, flood control, aquaculture, etc., the establishment or re-establishment of a fully functional wetland ecosystem may well be the most challenging goal. Evaluating the ecological functionality of a wetland is often difficult; and the ecology of the wetland may take many years to develop and stabilize. However, ecological evaluation is often necessary in order to compare various restoration strategies as well as to satisfy legal requirements or public sentiment.

The most obvious way to ascertain the functional success of a wetland creation or restoration project is to compare the associated plant and animal communities of the creation site with those of natural, undisturbed wetlands. However, Zentner (1992) cautioned that the authenticity of a restored landscape is not its immediate resemblance to a "model landscape" but its resemblance to the model system at a comparable stage of development and its potential to approximate that system. Since it can take decades for a created or restored wetland to approximate a natural condition, periodic examination of the plant and animal communities is a useful means of monitoring the progress of the ecosystem's development. Since amphibians and some reptiles are very dependent on wetlands (often a *particular* wetland) and are generally less transient and exhibit smaller home ranges than other organisms such as mammals or birds which may utilize several wetlands in an area, studies of their populations in association with the project wetland are especially useful. As one aspect

of the monitoring and evaluation phase of the restoration of Pen Branch, we attempt to characterize the associated reptile and amphibian populations in a standard and duplicable manner.

STUDY SITE

Pen Branch is a third order tributary of the Savannah River in the southeastern United States. It is located entirely within the boundaries of the Savannah River Site, a Department of Energy industrial and former nuclear production facility which lies along the western border of South Carolina, approximately 90 miles from the coast. Prior to 1954, the Pen Branch corridor was characterized by closed-canopy bottomland hardwood forest.

Beginning in 1954, water was pumped from the Savannah River to cool a nearby nuclear reactor. The resulting thermal effluent was discharged into the headwaters of Pen Branch. As a result, flow rates increased by 1-2 orders of magnitude (Dulohery et al. 1995) and water temperatures in the lower reaches of the stream exceeded 50°C. From 1954 to 1989 the hot water and increased flow killed the existing vegetation in the stream corridor and in a delta-shaped area of the Savannah River riverswamp into which Pen Branch empties. By the mid-1980's, severe canopy defoliation reached a maximum area of about 152 ha in the delta and 113 ha in the corridor (Wike et al. 1994). Additionally, the increased flow altered the topography of the stream-bed and floodplain creating a braided flow pattern.

Presently, the stream enters the top of the study area as a typical single-channel stream. About 100m downstream, the channel divides into several smaller streams which continue to divide and converge for approximately 2.5km until the stream corridor opens into the riverswamp. The stream floodplain in the upper portion of the study area is about 100m wide and generally has 2-3 stream channels and 2-3 wet sloughs which only receive flow during higher water stages. The inter-channel areas in this section are

generally 1-2 feet above the water surface and are fairly well drained. The floodplain in the lower portion of the study site exceeds 300m in width at points and may have 2-4 main stream channels and as many as 6-10 smaller ones. The inter-channel areas here are just slightly above the water surface and are poorly drained and often boggy.

When thermal discharges ceased in 1989, the corridor and delta were rapidly colonized by early successional vegetation. By the early 1990's, the disturbed areas were heavily vegetated with woody species including black willow (*Salix niger*) as well as the less abundant wax myrtle (Myrica cerifera), alder (Alnus spp.), and red maple (Acer rubrum). Herbaceous vegetation present included rushes, such as *Juncus* sp. and *Scirpus* sp., and cattails (*Typha* sp.). *Myriophyllum aquaticum* and *Hydrocotyle* sp. were prominent in the stream. The vegetation of the upland areas adjacent to the stream corridor consisted of a thin boundary (20-100m in width) of mature hardwood forest, predominately oak and hickory, beyond which is planted in *Pinus* sp. in various stages of maturity. Natural regeneration of typical bottomland hardwood trees was determined to be insufficient and the DOE initiated a reforestation plan in 1991 (Dulohery et al. 1995). As part of this plan, the corridor and delta were divided into seventeen sections, eight of which remained as non-treated control zones. The other nine zones were planted after either no preparation, herbicide application, or herbicide application followed by prescribed burning. Although herbicide application and burning were done primarily to allow access by the planting crews and to reduce competition from existing vegetation, the design allows for the comparison of several reforestation approaches. A detailed description of the restoration plan can be found in Dulohery et al. (1995).

MATERIALS AND METHODS

In this study the stream corridor and the immediately adjacent upland area were intensively sampled. Since the target group of animals is composed of a large number of species with varied microhabitats, life histories, and

morphologies, a variety of sampling methods were used to ensure a thorough sampling. The herpetological sampling methods employed were:

1) Linear drift fences (Gibbons and Semlitsch 1981), thirty meters long, constructed of aluminum flashing with a pair of five gallon metal bucket pitfall traps at each end and 10m from each end.

2) Arrays of metal (0.61m x 1.22m x 16 gauge surplus stainless steel) and wood (0.61m x 1.22m x 1.91cm plywood) cover-boards.

3) Hoop net turtle traps (0.76m diameter, 3.8cm mesh, approximately 1.8m long).

4) Funnel wire-mesh eel traps with the entrance holes enlarged to approximately 5cm.

5) Hand collecting, visual and audible observations.

Ten trails were established across the corridor at fairly regular intervals during the fall of 1994. The spacing and path of the trails was governed by the need to avoid existing experimental plots established by Savannah River Ecology Laboratory (SREL) and United States Department of Agriculture Forest Service. Eight arrays of cover-boards, each consisting of two metal and two wood boards, were placed along each trail. One array was placed in undisturbed uplands at both ends of each trail; one array was placed at the upland-wetland interface near the ends of each trail; and the remaining four arrays were placed in the corridor at approximately even intervals along the trails. Drift fences were constructed in the uplands area near the east end of five of the trails and near the west end of the other five. The drift fences and cover-boards were checked daily for reptiles and amphibians. Any reptiles or amphibians o bserved while walking the trails were noted and captured, if possible. Additionally, aquatic trapping was utilized for one week of each month. Two turtle traps and three eel traps

were placed in suitable locations along each trail and baited with sardines packed in soy oil. The placement of the sampling units corresponded with the reforestation planting and control areas so that any differences in the herpetofauna between the planted areas and the non-planted controls could be detected.

Captured amphibians and lizards (except for *Ophisaurus* sp.) were marked with the same generic toe-clip (Ferner 1979) so that recaptured animals could be identified. The species, sex and age class (if possible), capture date and location were recorded for each animal prior to released. Snakes, turtles, and alligators were retained for sex and age determination, body measurements, and marking with a unique identification code. Snakes were marked by clipping ventral scales, turtles were marked with notches cut into the marginal scutes (Gibbons 1988), and alligators were marked by cutting the keels of dorsal scutes. The animals were released as soon as practical at the point of capture. All non-target animals were noted and immediately released at the point of capture.

RESULTS

Approximately 10,500 reptiles and amphibians, representing 69 different species, were captured. This number includes about 1160 captures of animals that had been previously marked. The number of individuals captured of each species, not counting recaptures, is shown in Table 1.

Table 1. Reptiles and Amphibians Collected at Pen Branch in 1995

Species		Number Collected
SNAKES		
Agkistrodon piscivorus	Eastern Cottonmouth	23
Carphophis amoenus	Eastern Worm Snake	11
Cemophora coccinea	Scarlet Snake	15
Coluber constrictor	Black Racer	21
Crotalus horridus	Timber Rattlesnake	2
Diadophis punctatus	Southern Ringneck	16
Elaphe guttata	Corn Snake	2
Elaphe obsoleta	Rat Snake	7
Farancia abacura	Mud Snake	7
Farancia erytrogramma	Rainbow Snake	23
Heterodon platirhinos	Eastern Hognose Snake	1
Lampropeltis getula	Eastern Kingsnake	7
Lampropeltis triangulum elap.	Scarlet Kingsnake	1
Nerodia erythrogaster	Redbelly Water Snake	15
Nerodia fasciata	Banded Water Snake	61
Nerodia taxispilota	Brown Water Snake	19
Opheodrys aestivus	Rough Green Snake	7
Storeria dekayi	Northern Brown Snake	37
Storeria occipitomaculata	Redbelly Snake	57
Tantilla coronata	SE Crowned Snake	1
Thamnophis sauritus	Eastern Ribbon Snake	10
Thamnophis sirtalus	Eastern Garter Snake	9
Virginia striatula	Rough Earth Snake	1

Table 1. Continued.

Species		Number Collected
LIZARDS		
Anolis carolinensis	Green Anole	162
Cnemidophorus sexlineatus	Six-lined Racerunner	8
Eumeces fasciatus	Five-lined Skink	85
Eumeces inexpectatus	SE Five-lined Skink	9
Eumeces laticeps	Broad-headed Skink	106
Eumeces sp.	species not determined	12
Ophisaurus ventralis	Eastern Glass Lizard	2
Sceloporus undulatus	Eastern Fence Lizard	31
Scincella lateralis	Ground Skink	69
TURTLES		
Chelydra serpentina	Snapping Turtle	81
Clemmys guttata	Spotted Turtle	1
Kinosternon baurii	Striped Mud Turtle	15
Kinosternon subrubrum	Eastern Mud Turtle	21
Pseudemys concinna	Eastern River Cooter	4
Pseudemys floridana	Florida Cooter	8
Sternotherus odoratus	Common Musk Turtle	13
Terrapene carolina	Eastern Box Turtle	6
Trachemys scripta	Yellowbelly Slider	215

Table 1. Continued.

Species		Number Collected
CROCODILIANS		
Alligator mississippiensis	Alligator	2
FROGS		
Acris gryllus	Southern Cricket Frog	24*
Bufo terrestris	Southern Toad	1556
Gastrophryne carolinensis	Eastern Narrowmouth Frog	3259
Hyla avivoca	Bird-voiced Treefrog	1
Hyla chry./versi.	Gray Treefrog	1
Hyla cinerea	Green Treefrog	8*
Hyla femoralis	Pine Woods Treefrog	1
Hyla squirella	Squirrel Treefrog	1
Pseudacris crucifer	Spring Peeper	3*
Pseudacris nigrita	Southern Chorus Frog	4
Pseudacris ocularis	Little Grass Frog	1
Pseudacris ornata	Ornate Chorus Frog	4
Rana catesbeiana	Bullfrog	34
Rana clamitans	Green Frog	142
Rana utricularia	Southern Leopard Frog	1395*
Scaphiopus holbrooki	Eastern Spadefoot	27
SALAMANDERS		
Ambystoma maculatum	Spotted Salamander	25
Ambystoma opacum	Marbled Salamander	503
Ambystoma talpoideum	Mole Salamander	47
Amphiuma means	Two-toed Amphiuma	1
Eurycea cirrigera	Southern Two-lined Sal.	3
Eurycea longicauda	Three-lined Salamander	28

Table 1. Continued.

Species		Number Collected
Notophthalmus viridescens	Red-spotted Newt	2
Plethodon glutinosus	Slimy Salamander	533
Pseudotriton montanus	Mud Salamander	1
Siren intermedia	Lesser Siren	49
Siren lacertina	Greater Siren	2

* indicates that many more individuals were observed.

DISCUSSION

Many of the reptile and amphibian species collected are not dependent on the wetland and were probably largely unaffected by its disturbance. Some species, such as *B. terrestris*, *A. gryllus*, *G. carolinensis*, and *R. utricularia* are known to use Pen Branch as a reproduction area, at least to some extent, but also utilize other nearby habitats and apparently do not rely entirely on Pen Branch. Most of the turtles, water snakes, and aquatic salamanders rely very heavily on the habitat and probably few members of those species survived in the immediate area. The riverswamp into which Pen Branch empties, nearby Carolina bays (Schalles et al. 1989), and other nearby wetlands likely served as refuges from which emigrating reptiles and amphibians began to recolonize the Pen Branch corridor as it became suitable.

The abundance of reptiles and amphibians and the diversity of species at Pen Branch is testimony to their ability to rapidly utilize and colonize a newly available habitat. The ability to reproduce at a young age, the production of large numbers of offspring and dispersal behavior of offspring contribute to this ability, especially in the case of amphibians. Although the habitat now supports a wide variety of species and a large number of individuals, the assemblage does not appear to be typical of bottomland hardwood forested wetlands. Few studies have been conducted in the region to characterize the herpetofauna of these forested wetlands, so comparisons are difficult. However, Wharton et al. (1981) provided a list of species known to occur within such wetlands. The following species would be likely inhabitants of bottomland forested wetlands in the area of Pen Branch:

Siren intermedia
Amphiuma means
Desmognathus auriculatus
Eurycea quadridigitata
Pseudotriton montanus
Ambystoma maculatum
Ambystoma opacum
Ambystoma talpoideum
Rana clamitans
Rana utricularia
Acris gryllus
Pseudacris nigrita
Hyla avivoca
Hyla versicolor/chrysoscelis
Kinosternon subrubrum
Kinosternon baurii
Trachemys scripta
Terrapene carolina
Rigina rigida
Farancia abacura
Nerodia erythrogaster
Nerodia taxispilota
Agkistrodon piscivorus
Elaphe obsoleta
Coluber constrictor
Lampropeltis getula
Thamnophis sauritus
Anolis carolinensis

At least a few individuals of nearly all of these species have been captured or observed. The exceptions are *D. auriculatus*, *E. quadridigitata*, and *Rigina rigida*. *D. auriculatus* is found elsewhere on the Savannah River Site although it is not abundant (Gibbons and Semlitsch 1991). It may truly be locally absent at Pen Branch or the sampling to date may have been inadequate to disclose its presence. *E. quadridigitata* reportedly prefers temporary pond and Carolina Bay type habitats(Gibbons and Semlitsch 1991). *R. rigida* is very uncommon on the SRS and has only been captured on a few occasions(Gibbons and Semlitsch, 1991). It may exhibit a preference for still water habitats although continued monitoring may reveal its presence. Some species are present at Pen Branch in greater numbers than would generally be expected in an undisturbed forested bottomland/swamp (e.g. *C. constrictor*, *O. aestivus*, *H. cinerea*, and *A. carolinensis*). These species generally are found in sunny, open canopy areas with abundant ground vegetation and/or low-growing trees. These habitats presently abound in the Pen Branch corridor. It is likely that as planted hardwoods grow towards maturity over the next few decades Pen Branch will become less suitable for these animals and their populations

will diminish. Likewise, it is anticipated that the populations of more typical forested wetland species will approach expected levels.

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EFFECTS OF *LUDWIGIA* AND *TYPHA* ON TREE GROWTH ON PHOSPHATE MINED LAND IN FLORIDA

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ABSTRACT

Florida regulations require that "nuisance" species, including the native primrose willow (*Ludwigia peruviana*) and cattail (*Typha spp.*), be limited to less than ten percent of the cover on reclaimed wetlands. Controlling these two species is expensive and may be unnecessary. It is hypothesized that shade-tolerant trees can grow through, overtop, and shade out these sun-requiring "nuisance" species. This study, conducted on reclaimed phosphate mined lands in central Florida, compared tree growth in primose willow stands (P1 and P2 sites) and in cattail stands (C1 and C2 sites) versus in stands from which the nuisance species had been removed (cut or herbicided). Through three growing seasons at the P1 site, primrose willow had little or no effect on baldcypress (*Taxodium distichum*) height growth, but by the sixth growing season, baldcypress was 25 percent taller with primrose willow removed. Nevertheless, baldcypress growth was substantial in the presence of primrose willow. After three growing seasons 50 percent of the baldcypress trees had grown through and overtopped the primrose willow, and that figure had increased to 85 percent after the sixth growing season. Popash (*Fraxinus caroliniana*), baldcypress, and red maple (*Acer rubrum*) heights were only slightly affected by primrose willow competition at the P2 site after three years, although some trees were bent over by the primrose willow. Cattail had little or no effect on heights of baldcypress or popash after two years. The modest effect of primrose willow on baldcypress indicates control measures, which are expensive, may not be necessary. If trees are planted at a sufficient density, they will eventually develop a canopy cover that will

probably shade out the primrose willow.

INTRODUCTION

Primrose willow (*Ludwigia peruviana*) and cattail (*Typha spp.*) are two wetland species native to Florida (Ramamoorthy and Zardini 1987; Dressler et al. 1987). However, due to their invasiveness, the Florida Department of Environmental Protection-Bureau of Mine Reclamation (FDEP-BMR) lists them as "nuisance" species. This designation is legally important in the reclamation of forested wetlands on phosphate mined lands, because one success (release) criterion on many FDEP-BMR permits limits "nuisance" species to less than 10% of the total cover (Robbin Trindell, FDEP-BMR, personal communication).

In order to comply with this criterion, primrose willow and cattail must be controlled with herbicides, mowers, or by labor intensive means. Yet, is this control necessary ecologically or just aesthetically? Several observations (Hawkins and Ruesch 1988; Clewell 1981; Clewell and Raymond 1991; Richardson et al. 1994) suggest that tree species can grow through, overtop, and shade out the "nuisance" species.

If this hypothesis is true, then battling nature with herbicides, machines, and labor is a waste of money. Whereas, a reclamationist could save money working with nature by planting a sufficient number of trees and by being patient.

The literature regarding the competitive effects of primrose willow and cattails on tree growth on phosphate mined lands is limited. Clewell and Raymond (1991) stated that primrose willow's canopy might benefit tree seedlings by insulating them from direct sunlight and by maintaining constant and moderate root zone soil temperatures. As for cattails, Rushton (1988) detected no differences in the heights (after one year) of baldcypress (*Taxodium distichum*) seedlings grown on clay settling ponds whether the sites were cleared or not at the time of planting.

Clewell (1981) and Erwin et al. (1985) focused on assessing success of wetland reclamation techniques, but made important observations. Erwin et al. (1985) observed profuse cattail that appeared to restrict baldcypress and Florida maple (*Acer saccharum floridanum*) seedling growth and/or survival. Yet, seedlings that survived this competition appeared to have high growth rates. While Clewell (1981) observed green ash (*Fraxinus pennyslvanica*) vigorously growing in a dense cattail stand, indicating that densely planted ash might grow, overtop, and replace cattails.

The objective of this study was to determine to what extent primrose willow and cattail may be detrimental to baldcypress, popash, and red maple (*Acer rubrum*) growth. The longer term objective is to determine if these tree species can overtop primrose willow and cattail.

METHODS

To evaluate the effects of primrose willow and cattail competition on tree growth, two stands of primrose willow (P1 and P2 Sites) and two stands of cattail (C1 and C2) were selected on reclaimed wetlands.

Primrose willow: At each primrose willow site, the stand was cut to the ground with a tractor drawn mower and eight replicate blocks of paired plots were located within similar hydric regimes. Within each block, a plot was randomly selected to be the weeded plot and the other plot to be the untreated plot. The primrose willow in the weeded plots was repeatedly mowed or herbicided, and the primrose willow in the untreated plots was allowed to regrow.

P1 Site: This primrose willow stand was in a reclaimed, seepage wetland at the IMC-Agrico Clear Springs Mine (11 km southeast of Bartow). On 12 May 1989, five baldcypress (*Taxodium distichum*) trees, grown in 15 cm diameter pots, were planted 2 m apart in each 6 m x 10 m plot. The primrose willow in the weeded plots were mowed three to four times during each growing season with a motorized trimmer until the end of

March 1993 and then herbicided two times with triclopyr (Garlon^R 3A) until early August 1993.

Baldcypress heights were measured at the time of planting and each subsequent winter. Primrose willow height was measured next to each tree after the sixth growing season. Diameter at Breast Height (DBH) at 1.4 m and crown diameters (average of two perpendicular measurements) were measured at the end of the fourth and sixth growing seasons.

P2 Site: This stand of primrose willow was in a reclaimed wetland along McCullough Creek at the Mobil Fort Meade Mine (3 km north of Fort Meade). On 10 November 1992, five trees each of baldcypress, popash (*Fraxinus caroliniana*), and red maple (*Acer rubrum*), grown in 35 cm x 10 cm sacks, were planted 2 m apart in each 8 m x 12 m plot. The primrose willow in the weeded plots were herbicided three to four times each growing season with triclopyr (Garlon^R 3A). During these sprayings, the trees were protected with plastic bags.

Tree mortality was assessed in June and October of 1993. Dead trees were replaced after each assessment to assure sufficient trees for the growth comparisons. Tree heights and basal diameters (at 15 cm height) were measured at the time of planting and each subsequent winter. Because some trees were bent by the primrose willow, height was measured vertically from the ground to the highest growing point, which may not have been the apical bud on bent trees.

Cattail: At each cattail site, eight replicate blocks of paired (weeded and untreated) 6 m x 12 m plots were located within similar hydric regimes and cattail densities. Weeded plots were initially macheted in early May 1994 and were maintained three to four times each growing season by cutting resprouts below the water surface with pruning shears. Cattails in the untreated plots were not cut, but allowed to continue growing. In each plot, five baldcypress and five popash trees, grown in 35 cm x 10 cm plastic sacks, were planted at the end of May 1994.

Trees heights, stem diameters (at 50 cm height, since the water was 20 to 35 cm deep), and crown diameters were measured in December 1994 and 1995. Tree heights were measured vertically from the ground to the highest growing point.

C1 Site: This stand of cattail was located at the Cargill Hooker's Prairie Mine (14 km southwest of Bartow) in "Cell B", an experimental wetland containing a layer of phosphatic clay on top of overburden. In each plot, the five baldcypress and five popash trees were planted 2 m apart.

C2 Site: This stand of cattail was in a reclaimed wetland at the Mobil Fort Meade Mine 2 km north of Fort Meade and two miles east of U.S. Highway 17. The site had been regraded in 1991 and planted with a variety of tubeling trees between September 1991 and February 1992. The previously planted trees (mostly baldcypress) were located, mapped, and marked with tags and flagging. In each plot, the new trees were planted among the existing trees no closer than 1.5 m apart.

Statistical Methods: All parameters in both the primrose willow and cattail experiments were statistically analyzed by the SPSS (1988) package using the non-parametric, Mann-Whitney comparison (Mann and Whitney, 1947). This comparison was utilized to take into account the potential of great variation in tree height.

RESULTS

P1 Site: There was very little difference in baldcypress height between the two treatments in the first three years (Figure 1). Tree height was actually slightly greater ($P<.05$) in the presence of primrose willow the first year, but growth was enhanced by primrose willow removal in subsequent years. By the sixth year, the compounding of yearly growth enhancement had led to a 25% greater height with primrose willow removal ($P<.01$). Nevertheless, baldcypress height growth was substantial in the presence of primrose willow. The trees in the untreated plots were 5 m tall after the

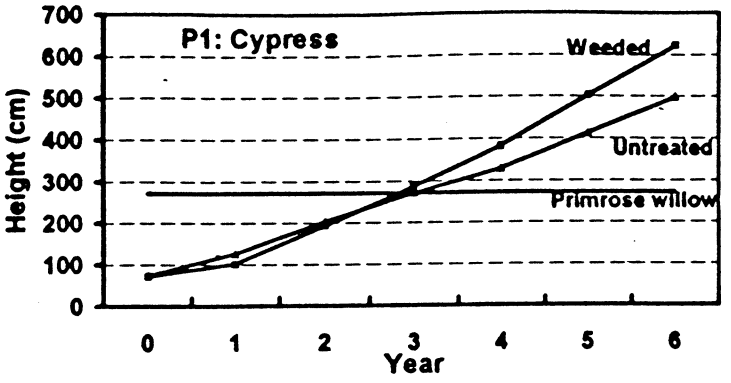

Figure 1. Average yearly height (cm) of baldcypress in weeded and untreated primrose willow plots at the P1 site.

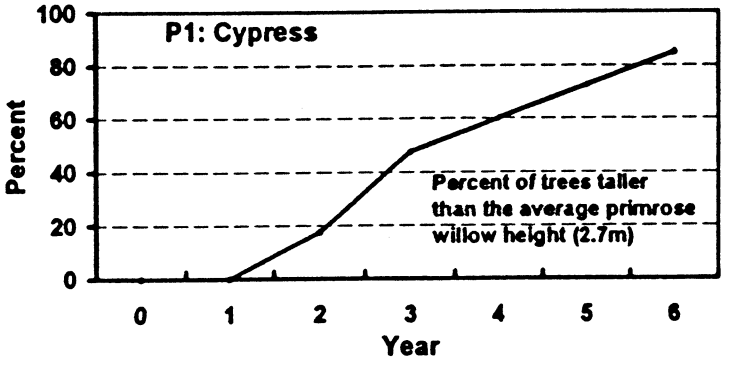

Figure 2. Yearly percent of baldcypress in the untreated primrose willow plots taller than 2.7m (average height of primrose willow) at the P1 site.

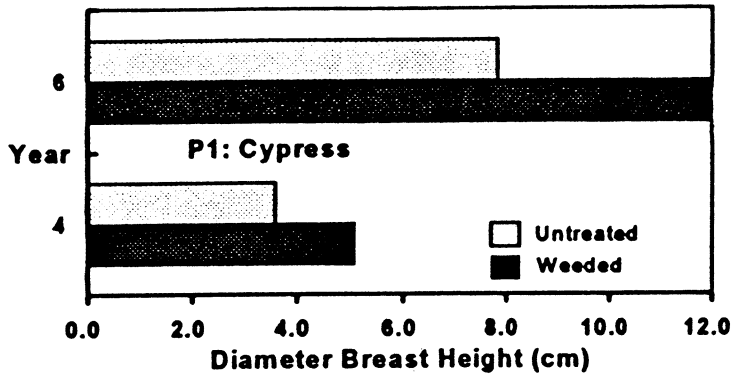

Figure 3. Average diameter at breast height (cm) of baldcypress in weeded and untreated primrose willow plots at the P1 site.

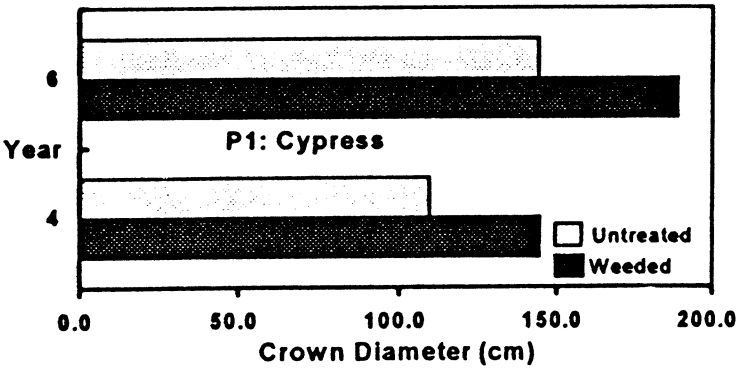

Figure 4. Crown diameter (cm) of baldcypress at the P1 site.

sixth year, over 2m taller than the primrose willow canopy. Baldcypress trees in the untreated plots first began to surpass the average primrose willow height of 2.7 m (Figure 2) by the end of the second year, when 18% of the trees were taller than 2.7 m. The percentage of trees taller than 2.7 m increased to 48% after the third year, to 60% after the fourth year, and to 85% after the sixth year.

Primrose willow competition had a greater effect on stem diameter and crown diameter than on tree height. Baldcypress diameters at breast height (DBH) after the sixth year averaged 8 cm with primrose willow as compared to 12 cm in the weeded plots (difference significant at $P<.01$) (Figure 3). After the fourth year, baldcypress trees in the primrose willow plots had developed an average crown diameter of 109 cm as compared to 145 cm in the weeded plots (difference significant at $P<.01$) (Figure 4). After the sixth year, baldcypress in the primrose willow plots had developed an average crown diameter of 144 cm as compared to 189 cm in the weeded plots (difference significant at $P<.01$).

There was no difference in percent survival of baldcypress after six years between the treatments (weeded plots had 100% tree survival and untreated plots had 98% tree survival).

P2 Site: All three tree species at this site had significant mortality during the first year. Popash and red maple had lower mortality under a primrose willow canopy than in the weeded plots. By June of 1993, red maple trees had suffered 70% mortality in the weeded plots and 58% in the untreated plots. Whereas, popash had a 13% mortality rate in the weeded plots and 0% in the untreated plots. Baldcypress mortality was 14% in both treatments.

Dead trees were replaced after the June 1993 mortality assessment, and survival determinations in subsequent years included the replacement trees. After the third year, survival of baldcypress was reduced from 98% in the weeded plots to 63% in the untreated primrose willow plots. Primrose willow did not affect survival of either popash (untreated 88%, weeded

85%) or red maple (untreated 78%, weeded 75%).

The P2 site was more complicated to analyze than the P1 site, because the more dense and more tangled primrose willow at the P2 site leaned on the saplings, causing some trees to be bent in the untreated plots. Thus data from the untreated plots are presented as averages of the straight trees alone and as averages of the straight plus bent trees combined. The effect of primrose willow on bending of trees was only slight the first year, but it became more of a factor in subsequent years.

In the first year, baldcypress (Figure 5) was clearly taller ($P<.01$) under the primrose willow canopy than in the weeded plots, as occurred with baldcypress at the P1 site, whereas popash (Figure 6) and red maple (Figure 7) heights were unaffected.

In the second year, popash (Figure 6) trees were taller ($P<.05$) in the weeded plots than under the primrose willow. Heights of baldcypress (Figure 5) trees were similar in weeded and untreated plots in the second year. Red maple (Figure 7) heights were also unaffected by the presence or absence of primrose willow.

By the third year the primrose willow was very tangled and had bent 40% of the baldcypress, 46% of the popash, and 48% of the red maple trees. Consequently, the effects of bending became evident in the height data. Baldcypress trees in the weeded plots were slightly, but not significantly, taller than the straight trees in the untreated primrose willow plots, but the trees in the weeded plots were 28% taller ($P<.05$) than the straight plus bent trees in the untreated plots (Figure 5). Popash height in the weeded plots was not significantly different from that of straight trees, or straight plus bent trees, in the untreated primrose willow plots (Figure 6). Red maple trees in the untreated primrose willow plots were slightly, but not significantly, taller than the trees in the weeded plots (Figure 7).

Because of the trajectory of the growth curves in Figures 5, 6, and 7, it is anticipated that all three tree species will begin to overtop the primrose

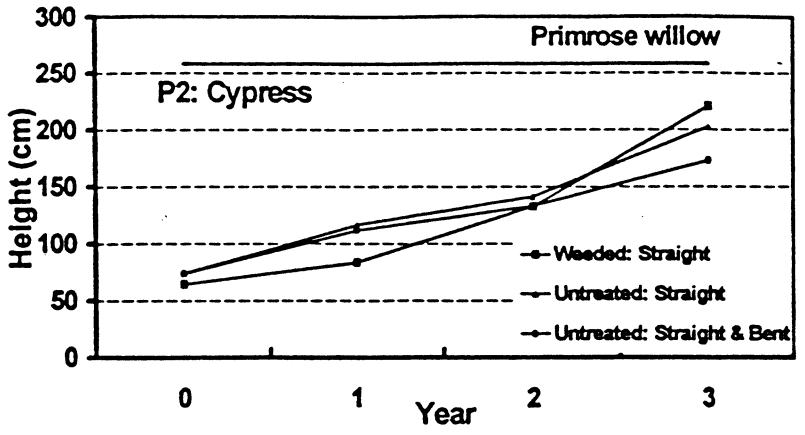

Figure 5. Average yearly height (cm) of baldcypress in weeded and untreated primrose willow plots at the P2 site.

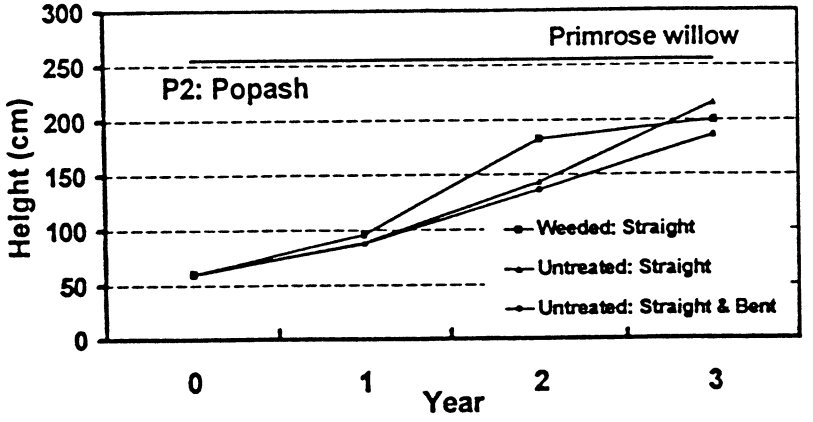

Figure 6. Average yearly height (cm) of popash in weeded and untreated primrose willow plots at the P2 site.

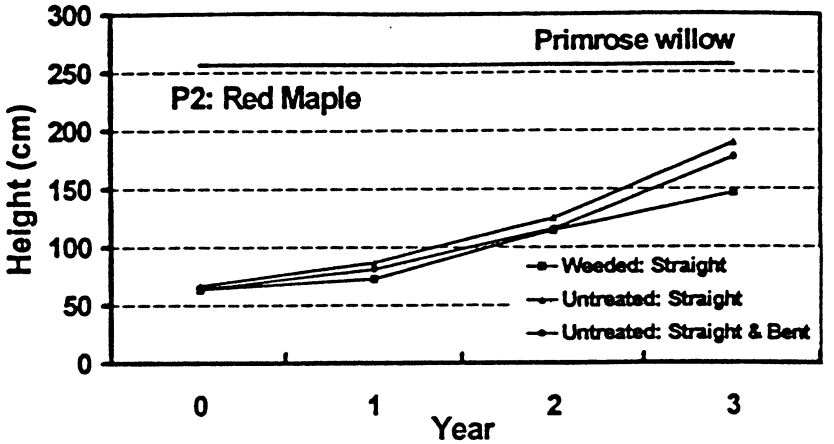

Figure 7. Average yearly height (cm) of red maple in weeded and untreated primrose willow plots at the P2 site.

willow toward the end of the next growing season.

Basal diameter growth of baldcypress (Figure 8), popash (Figure 9), and red maple (Figure 10) was clearly greater for the trees in the weeded plots than in the untreated plots. By the third year the average basal diameter was 155% greater for baldcypress ($P<.01$), 53% greater for popash ($P<.01$), and 35% greater for red maple ($P<.05$), in the weeded plots than in the untreated primrose willow plots.

C1 and C2 Sites: The baldcypress and popash trees behaved similarly at both sites even though the cattail was more dense and about 1.5 m taller at the C1 site than at the C2 site.

After the first growing season, both baldcypress (Figure 11) and popash (Figure 12) at the C1 site were slightly taller ($P<.01$) in the presence of cattail than in the weeded plots, but baldcypress and popash were not significantly affected by treatment at the C2 site (Figures 13 and 14). At the end of the second year, the average heights of baldcypress and popash at the C1 site were basically the same in weeded or untreated plots (Figures 11 and 12). At the C2 site, baldcypress (Figure 13) was slightly taller ($P<.05$), and popash (Figure 14) was marginally taller ($P<.10$), in the weeded than in the untreated cattail plots. The effect of cattail on stem diameters was inconsistent in the first year, but stem diameters were clearly greater ($P<.01$) after the second year at both sites with cattail removed (Figures 15, 16, 17 and 18).

The cattail canopy did not have a major effect on tree survival after the second year at either site. Although baldcypress survival at the C1 site was minorly reduced from 98% to 85%, it was unaffected at the C2 site (both treatments 100%). Popash survival was unaffected by cattail (about 85 % for both treatments at C1 and about 97 % for both treatments at C2).

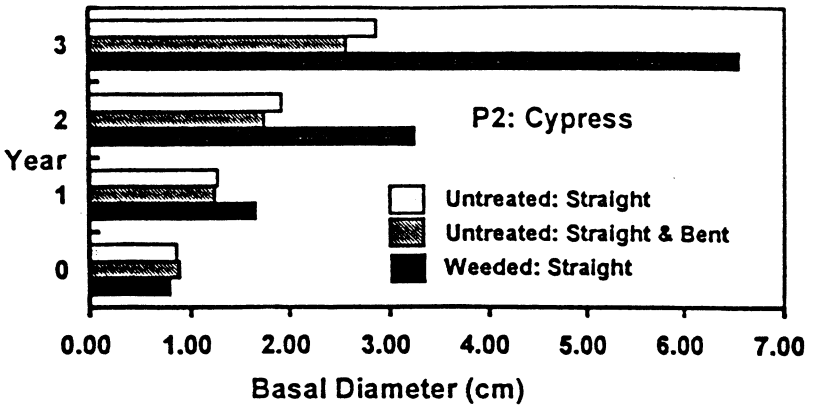

Figure 8. Average yearly basal diameter (cm) measured at 15 cm height of baldcypress in weeded and untreated primrose willow plots at the P2 site.

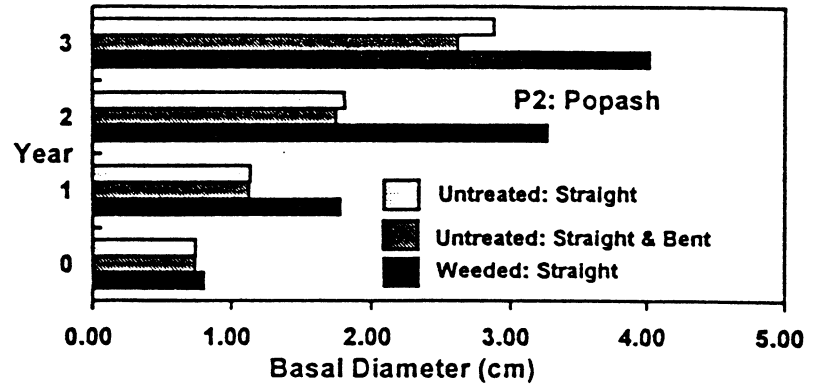

Figure 9. Average yearly basal diameter (cm) measured at 15 cm height of popash in weeded and untreated primrose willow plots at the P2 site.

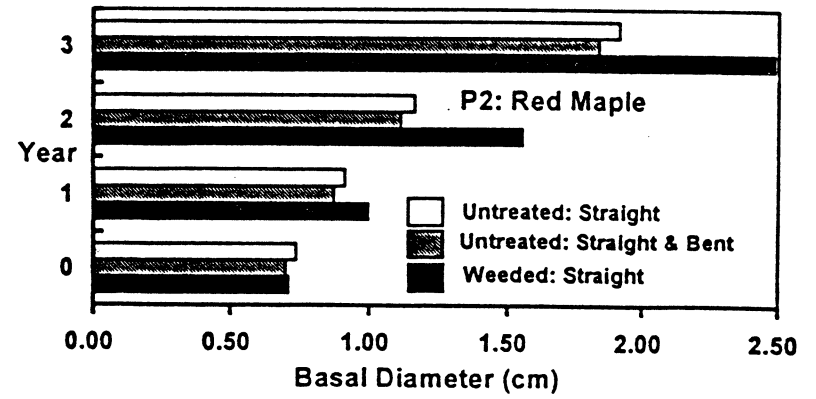

Figure 10. Average yearly basal diameter (cm) measured at 15 cm height of red maple in weeded and untreated primrose willow plots at the P2 site.

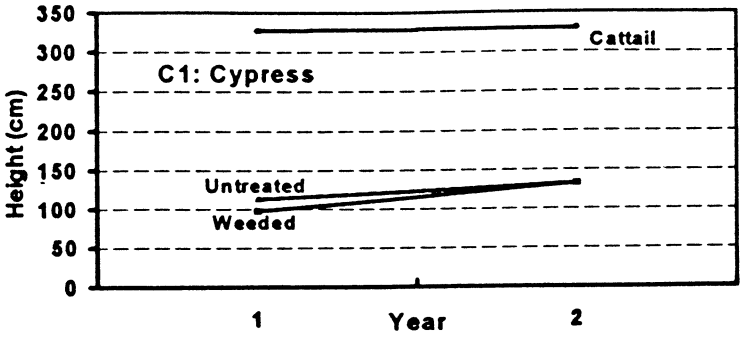

Figure 11. Average yearly height (cm) of baldcypress in weeded and untreated cattail plots at the C1 site.

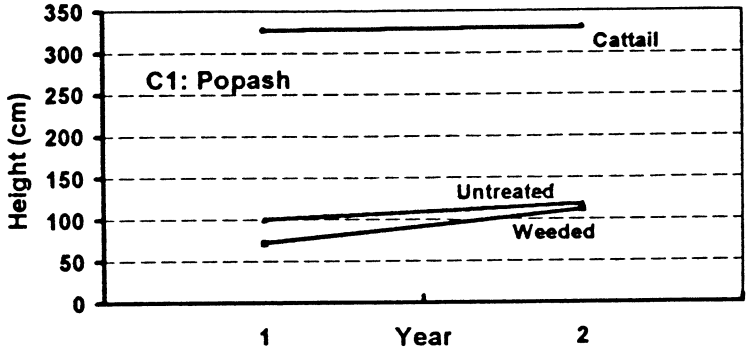

Figure 12. Average yearly height (cm) of popash in weeded and untreated cattail plots at the C1 site.

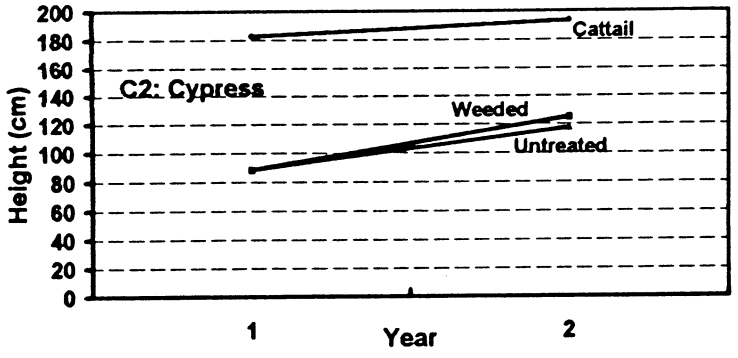

Figure 13. Average yearly height (cm) of baldcypress in weeded and untreated cattail plots at the C2 site.

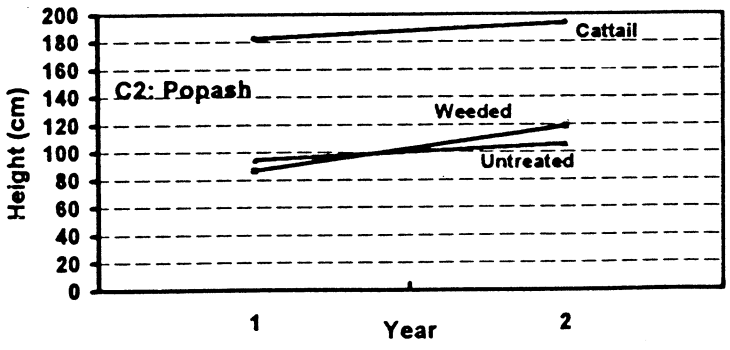

Figure 14. Average yearly height (cm) of popash in weeded and untreated cattail plots at the C2 site.

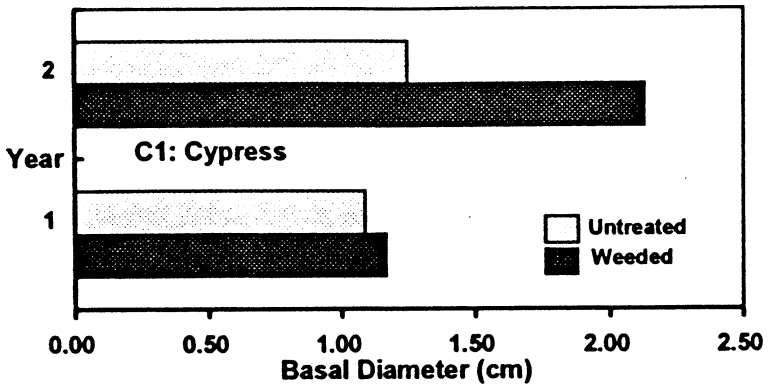

Figure 15. Average yearly basal diameter (cm) measured at 50 cm height of baldcypress in weeded and untreated cattail plots at the C1 site.

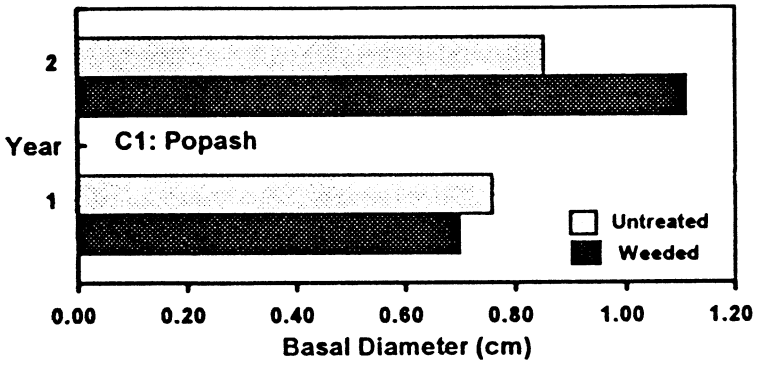

Figure 16. Average yearly basal diameter (cm) measured at 50 cm height of popash in weeded and untreated cattail plots at the C1 site.

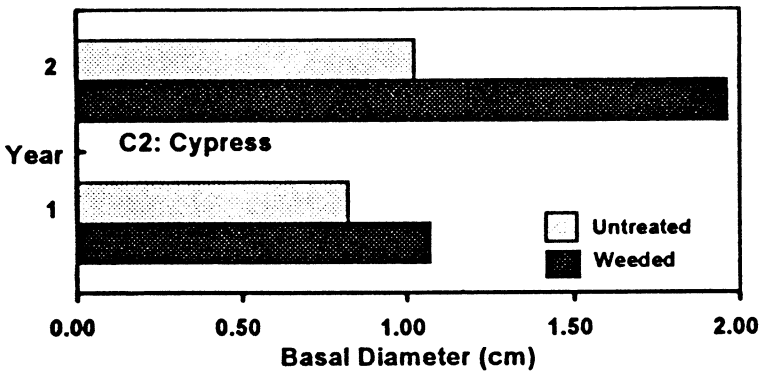

Figure 17. Average yearly height (cm) measured at 50 cm height of baldcypress in weeded and untreated cattail plots at the C2 site.

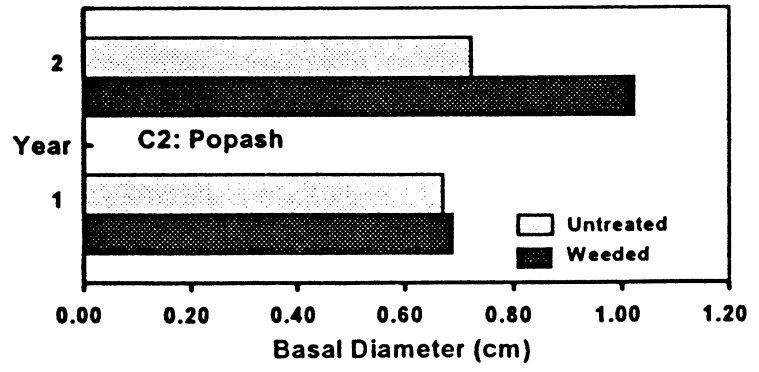

Figure 18. Average yearly basal diameter (cm) measured at 50 cm height of popash in weeded and untreated cattail plots at the C2 site.

DISCUSSION

Primrose willow had only a slight effect on heights of baldcypress through three growing seasons at the P1 site. In the first growing season, baldcypress height growth was greater in the presence of primrose willow than with primrose willow removed. This may have been due to a protective nurse crop effect or to an etiolation effect because of shading from the primrose willow. In subsequent years, growth rates were greater with primrose willow removed, so that by the end of the sixth growing season baldcypress height was 25 percent greater, and DBH was 50 percent greater, without primrose willow than with primrose willow competition.

It is important to note that baldcypress growth rates in the presence of primrose willow were still substantial at the P1 site. Eighteen percent of the baldcypress had overtopped the 2.7 m tall primrose willow by the end of the second growing season. The percentage increased to 50% by the end of the third growing season and to 85% by the end of the sixth growing season.

Baldcypress growth at the P1 site was rapid. Even with primrose willow competition, tree height was greater on this reclaimed mine site than was height of comparably aged trees on unmined mitigation wetlands as determined by Denton (1990). DBH was also greater, but crown diameter was slightly less on this reclaimed mined site with primrose willow than on the unmined wetlands. Baldcypress height, DBH, and crown diameter were all much greater on the weeded P1 plots than on the several mitigation wetlands.

Baldcypress height growth was slower at the P2 site than at the P1 site, but it was still comparable to the mitigation wetlands (Denton 1990). The trees at the P1 site had reached an average height of 2 m by the end of the second growing season, while it required three growing seasons for the P2 trees to reach 2 m in height. However, similar to the P1 site, baldcypress at the P2 site also exhibited greater growth the first year in the presence of

primrose willow than with the primrose willow competition removed. There was an additional complication at the P2 site in that the primrose willow there was more dense and tangled than at the P1 site and was more prone to lean on and bend some of the trees.

Popash heights were similar after three years either with or without primrose willow competition at the P2 site. Height growth rates of popash were similar to those of baldcypress. Red maple height growth was slower than that of popash or baldcypress, but red maple heights were slightly, but not significantly, greater in the presence of primrose willow at the end of the third growing season than with the primrose willow removed. Removal of primrose willow enhanced stem diameter growth for red maple, popash and baldcypress.

Removal of cattail caused greater stem diameter growth of popash and baldcypress, but it had little effect on tree height after two growing seasons. Baldcypress and popash at the C1 site and popash at the C2 site tended to be taller (and also appeared more robust) in the presence of cattail the first year than with the cattail removed. In the subsequent year there appeared to be a slight trend for greater height growth rates with cattail removed, but it is too early in the study to draw firm conclusions on the magnitude of long term effects of cattail on tree growth. The initial effect seems small.

The modest effect of primrose willow on tree height growth indicates that control measures, which are often expensive, may not be necessary in establishing forested wetlands on reclaimed phosphate mined lands. Baldcypress trees at the P1 site were able to grow through and overtop the primrose willow, and trees at the P2 site seem to be following suit. If trees are planted at a sufficient density, it seems likely that they will eventually develop a canopy cover that will shade out the primrose willow. The presence of primrose willow may even have a temporary beneficial effect on understory development. We observed an abundance of ferns and begonias beneath the shade of the primrose willow at the P1 site, but in the plots where primrose willow was removed, the result was a ground cover

of weedy species.

One factor that warrants further consideration, however, is the effect of primrose willow at the P2 site on bending of some of the trees. We have observed that elderberry (*Sambucus canadensis*) tends to grow more erect than primrose willow but is able to occupy similar habitat and may have greater wildlife value. The presence of some elderberry intermixed with the primrose willow at the P1 site may have helped prevent bending of trees. In contrast, there was virtually no elderberry at the P2 site. We have begun experiments that will examine the possibility of using elderberry as a forested wetland "nurse crop." It is hypothesized that elderberry will occupy space that otherwise might be inhabited by primrose willow, and the elderberry should provide canopy cover and habitat for shade requiring understory plants.

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FISH HABITAT RESTORATION OF AN OXBOW LAKE IN THE MISSISSIPPI DELTA

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ABSTRACT

Lake Whittington, historically contiguous with the Mississippi River, now experiences annual de-watering. A weir proposed for the inlet/outlet channel would permit water exchange during high river stages, and pool water when river stages are low. We developed regression models from empirical data and, using Habitat Evaluation Procedure (HEP) incrementally quantified fishery benefits associated with alternative weir heights.

Fish-habitat data were collected in summer 1995 from 3 oxbow lakes in the Mississippi River: Lake Bolivar (permanently isolated from the Mississippi River); Lake Whittington; Lake Beulah (seasonally connected, with a weir pooling water during low river stages). Depth and water quality were measured at 9 stations in each lake. Fishes were sampled using overnight sets of floating Plexiglas traps "baited" with chemical light sticks. Young-of-year sunfishes (*Lepomis* spp.) and inland silverside (*Menidia beryllina*) were numerically dominant. Fish abundances were used as dependent (response) variables, physical habitat data as independent variables in regression models. Both taxa were negatively correlated with mean cross-sectional lake depth ($p < 0.01$), sunfish predominating at depths < 2 m, silversides at 6 m. Habitat Suitability

Index (HSI) curves, values of which range from 0.00 (no habitat value) to 1.00 (optimum habitat), were constructed from standardized fish frequency distributions and regression. Habitat under existing conditions and for 5 weir heights was quantified as Habitat Units (HU's) by multiplying surface area of specific lake depths with corresponding HSI values. HU's for both taxa increased substantially at the lowest weir height. Sunfish HU's asymptoted at an intermediate weir height, silverside HU's increased linearly over the full range of weir heights. Model verification is not possible until a weir is constructed, but abundance of littoral fishes in the three lakes strongly supports the results of HEP simulations.

INTRODUCTION

Lake Whittington is an oxbow lake of the Mississippi River, commercially and recreationally fished for more than 50 years, but now experiencing declining fisheries and progressive seasonal dewatering. Standing crops of black crappie (*Pomoxis nigromaculatus*) have decreased and common carp (*Cyprinus carpio*) increased; reproduction of largemouth bass and densities of intermediate sized sunfishes are low (Lucas, 1992). Dewatering may reduce availability of spawning area in littoral zones and concentrate fishes thereby increasing predation.

During high and intermediate stages on the Mississippi River, the surface elevation of Lake Whittington is controlled by the Mississippi River via an inlet/outlet channel. Bottom of the inlet/outlet channel and natural obstructions in the lake control the surface elevation of the lake during low stages of the Mississippi River. During low stages, the lake is dewatered, isolated from the river, and dries partially to form three separate pools consisting of approximately 405 hectares. Bankfull surface area of the lake is 1215 hectares. The lake typically dewaters during late summer and may remain in this condition through late autumn. Users of the lake have noted that the lake elevation declines to lower elevations with greater frequency, duration, and magnitude than previously. Consequently, a water control structure in the inlet/outlet channel was recommended to

maintain a minimum pool during late summer and autumn. Engineering studies indicate that a compacted earth and stone weir is structurally adequate for the weir elevations investigated. Construction would be performed during late summer when the Mississippi River is at its lowest stages.

A weir would minimize negative effects of dewatering on the fishery. To determine weir height providing greatest fish habitat benefits, we evaluated 5 alternative weir elevations. We quantified spawning and foraging habitat by modeling abundances of young-of-year sunfishes and inland silverside respectively. Sunfishes, as a group and as individual species, spawn from mid-April through September (Robison and Buchanan, 1988; Turner *et al.*, 1994). They are abundant in oxbow lakes and intensively exploited by anglers. Inland silversides also exhibit a prolonged reproductive season, possibly from March through September (Hildebrand, 1922; Hubbs, 1982; Middaugh and Hemmer, 1992). They were historically abundant in Mississippi delta oxbows (Cook, 1959), but recent surveys suggest low numbers (Robison and Buchanan, 1988; Baker et al., 1991), possibly due to very high water temperatures (> 37 ⁻C) recorded during low water (Lowery *et al.*, 1987; Lucas 1992). Temperatures over 30 ⁻C are believed critical in cessation of spawning, and in post-spawning mortality (Hubbs and Bailey, 1977; Hubbs and Dean, 1979). Inland silversides are an important link between invertebrates and piscivorous fishes; they eat zooplankton and surface insects (McComas and Drenner, 1982) and are consumed in large numbers by temperate and black basses (Mense, 1967; Echelle and Mense, 1968; Matthews et al., 1992).

STUDY SITES

We surveyed Lake Whittington and two nearby oxbow lakes of the Mississippi delta, north of Greenville, MS (Fig. 1). Lake Bolivar occurs near river km 904 and is permanently isolated from the Mississippi River by a mainline levee constructed in the 1930's. It is surrounded by agricultural fields. There is a thin line of trees around the perimeter, and

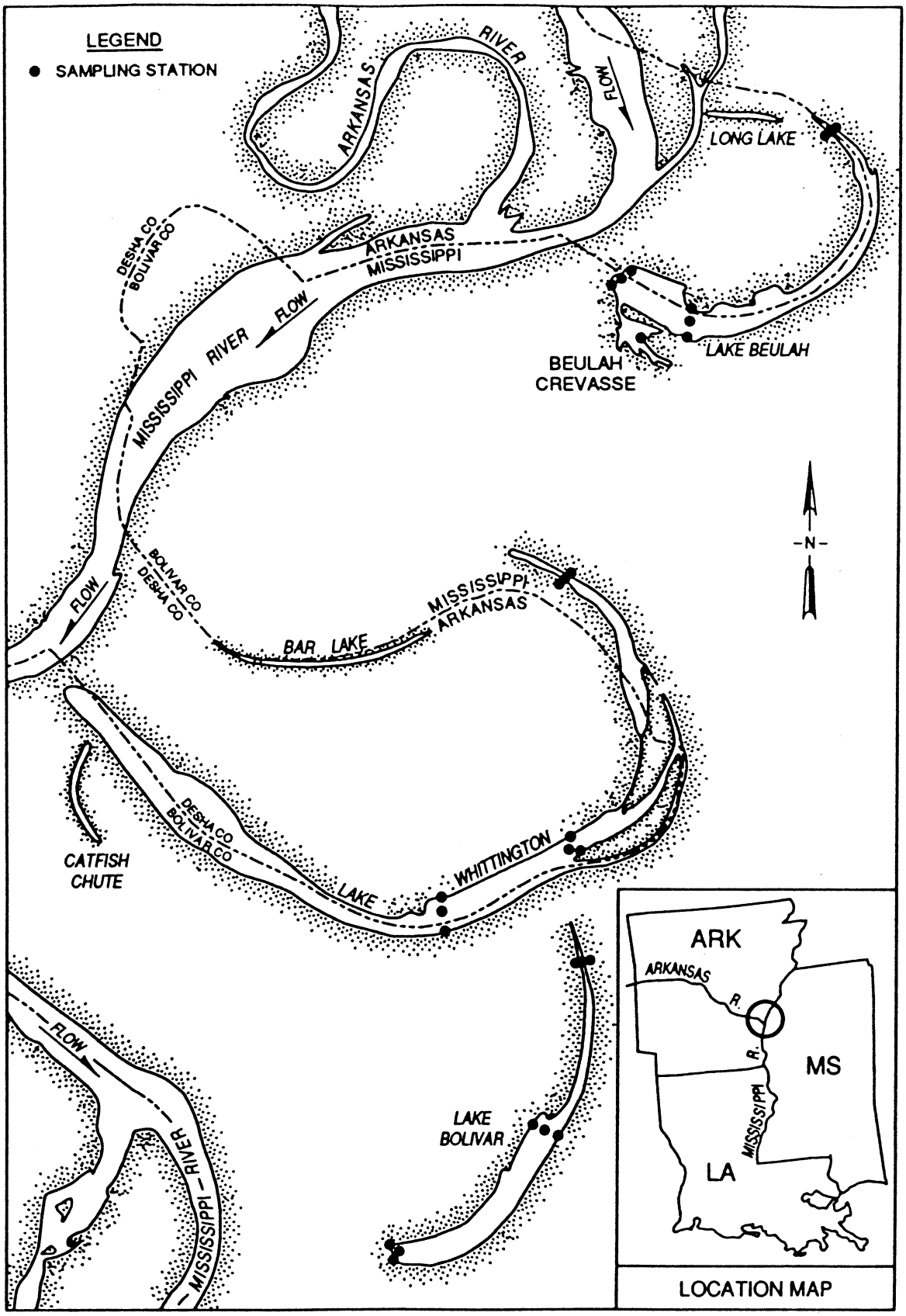

Fig. 1. Oxbow lakes and stations sampled by light-trap

hardwood forests at the upper end of the lake. Lake Whittington occurs near river km 926, and was formed in 1937 by the U.S. Army Corps of Engineers after completion of Caulk Island Cutoff. Several residences and fishing camps are located on the edge of the lake. Lake Beulah occurs east of the Rosedale Bend near river km 939. It was created in 1863 as a result of the Napoleon Bend cutoff, and a weir constructed in 1955 (Lucas, 1992). Lakes Whittington and Beulah occur on the Mississippi River floodplain, riverward of mainline levees, and are largely surrounded by hardwood forests. They are flooded seasonally (Jan-Apr), and dewatered in late summer-early autumn, especially in Lake Whittington, in which water is not pooled by a weir.

MATERIALS AND METHODS

Field sampling - Floating Plexiglas light traps were used to collect surface fishes (Killgore 1994). Light traps (n=3) were placed at nine stations: upper, middle, and lower lake at outer shoreline, mid-channel, and inner shorelines (Fig. 1). Lake Bolivar was sampled 13-14 Jun 95, Lake Whittington 14-15 Jun 95, and Lake Beulah, 15-16 Jun 95. At Lake Beulah, a fourth station, Beulah Crevasse (also known as Merigold Lake), was sampled. Each trap was "baited" with a yellow Cyalume™ chemical light stick (minimum illumination of 12 hours), deployed early evening (1730-2000), and recovered the following morning. All fishes were preserved in 5% formalin. In the laboratory, fishes from each sample were sorted; juveniles and adults were transferred to 50% isopropanol, larvae were stored in 5% formalin. Specimens were deposited in collections of Northeast Louisiana University Museum of Zoology and the Waterways Experiment Station Environmental Laboratory.

Depth and water quality were measured at each light trap. A cross-sectional transect was designated and water depth measured at 10 approximately equi-distant points with a surveyor's stadia rod (< 5 m) or with a portable SONAR depth gauge (> 5 m). Turbidity of surface water was measured using a Hach 2100P turbidimeter. Water temperature,

conductivity, dissolved oxygen, and pH were measured using a Hydrolab submersible water quality probe at lake surface and near lake bottom.

Habitat Evaluation Procedure - Benefits were calculated using the Habitat Evaluation Procedure (HEP) which expresses habitat quantity as habitat area weighted by habitat quality (USFWS, 1980), i.e.,

Habitat Units = Habitat Area X Habitat Suitability Index

Empirical (curvelinear) relationships between fish abundance and physical habitat may be used to objectively identify significant habitat variables and quantify habitat value. This approach necessitates two assumptions:

1) Fish abundance is a positive, direct expression of habitat quality.
2) Relevant habitat variables, and ranges for those variables, were included in analysis.

To identify relevant habitat variables, we used Pearson product moment correlation coefficients and multiple regression analysis, MAXR (SAS, 1985). This provided quantitative models of relationship between one or more independent (habitat) variables and a single dependent (fish abundance) variable. Regression models were expressed as

$$\textbf{Fish abundance} = b + m_1\textbf{Habitat Correlate}_1 + m_2\textbf{Habitat Correlate}_2...$$

in which habitat correlates are parameters significantly correlated with fish abundance, b (y-intercept) and m (slope) are constants. Model allows abundance of fish (number/light-trap) to be predicted for specific values of habitat variables. Regression equations were used as formula to calculate indices of habitat quality or Habitat Suitability Indices (HSI's). This was done by dividing by the maximum predicted value for the dependent variable: i.e., number of fish calculated for modal value(s) of primary habitat correlate(s).

$$HSI = \frac{b + m_1 \text{Habitat Correlate}_1}{\text{Maximum fish abundance}}$$

Equation standardizes calculated values of fish abundance to a scale ranging from 0.00 (no habitat value) to 1.00 (maximum habitat value).

Bathymetric data for Lake Whittington were used to calculate surface areas of different intervals of water depth for existing conditions and for alternative weir elevations using planimetry.

RESULTS

Physical habitat - Lake Bolivar was shallow (< 2 m), moderately conductive (103-167 umhos), with a longitudinal turbidity gradient (upper-lower ends of lake, 10-382 NTU's). Lake Whittington was deep (11-18 m), highly conductive (349-363 umhos), and clear (4.9-9.1 NTU's). Lake Beulah was moderately deep (7-11 m), highly conductive (350-381 umhos), and clear (6.2-15.3 NTU's). In all three lakes, water temperatures were warm (24-27 ° C), and pH was neutral to basic (7.2-8.5). Most values of dissolved oxygen ranged from 4-10 mg/l. Hypoxia (< 2.0 mg/l) was observed, however, in Lake Whittington: surface and bottom waters in the upper lake, bottom waters middle and lower lake. In Lakes Beulah and Whittington, ranges were narrow for surface water temperature (< 2.2 ° C), conductivity (< 35 umhos), and pH (< 0.8), so these variables were excluded from fish-habitat correlation analyses.

Fish-habitat relationships - Light-traps (n=83) collected 3208 fish, representing at least 12 species (Table 1). Catch rates ranged from 0 to 299 fish/trap-night. Inland silversides and young-of-year sunfishes respectively constituted 73 and 22 % of all fishes. The third-most abundant species, threadfin shad, constituted only 2.5% of fish. No inland silversides were collected in Lake Bolivar; mean number was

8.0/trap-night in Lake Whittington and 70.8/trap-night in Lake Beulah. Conversely, maximum abundance of sunfishes was observed in Lake Bolivar, 22.6/trap-night; moderately low catches were obtained in Lake Whittington, 1.5/trap-night, and Lake Beulah, 1.0/trap-night.

Habitat correlates were similar for both taxa. Sunfishes and silversides were negatively correlated with mean water depth, range of water depths, and water depth at trap ($r < -0.36$, $p < 0.010$), and positively correlated with turbidity ($r > 0.43$, $p < 0.001$). Silversides were positively correlated with dissolved oxygen ($r = 0.27$, $p = 0.04$), but sunfishes were not ($r = -0.02$, $p = 0.83$). Stepwise multiple regression resulted in the following models for both taxa:

Sunfishes = 14.518 - 1.096(Mean Depth) + 0.039(Turbidity),
$r^2 = 0.309$, $p = 0.0001$

Silversides = 46.401 - 5.768(Mean Depth) + 5.543(Turbidity),
$r^2 = 0.243$, $p = 0.0006$

Modified habitat models - Although statistically significant, above models assume a consistent negative (linear) relationship of fish abundance with depth. Fish abundance, however, was highest at some intermediate depth: 1.3 m for sunfish, 6.3 m for silversides (Fig. 2). Regressions were recalculated so that depth corresponding to modal fish abundance was used as a starting point for two separate regressions: a positive regression from 0 depth to depth of modal abundance, and a negative regression from depth of modal abundance to maximum depth of occurrence. Since turbidity is not expected to change as a result of the project, "second generation" models did not include that variable. Regressions were standardized by dividing by maximum predicted fish abundance:

Table 1. Total numbers of fishes collected in three Mississippi River oxbow lakes using light-traps, 13-15 Jun 1995. Silversides and mosquitofish included larvae, juveniles, and adults. Other taxa were comprised exclusively by larvae and small juveniles.

	Bolivar n = 27	Whittington n = 26	Beulah n = 30
Sunfishes			
Bluegill, *Lepomis macrochirus*	1	3	1
Warmouth, *L. gulosus*	1		
Unidentified sunfishes, *Lepomis* sp.	611	68	31
White crappie, *Pomoxis annularis*	1		
Forage Fishes			
Threadfin shad, *Dorosoma petenense*		38	41
Unidentified clupeids, *Dorosoma* sp.		6	14
Unidentified topminnows, *Fundulus* sp.			1
Brook silverides, *Labidesthes sicculus*		3	10
Inland silversides, *Menidia beryllina*		207	2124
Unidentified silversides, Atherinid sp.		1	
Other Fishes			
Spotted gar, *Lepisosteus oculatus*	1	2	
Common carp, *Cyprinus carpio*		4	
Yellow bullhead, *Ameiurus natalis*	1		
Channel catfish, *Ictalurus punctatus*	1		
Mosquitofish, *Gambusia affinis*	31	2	4
Total fishes	648	334	2226

$$HSI_{Sunfish > 1.3\ m} = \frac{21.699 - 1.693(\text{Mean depth})}{19.6}$$

$$HSI_{Sunfish < 1.3\ m} = \frac{-31.608 + 47.664(\text{Mean depth})}{26.5}$$

$$HSI_{Silversides > 6.3} = \frac{122.724 - 8.094(\text{Mean depth})}{73.4}$$

Because no silversides were collected in Lake Bolivar where depths were less than 1.3 m, we assumed an HSI of 0.00. In the other two lakes, no silversides were collected at mean depths less than 6.3 m but we assumed fish were present at these depths. Consequently, a positive linear relationship for the depth interval of 1.3 to 6.3 m was obtained from Figure 3 using HSI values as the dependent variable in the regression:

$$HSI_{Silversides < 6.3\ m} = - 0.250 + 0.203(\text{Mean depth})$$

Benefits analysis - Existing 405 hectare low-water pool in Lake Whittington has a weighted average water depth of approximately 2 m. Lowest weir elevation evaluated will create a permanent pool of 964 hectares with a weighted average water depth of 3 m; highest weir elevation evaluated will maintain a minimum of 1211 hectares and weighted average water depth of approximately 5 m.

Shallow water, 1-3 m, predominates under existing conditions, so sunfish habitat is relatively high (230 HU's) compared with that of silversides (65 HU's)(Fig. 4). At the lowest weir elevation, substantial increases in shallow and moderate depths, 4-8 m, occur and there are correspondingly substantial increases in habitat of both species. With successively

increasing weir elevations, however, shallow water persists or increases slightly but moderately deep water increases linearly. As a result, sunfish habitat asymptotes at an intermediate weir elevation and silverside habitat increases linearly. Benefits expressed as gained HU's (and percentage increase over existing conditions) are:

Weir	Sunfishes	Silversides
110.0	436 (189%)	238 (366%)
112.5	540 (235%)	348 (535%)
115.0	587 (254%)	470 (723%)
117.0	590 (256%)	568 (874%)
119.5	561 (244%)	685 (1055%)

DISCUSSION

Multiple regression equations function similarly to HSI "blue book" equations, by providing measures of habitat quality based on multiple habitat variables (e.g., Stuber *et al.*, 1982). Multiple regression equations, however, offer several refinements, because they are empirical and do not entail *a priori* decisions regarding relationships between habitat parameters and fishes, thus reducing institutional bias. Use of multiple regression statistics allows the elimination of irrelevant variables from the final predictive model and quantification of the degree of correlation between habitat variables and fish communities (via correlation coefficients, and probability levels).

Models generated here are consistent with published accounts of fish habitat. Nine species of sunfishes occur in the oxbow lakes, but bluegill, *Lepomis macrochirus*, are numerically dominant (unpublished data). Bluegill, like most members of their genus, spawn preferentially in depths < 2 m (Breder, 1936; Robison and Buchanan, 1988). Silversides are found in littoral and pelagic habitats, apparently migrating into shallow areas during the day, offshore into deeper water at night (Mense, 1967; Hubbs, 1977).

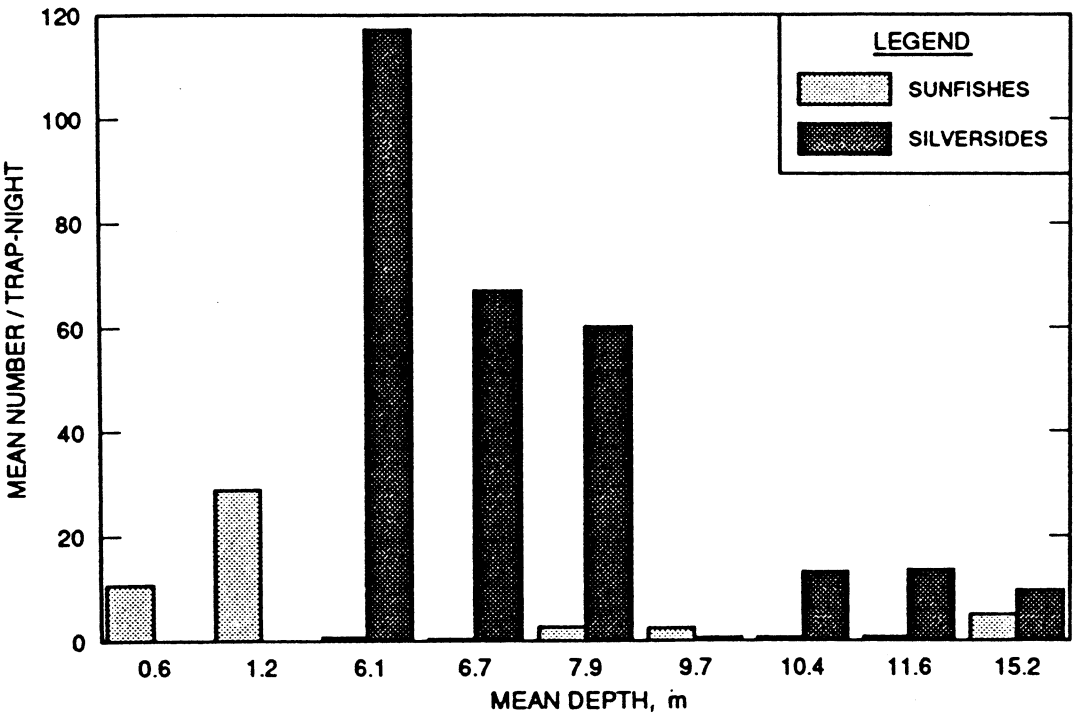

Fig. 2. Distribution of two taxa of fishes by lake depth; x-axis is not drawn to scale.

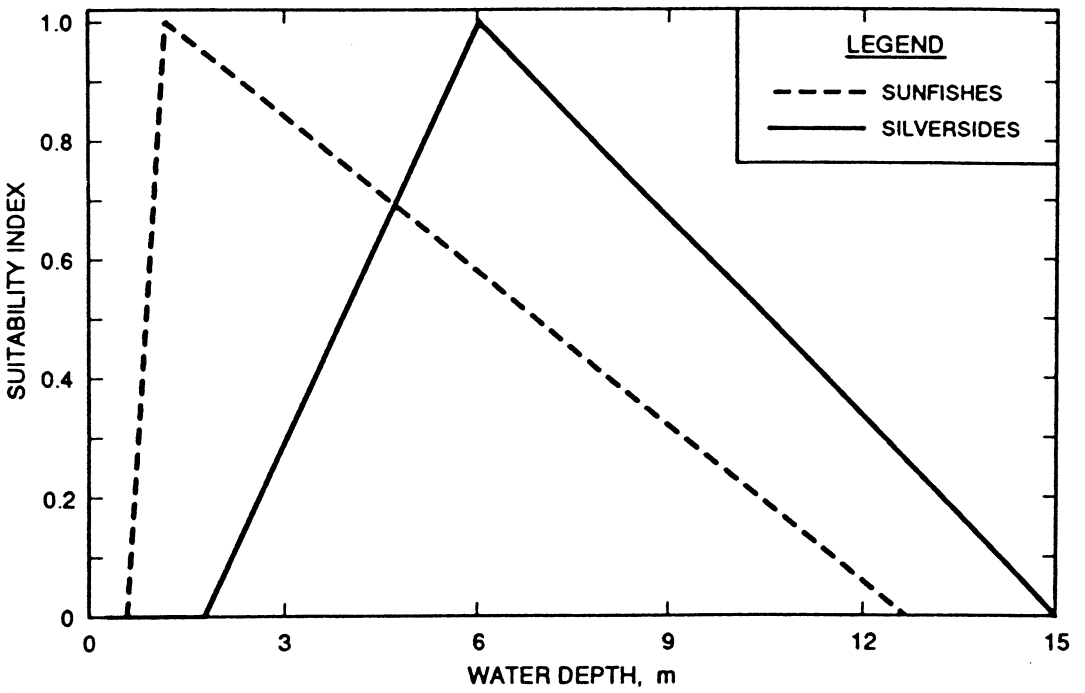

Fig. 3. Habitat suitability curves of lake depths for two taxa of fishes.

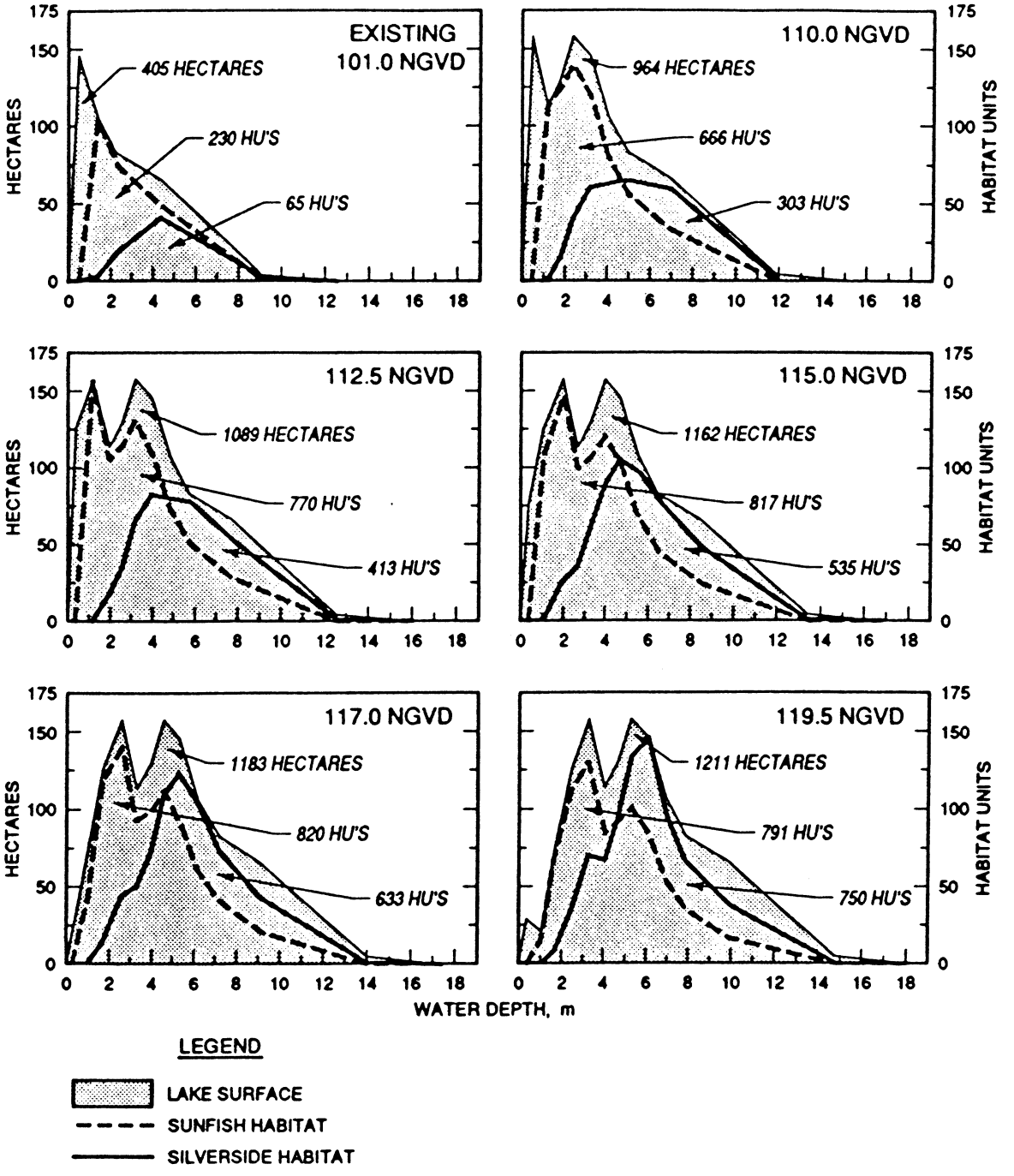

Fig. 4. Low water acreage and corresponding fish Habitat Units, by depth interval, for existing conditions and project alternatives. Total acreage and total Habitat Units for each taxon are indicated with each curve.

Pooled water in late summer will provide increased habitat for both taxa, probably resulting in higher numbers of fish, due to increased habitat volume, cooler temperatures, and increased production of zooplankton. Due to different depth preferences, benefits for the two taxa are maximized at different project alternatives. Drought negatively affects larval sunfishes less so than other taxa, but stable water levels in late season allow multiple spawnings following floods (Brown and Coon, 1994). An ecosystem-level simulation of environmental perturbations in Lake Texoma indicated that stabilized water levels, while affecting no other fish or fishery component, increased production of larvae (Patten et al., 1975). Conservatively, a weir height of 112.5' NGVD would provide near maximum habitat benefits for sunfish young-of-year, and a substantial increase for silversides.

Littoral densities of juveniles and adult fish (unpublished data), corresponding to those of larval fish (Table 1), suggest that increased acreages of shallow and moderately deep water from weir construction will not only result in greater spawning success of both groups, but that will translate into higher juvenile and adult fish densities. Shallow Lake Bolivar exhibits high littoral densities of sunfish (11.5/seine haul) and no silversides. Moderately deep Lake Beulah exhibits intermediate densities of sunfishes (8.3/seine haul) but maximum density of inland silversides (82.3/seine haul). Deeper Lake Whittington exhibits lowest density of sunfishes (4.2/seine haul), intermediate density of inland silversides (41.7/seine haul).

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STRATEGIC WETLAND ANALYSIS AND MITIGATION PROGRAM

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ABSTRACT

The North Carolina Department of Transportation (NCDOT) is the largest state DOT in the United States, responsible for over 77,000 miles of state roads and bridges. As a result, NCDOT has one of the nation's most comprehensive transportation improvement plans. For the period 1996-2004 the North Carolina Department of Transportation (NCDOT) has scheduled over 2,500 highway and bridge improvement projects. North Carolina also has an abundance of environmentally sensitive wetlands along it expansive coast and coastal plain areas. Until recently, the NCDOT Environmental Permits and Wetland Mitigation Unit had no systematic way of predicting the impact of these highway improvements on wetlands. In 1995, however, the NCDOT in conjunction with the Institute for Transportation Research and Education (ITRE), conducted a study utilizing Geographic Information Systems (GIS) to estimate wetland impacts caused by highway construction projects programmed for 1996-2004 in order to forecast wetland mitigation needs. Wetland

impacts were estimated by measuring the overlap of highway improvement projects with wetland areas using two primary data sources: the 1996-2002 Transportation Improvement Program (TIP) and the National Wetland Inventory (NWI). Using geometric and coincidence modeling techniques afforded by GIS wetland impacts were identified, quantified, summarized, and reported by amount, type, location, and timing. Total wetland impacts caused by highway improvements programmed through 2004 were calculated to be 5,800 acres. Over half bottomland hardwood and swamp hardwood habitats; 70% occur in the Cape Fear, Neuse, and Pasquotank River basins: 85% occur in the coastal Plain Physiographic Region; and 65% occur in coastal plain Physiographic Region. For planning purposes a ration of 3.1 mitigation acres per impacted estimated to be 18,000 acres. Only 2,200 acres were found to be available now, thus the net mitigation need is 15,800 acres. In order to mitigate wetland impacts in step with their incidence for the period 1996-2004, the mitigation acreage obtained by NCDOT may need to increase to three times its current annual rate.

INTRODUCTION

Objective

This study was conducted to estimate wetland impacts in acres of highway and major bridge construction projects programmed for 1996-2004 in order to forecast wetland mitigation needs of the North Carolina Department of Transportation (DOT).

Methodology

Wetland impacts were estimated by measuring the overlap of highway improvement projects with wetland areas using two primary data sources: the 1996-2002 Transportation Improvement Program (TIP), and National Wetland Inventory (NWI) maps. Projects under construction were not

included, and all impacts were assigned to the year of the earliest construction section of the project. Using GIS and database research methods, these impacts were identified, quantified, summarized, and reported by amount, type, location, and timing. The years are reported as federal fiscal years (FFY), October 1 to September 30.

Results

Total wetland impacts of all projects are estimated to be 5,800 acres. It is worth noting that one-half of these impacts occur within the first three years of the nine-year study period (1996-2004). Also, of all impacts over the nine-year period: 80% occur in bottomland hardwood and swamp hardwood habitats; 70% occur in the Cape Fear, Neuse, and Pasquotank River Basins; 85% occur in the Coastal Plain Physiographic Region; and, 65% occur in Highway Divisions 1, 2, and 3.

Findings

Total wetland mitigation needs related to these impacts are estimated as 18,000 acres, however only 2,200 acres were found to be ready now. Thus the net mitigation need is 15,800 acres.

Conclusion

In order to mitigate wetland impacts in step with their incidence during 1996-2004, the mitigation acreage obtained needs to increase to three times its current annual rate.

Recommendations

Budget, staffing, equipment, training, and consulting needs of the mitigation program should be evaluated relative to projected impacts. The computer model created for this study should be developed into a tool for research, planning, education, and training to facilitate impact identification and to enhance mitigation efforts. This study should be

updated annually after the adoption of the annual update of the TIP.

1. Methodology

This study examined the overlap of proposed construction projects and wetland areas. The main data source for highway projects was the 1996-2002 TIP. For wetlands, U.S. Fish and Wildlife Service (USFWS) NWI base maps were used. Other scientific data used in this study include: U.S. Geological Survey (USGS) physiographic region maps, North Carolina Center for Geographic Information and Analysis (CGIA) river basin and sub-basin maps, U.S. Department of Agriculture (USDA) hydrographic unit maps, and N.C. Department of Environment, Health, and Natural Resources (DEHNR) watershed maps.

The NWI maps have been digitized into a GIS format for all but 12% of North Carolina's land mass, and were considered adequate for this study, with two caveats. The first caveat is about the few NWI quadrangles that have not been mapped. Approximately two-thirds of the NWI quadrangles that are not mapped are of the Mountains, but few wetlands exist in that area. The other NWI quadrangles that are not mapped are of the Piedmont, where more wetlands exist, but are far fewer than in the Coastal Plain. The gap in NWI maps was thus assumed to be an insignificant drawback, the involved areas were simply not included in the study, and no attempt was made to use surrogates. The second caveat is that they systematically underrepresent wetland acreage by an estimated ratio of 2-to-1, so to correct for this map error, impact acres obtained from the NWI maps were doubled prior to analysis and reporting. In a few instances, it was possible to use actual field data instead.

For planning purposes, this study also assumed a ratio of 3.1 mitigation acres per impacted acre based on the experience of the Wetland Mitigation Section. Federal guidelines actually call for mitigation at various ratios: restoration at 2:1, creation at 3:1, enhancement at 4:1, and

preservation at 10:1.

Wetlands were categorized in two ways for this study. The first involved the full spectrum of detailed NWI technical classification codes. The other, summarized these numerous NWI classifications into a smaller number of habitat types using more common terminology.
The NWI classifications were used to facilitate detailed technical review during this study. The habitat types were intended to present a meaningful overview and summary for the report. This study thus identified wetland impacts by both NWI codes and habitat types, but only habitat types are included in this report.

Two categories of impacts for which mitigation is frequently not required are bridge projects eligible under the Categorical Exclusion (NWP-23) provisions, Nationwide Wetland Permits (NWP), and General Permits (GP). Wetland impacts of bridge projects were thus not included in this study, except for a few bridge projects that were treated as highway projects for analytical purposes, and for which actual impact mitigation estimates were used. NWPs exclude most impacts of less than one acre from mitigation requirements.

The main data source for highway projects was the 1996-2002 TIP. Other highway data sources used in this study include: roadway design corridor maps, project breakdown maps, a let/production list, a Planning Status Report, and post-TIP projections. In most cases, the maps are preliminary, and only show sketch plans, sites, or general areas of projects, and all are subject to change. Digitized county boundaries were obtained from USGS maps.

Wetland mitigation acres already applied to projects were not included in this study. Projects under construction or already completed were also not included. For each project examined, phased construction was not

considered, so all impacts were assigned to the earliest known construction year. This view is extremely meaningful for wetland mitigation purposes as the permitting agencies would like for all projects to be fully mitigated prior to starting construction.

Digitizing TIP projects was performed by ITRE staff. Lines in the TIP maps book were interpreted so that centerlines could be digitized for each project. For variable width and median-divided projects, the centerline was drawn symmetrically midway between the edges of the project. The inaccuracy inherent in using TIP map lines as project centerlines was not considered an unreasonable limitation because the project mission assumes TIP maps are accurate enough for this planning study.

For widening projects, the width attribute was centered on the roadway centerline in all cases, although widening can be done to variable distances from the centerline, or can occur on one side or the other instead of symmetrically. Impact widths meaningful for wetland mitigation purposes were then assigned to the line work, resulting in rectangular polygons, or "footprints." These coverages allow wetland impacts to be reported by various categories.

Cross-tabulations of the impact data were produced, and are shown in Tables 1 through 5. More complex tables were also produced, such as sorting first by years, then by river basins, then by habitat types, then by acreage; however, these tables are not included in this report. This study was quality controlled by comparing this data with expert knowledge and extensive field experience of involved staff persons.

It is important to note that project footprints of this study were developed solely for wetland mitigation purposes. These areas are not equal to pavement widths, rights-of-way, *etc.* Also, this study did not attempt to find impacts beyond the footprints, which might occur if project drainage were to alter hydrographic attributes in an area beyond a project. Some

projects were given various width attributes for different sections.

2. Analysis

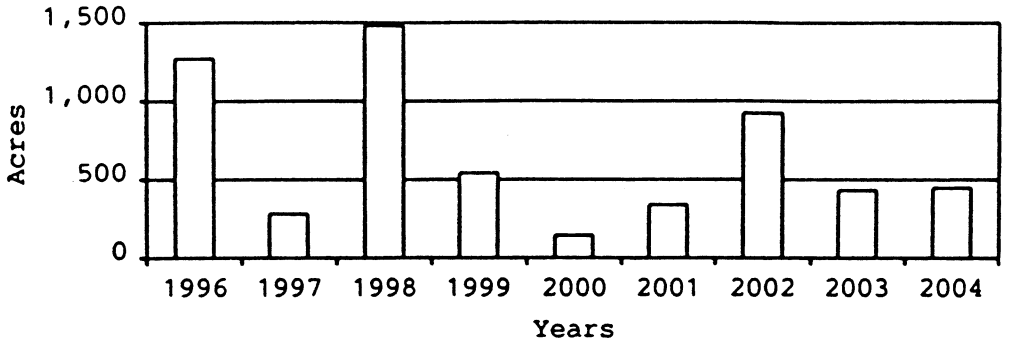
A. Impacts by Years

Wetland impact acres identified by this study for 1996-2004 are summarized by years as follows:

Table 1: Wetland Impact Acres by Year

Year	Impact Acres	Percentage
1996	1,271	22.1
1997	281	4.9
1998	1,481	25.7
1999	451	7.8
2000	143	2.5
2001	337	5.8
2002	925	16.0
2003	431	7.5
2004	444	7.7
Total	5,764	100.0

Graph 1: Wetland Impact Acres by Year



The above table and graph show that the impact acres estimated for the first three years of the period studied (1996-1998) subtotal to 3,033, or 53% of the total. Note: all impacts for each project are assigned to the year in which construction is planned to start.

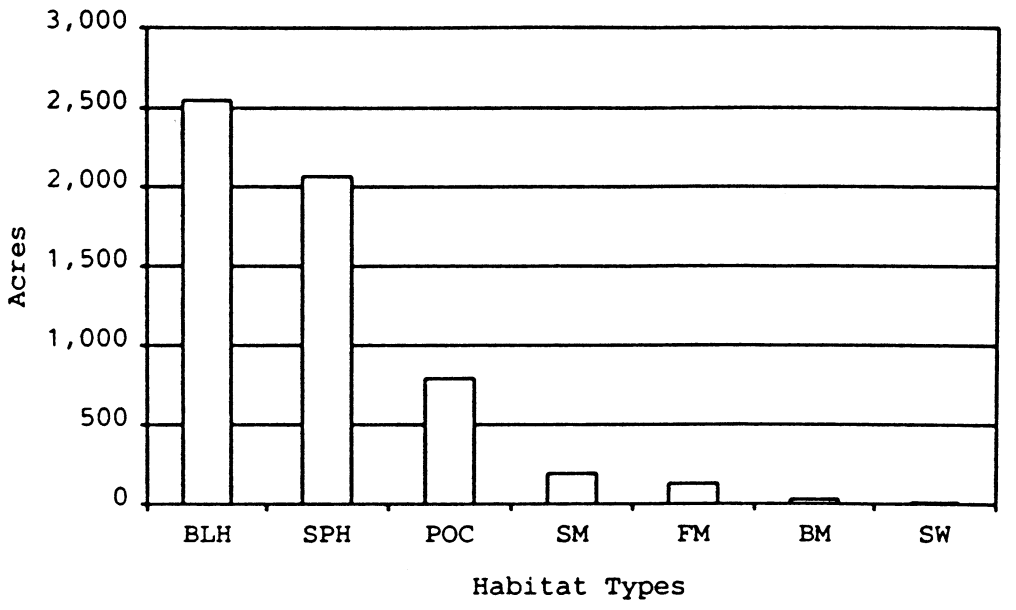
B. Impacts by Habitat Types

Wetland impacts identified by this study for 1996-2004 are summarized in descending amounts by habitat types as follows:

Table 2: Wetland Impact Acres by Habitat Type

Habitat Type	Impact Acres	Percentage
Bottomland Hardwoods (BLH)	2,545	44.1
Swamp Hardwoods (SPH)	2,067	35.9
Pocosin (POC)	791	13.7
Saltwater Marsh (SM)	194	3.4
Freshwater Marsh (FM)	129	2.2
Brackish Marsh (BM)	29	0.5
Surface Waters (SW)	9	0.2
Total	5,764	100.0

Graph 2: Wetland Impact Acres by Habitat Type



The above table and graph show that the impact acres estimated for bottomland hardwoods and for swamp hardwoods subtotal to 4,612 acres, or 80% of the total.

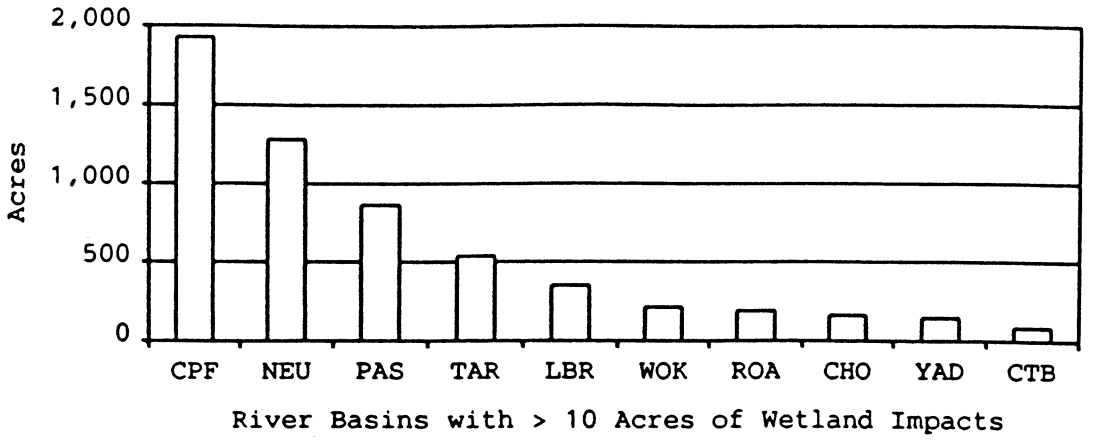
C. Impacts by River Basins

Wetland impact acres identified by this study for 1996-2004 are summarized in descending amounts by river basins as follows:

Table 3: Wetland Impact Acres by River Basin

River Basin	Impact Acres	Percentage
Cape Fear (CPF)	1,926	33.4
Neuse (NEU)	1,281	22.2
Pasquotank (PAS)	861	15.0
Tar-Pamlico (TAR)	535	9.3
Lumber (LBR)	349	6.1
White Oak (WOK)	215	3.7
Roanoke (ROA)	191	3.3
Chowan (CHO)	163	2.8
Yadkin (YAD)	146	2.5
Catawba (CTB)	86	1.5
French Broad (FRB)	6	0.1
New (NEW)	5	0.1
Broad (BRD)	0	0.0
Hiwassee (HIW)	0	0.0
Little Tennessee (LTN)	0	0.0
Savannah (SAV)	0	0.0
Watauga (WAT)	0	0.0
Total	5,764	100.0

Graph 3: Wetland Impact Acres by River Basin



The above table and graph show that the impact acres estimated for the Cape Fear, Neuse, and Pasquotank River Basins subtotal to 4,068 acres, or 71% of the total.

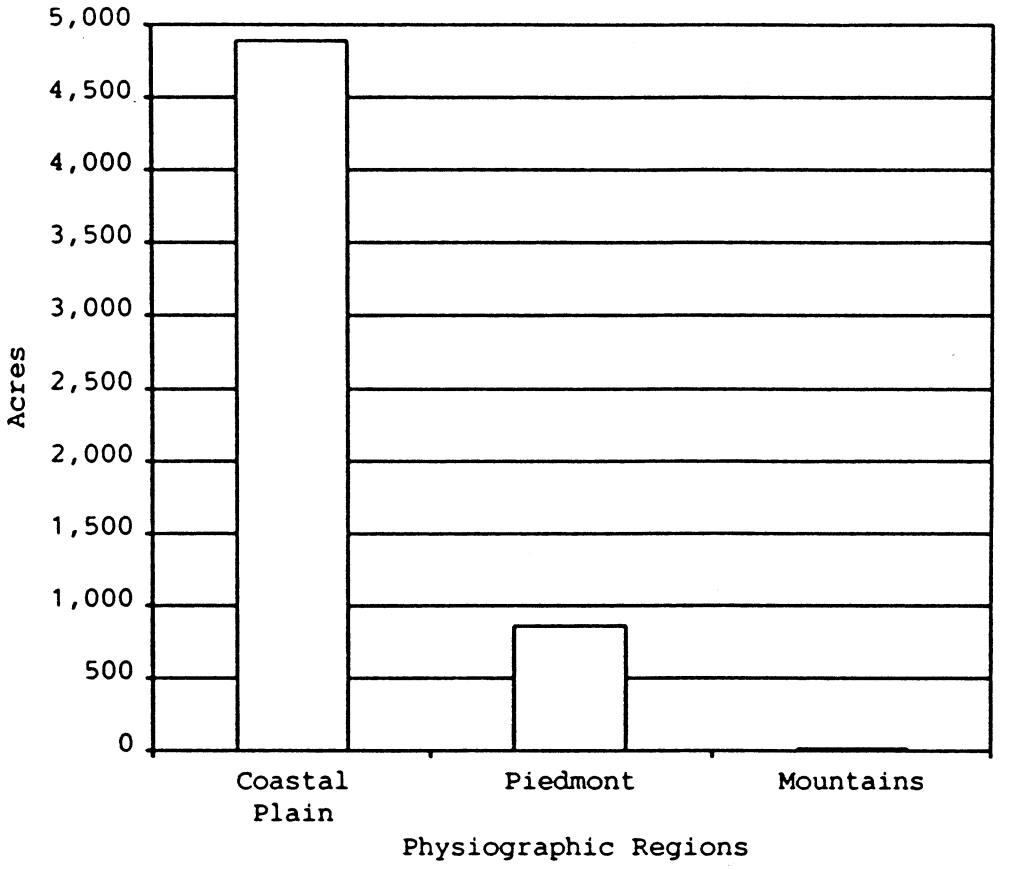
D. Impacts by Physiographic Regions

Wetland impact acres identified by this study for 1996-2004 are summarized in descending amounts by physiographic regions as follows:

Table 4: Wetland Impact Acres by Physiographic Region

Physiographic Region	Impact Acres	Percentage
Coastal Plain	4,889	84.8
Piedmont	861	15.0
Mountains	14	0.2
Totals	5,764	100.0

Graph 4: Wetland Impact Acres by Physiographic Region



Regarding the 4,889 impact acres estimated for the Coastal Plain Physiographic Region: 3,977 acres (81%) will occur in the Upper Coastal Plain, and 912 acres (19%) of impacts will occur in the Lower Coastal Plain.

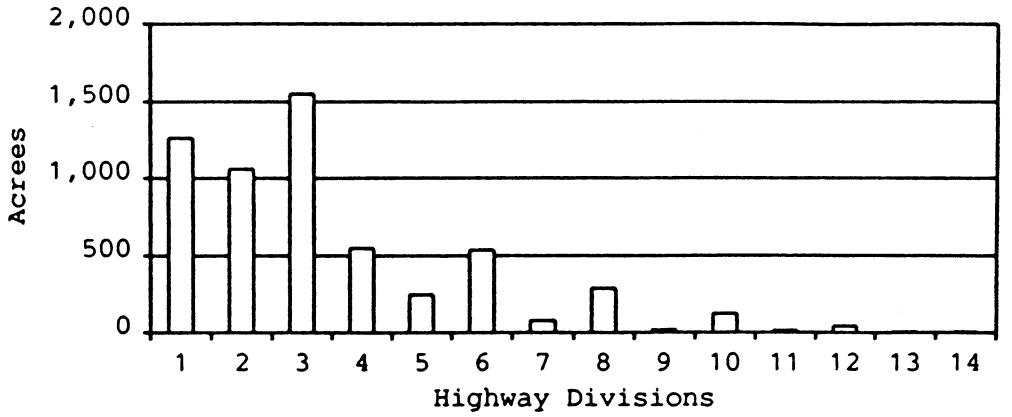
E. Impacts by Highway Divisions

Wetland impact acres identified by this study for 1996-2004 are summarized by Highway Divisions as follows:

Table 5: Wetland Impact Acres by Highway Division

Division	Impact Acres	Percentage
1	1,259	21.9
2	1,056	18.3
3	1,546	26.8
4	547	9.5
5	248	4.3
6	534	9.3
7	77	1.3
8	290	5.0
9	16	0.3
10	124	2.2
11	14	0.2
12	41	0.7
13	6	0.1
14	6	0.1
Total	5,764	100.0

Graph 5: Wetland Impact Acres by Highway Division



The above table and graph show that the impact acres estimated for Highway Divisions 1, 2, and 3 subtotal to 3,861 acres, or 67% of the total.

3. Findings

Total wetland mitigation needs related to the impacts identified in this study were found to be approximately 18,000 acres. An estimated 2,200 mitigation acres were found to be "currently available" to apply toward the total need. Most of these acres were found to be already fully available, though some were included by having been judged highly likely to become available in the next few months. After applying all mitigation acres currently available, a net need of 15,800 acres results for the nine-year period.

4. Conclusion

To address the 15,800 acres of net wetland mitigation need, the average annual production rate would have to increase to approximately 1,800 acres. This rate is nearly three times greater than the average annual mitigation production rate of the several years preceding this study.

5. Recommendations

The budget, personnel, equipment, training, and consulting needs of the mitigation program should be comprehensively evaluated as soon as possible in light of the impacts identified in this study. This evaluation would help ensure that needed wetland mitigation permits are obtained in a timely manner so as to avoid delaying construction. Also, the computer model created for this study should be developed further. The model should be made into a tool for research, planning, education, and training to facilitate impact identification and to enhance mitigation efforts. The study and computer model should be updated as soon as possible in readiness for the 1997-2003 TIP update.

Presentations of the findings of this study should be made in ARC/INFO and ARCVIEW/GIS for technical training of other DOT units such as Statewide Planning, Roadway Design, Roadside Environmental, Program Analysis, and Public Affairs. Presentations could also be made to regulatory agencies, such as: DEHNR, the N.C. Wildlife Resources Commission (WRC), the U.S. Environmental Protection Agency (EPA), the Federal Highway Administration (FHWA), USFWS, the U.S. Army Corps of Engineers (COE), and the National Marine Fisheries Service (NMFS). Universities, other organizations, and the public could also benefit from such presentations, and publication opportunities may be appropriate in various professional journals.

PRIVATE MITIGATION BANKING:

IMPLEMENTING AND SUSTAINING THE

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I. INTRODUCTION

Florida has created a sustainable program for preservation and restoration of environmentally sensitive lands through authorization of a private mitigation banking program. Developers no longer need to guarantee that on-site, wetlands and uplands mitigation, are available and successful for each and every proposed project. Mitigation banks are being implemented throughout the state to allow impactors the mitigation option of purchasing "mitigation credits" to offset unavoidable wetland and upland impacts caused by development projects.

Private mitigation banks improve currently degraded and impacted ecosystems through restoration and enhancement activities of both uplands and wetlands with perpetual maintenance. Large diverse ecosystems provide a sustainable development approach for both habitat and wildlife species. An ecosystem approach differs markedly from the historic postage stamp approach by utilizing a regional program of ecological biodiversity and ecosystem functionality.

II. HISTORY OF FLORIDA MITIGATION LAWS

1989 - Section 10 of the Federal Rivers and Harbors Act prohibits obstruction or alteration in navigable U.S. waters without a permit from the Army Corps of Engineers.

1972 - Section 404 of Federal Clean Water Act prohibits discharge of dredge and fill materials into all U.S. waters, including tributaries and wetlands, without a Corps of Engineers permit.

1972 - The Florida Water Resources Act establishes a statewide system of water management districts, with flood protection and water supply as primary concerns.

1975 - The state's Department of Environmental Regulation (now known as Department of Environmental Protection) initiates the first state wetland regulatory program, limited to dredge and fill activities in state waters and covering mostly permanently flooded wetlands connected to state waters, not isolated wetlands.

1977 - Management & Storage of Surface Waters (MSSW) regulations go into effect for portions of the St. Johns River Water Management District. The measure expanded district-wide in 1983. The primary concern is flood control at development sites. Late 1970's and early 1980's sees formulation of new rules by the management district regulating development of large isolated wetlands.

1984 - Mitigation is officially authorized after several years of formulation by the Warren S. Henderson Wetland Protection Act. It requires private development and public works projects to preserve, restore or create wetlands as compensation for wetlands destroyed by construction activities. The landmark statute also greatly expands the wetlands species list, cites habitat loss as a permitting concern for the first time and refines other criteria used in defining wetlands for permitting purposes. The changes bring vast amounts of previously "dry" development land under regulatory scrutiny.

1989 - Volusia County adopts the Wetland Alteration Ordinance, providing for local environmental review and approval of development. Its intent is to avoid or minimize wetland destruction through site planning or redesign with mitigation as a fallback option.

1991 - Flagler County adopts a wetlands ordinance with similar provisions and intent.

1994 - Mitigation banking rules adopted by the state as a complimentary program to traditional mitigation. Mitigation is acknowledged in statute language as a potentially better method to achieve "long-term, sustainable, regional ecological benefits." The law further encourages banks be near "areas of national, state or regional ecological significance," reflecting growing emphasis - in programs such as Florida Greenways - on regional and statewide ecological "corridors."

III. BACKGROUND

In 1993, the Florida legislature enacted the Environmental Reorganization Act of 1993. This legislation included the merger of the former Department of Environmental Regulation and the Department of Natural Resources into the Florida Department of Environmental Protection (FDEP). Additionally, this legislation included a provision titled "Mitigation and Mitigation Banking." *See* Fla. Stat. §373.4135. The

Florida legislature indicated that the intent of the mitigation banking statute was that adverse impacts of activities associated with the management and storage of surface waters (which includes the dredging and filling of wetlands) may be offset by creating regional mitigation areas, which will be identified in Florida as "mitigation banks." The statute directed the FDEP and the water management districts to enact regulations to implement the permitting and maintenance of mitigation banks. On February 2, 1994, the FDEP's regulations implementing mitigation banks were effective with the water management districts subsequently adopting similar administrative implementing regulations.

Mitigation banks provide a more reliable alternative to traditional mitigation options without the numerous uncertainties associated with traditional mitigation practices. Mitigation banks which utilize and implement ecosystem management principles for consolidated, large contiguous areas, are expected to provide greater assurances that the mitigation will yield long-term, sustainable, regional ecological benefits. Mitigation banking encourages the restoration and enhancement of degraded ecosystems by preserving uplands and wetlands as intact ecosystems, rather than the former practices which consisted of altering existing property to create wetlands on a limited, isolated basis. The intent is for mitigation banks to be located near or adjacent to areas of national, state or regional ecological significance, whenever possible. One chief purpose of the rule is to allow the use of mitigation banks when the adverse impacts of a certain project cannot be offset by onsite mitigation, or the mitigation bank would provide greater improvement in ecological value than on-site mitigation.

For example, consider the case where the destruction of several acres of wetlands is required for a certain development plan, and the remaining portions of the property are not appropriate to develop on-site man-made wetlands. The developer may choose to purchase mitigation bank credits to offset the adverse impacts of this project. The impactor could "purchase" the mitigation bank credits in an amount determined by the permitting agency, and would then avoid having to plan and construct on-

site mitigation as well as the costs associated with the loss of potentially developable land. The impactor also receives an irrevocable transfer to the banker for the long term monitoring and maintenance associated with mitigation. The FDEP and the water management districts also expect that a combination of both on-site mitigation and participation in a mitigation bank may be appropriate to offset specific impacts of a project.

Currently, impactors attempt to mitigate a known functioning habitat with an unproven plan which must be implemented at the expense of the impactor with a documented success rate of only ten percent (10%). On the other hand, mitigation banks use restoration and enhancement activities and are required to demonstrate ecological success which is assured through first, a long term maintenance trust fund, and second, the credit release schedule whereby credits are withheld from sale in the open market until project success is demonstrated. As such, greater mitigation certainty is guaranteed by the banking process and governing regulations.

IV. SUMMARY OF THE BANK PROGRAM

At the heart of the Mitigation Banking Rule is the establishment of the "banking system." When a mitigation banking application is complete, the permitting agency will assign a certain number of "mitigation credits" for the mitigation bank. Each mitigation credit is a unit of measure which will be representative of the increased "ecological value resulting from a successfully created acre of wetland." The credits, as authorized by the release schedule, are then available for sale at market conditions to impactors. In order to determine the number of mitigation credits which will be allocated to a mitigation bank, it will consider the following factors:

(a) Whether "target" hydrologic standards can be met;

(b) Whether the bank's management promotes natural ecological conditions;

(c) Whether the mitigation bank is adjacent to or near areas of national, state or regional ecological significance;

(d) Both the quality and quantity of wetland or upland restoration, enhancement, preservation or creation;

(e) The relationship between wetlands and uplands and the bank;

(f) Habitat provided for fish and wildlife in the bank, especially considering threatened and endangered species;

(g) Whether the mitigation bank land is already protected by existing regulations or restrictions;

(h) The extent that the mitigation bank lands would be adversely affected if they were not preserved; and

(i) Any special designation or classification of the affected waters and lands in the mitigation bank.

Once the number of mitigation credits are established by the agency, the credits will be available for withdrawal from the bank. These mitigation credits may be transferred, sold or otherwise used by the mitigation banker. The mitigation bank is similar to a financial bank. The mitigation bank permit provides for a ledger listing the total number of mitigation credits issued to the bank. The mitigation credits may then be sold in whole or in part at the mitigation banker's discretion and then deducted from the ledger. A similar ledger is maintained for the permitting agency. Credits are withdrawn from the mitigation bank through a minor permit modification to the mitigation bank permit.

To develop a mitigation bank, the banker must apply for, and be issued, a mitigation bank permit. All mitigation banks must meet the following criteria:

(a) Improve ecological conditions of the regional watershed;

(b) Provide viable and sustainable ecological and hydrological functions for the proposed mitigation service area;

(c) Be effectively managed (long-term);

(d) Not destroy areas with high ecological value;

(e) Achieve mitigation success; and

(f) Be adjacent to lands which will not adversely affect the mitigation bank.

A mitigation bank may be implemented in phases. Each phase, however, must independently meet the criteria discussed above.

To apply for a mitigation bank permit, the applicant must provide certain information. The mitigation bank permit application also includes all the information required for the management and storage of surface waters (MSSW permit application). Information concerning the mitigation bank which must be included in the mitigation bank application includes:

(a) A description of the location of the proposed mitigation bank, including detailed maps, aerial photographs and a legal description;

(b) A description of the ecological significance of the mitigation bank in regard to the regional watershed in which the bank property is located;

(c) A description and assessment of current mitigation bank site conditions, including soils, topographic and hydrological maps, current hydrological conditions, vegetation maps, and disclosures of other material facts that would affect the

mitigation bank;

(d) A mitigation plan, including construction drawings, vegetation planting schemes, and a long-term management plan and a monitoring plan (which must demonstrate the success of the mitigation project);

(e) An assessment of the improvements and ecological value that are anticipated from the proposed mitigation actions, including expectations of site conditions after the bank is implemented, wildlife habitat comparisons and expected ecological benefits to the regional watershed;

(f) Evidence of the banker's appropriate legal or equitable interest in the property;

(g) Documentation of financial responsibility; and

(h) Any other information which would assist in evaluation of whether the mitigation bank meets the required criteria.

After a mitigation permit is issued, but before any mitigation credits may be transferred from the bank, the mitigation bank property must be conveyed either by a fee simple interest to the state or by a conservation easement conveyed to the FDEP and/or the appropriate water management district.

The mitigation bank permit requires that the banker provides evidence of financial responsibility to "provide reasonable assurances" that the mitigation bank has the financial where-with-all to meet all of the requirements and permit conditions. All non-governmental mitigation bankers must provide proof of financial responsibility for both the construction and implementation phase of the bank, as well as the long-term management of the bank. For the construction and implementation phases, financial responsibility may be established by guaranty bonds,

performance bonds, insurance certificates, irrevocable letters of credit, trust fund agreements or securities. For the long-term management of the mitigation bank, the banker must establish a trust fund agreement. All financial responsibility mechanisms must be effective at least sixty (60) days prior to the beginning of that phase of the mitigation bank construction or management. The amounts of financial responsibility must be adjusted every two (2) years. When governmental entities operate mitigation banks, they must supply the same evidence of financial responsibility as is required of non-governmental entities.

The agencies have developed a concept of the "mitigation service area" to define certain lands which may apply for credits from a particular mitigation bank. A mitigation service area is the geographic area within which mitigation credits from a mitigation bank may be purchased or otherwise used to offset adverse wetlands impacts. A mitigation service area will be established for every mitigation bank which is permitted by the agencies. The size of the mitigation service area will depend upon the quality of the mitigation bank and its ability to offset adverse impacts from other properties in the mitigation service area.

If a certain project is not located completely within a mitigation service area, the project may still be able to use a mitigation bank if it meets one of the following three criteria:

(a) The project is located partially within the mitigation service area;

(b) The project is a "linear project," such as a road, transmission line or pipe line; and

(c) The project's total adverse impacts are less than one-half acre in size.

If the adverse impacts of the project are located within a regional watershed, even if they do not fall within the mitigation service area, the

mitigation credit requirement will be the same as that specified for mitigation on the project site. However, if the adverse impacts are located outside of the regional watershed, the mitigation credit requirements may be higher than that specified for mitigation on the project on-site. There is one exception to this rule; for "linear projects," when the specific impact being offset falls within the mitigation service area for the mitigation bank which is going to be used, the mitigation requirement will be the same as that of on-site mitigation.

As discussed earlier, an agency permit must be issued before the mitigation bank can begin operation. The agency may issue two types of authorizations for mitigation banks: the mitigation bank permit and the mitigation bank conceptual approval. The mitigation bank permit will authorize the construction, implementation and operation of the mitigation bank. The bank permit sets forth all the rights and responsibilities of both the banker and the agency for each particular mitigation bank. The mitigation bank permit will include: a description of the mitigation service area, the maximum number of mitigation credits available, the criteria by which the success of the mitigation bank will be evaluated, the financial responsibility mechanisms which will be used by the banker, information concerning the conveyance of a conservation easement or a fee interest in the property, the mitigation credit ledger, an implementation schedule for the bank and the long-term management requirements for the mitigation bank.

A mitigation bank permit will be perpetual, unless it is revoked or modified. There are two exceptions to the perpetual permit, however. The bank permit will only have a term of five (5) years if the mitigation banker has not recorded a conservation easement or conveyed a fee simple interest in the property to the government. In cases where no property interest is required to be recorded, the mitigation bank permit will expire if no construction has begun.

The agency may also issue a "conceptual approval." The conceptual approval will provide an estimate of the legal and financial requirements

necessary for the mitigation bank and will discuss the information necessary to evaluate the bank permit and potential mitigation credits which could be awarded under the permit. A conceptual approval, however, does not authorize any construction of a mitigation bank or any use or withdrawal of mitigation credits. The level of detail provided in the conceptual approval will depend on the information submitted by the applicant and the agency's review of that information. A mitigation bank conceptual approval will be valid for five (5) years from the date of issuance. The conceptual approval process should provide valuable assistance to potential mitigation bankers who wish to have the viability of their project assessed by the agency prior to the expensive and time-consuming permit application process.

A mitigation bank permit may be surrendered by a written request to the agency. If the property interest has been conveyed to a governmental entity, the agency will convey the property back to the grantor. A mitigation bank permit may also be transferred, as long as there are reasonable assurances that the agency's requirements will be met by the transferee.

V. WATER MANAGEMENT DISTRICT REVIEW COMMISSION[1]

Created by Chapter 94-270, Laws of Florida (1994), to "[p]erform a comprehensive review of Florida's system of regional water management," the Water Management District Review Commission ("Commission") embarked on a statewide journey to educate and enlighten its members on the past and present state of water resource management in Florida. In the twenty-something years since creation of the five regional water management districts in 1972, the statutory responsibilities of those

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[1] The text of part V, Water Management District Review Commission, was originally published in the December 29, 1995, *Bridge Over Troubled Water: Recommendations of the Water Management District Review Commission,"* as edited.

agencies of the state have expanded to include not only flood protection and surface water regulation, but also environmental resource permitting, water quantity and quality protection and enhancement, and the acquisition and management of ecologically sensitive lands. The Commission's intensive 15-month study included examination of the myriad aspects of districts' budgeting and operational programs, as well as the constitutional and statutory authority of districts to manage regional water resources.

To investigate the various regional concerns of Florida's citizenry, the Commission conducted seventeen (17) public hearings across the state to receive public comment regarding the specific programs and operations administered by water management districts.[2] Meeting agenda included detailed overviews of area-wide district responsibilities and functions, Commission subcommittee meetings, open-mike public hearings, and formal presentations by scientists, engineers and scholars, as well as representatives of federal, state and local governments, regulated entities, environmental concerns, and agricultural and industrial interests.

The following subcommittees were established within the Commission to conduct detailed reviews of the itemized topics, as described in the enabling legislation:

Financial Structure and Budgeting: District operating costs; funding mechanisms available to the districts to carry out their responsibilities; ways to improve district financial accountability; and the need to revise district budget development and adoption procedures, as well as ad valorem taxing authority and notice requirements of Chapters 200 and 373, Fla. Stat.

District Responsibilities and Operations: Legal responsibilities

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[2] Commission meetings were convened in Tallahassee, Tampa, Orlando, Jacksonville, Lecanto, Ft. Lauderdale, Live Oak, Clewiston, Pensacola and Sarasota.

assigned to water management districts, whether those duties should be modified, and ways to improve district operations and programmatic accountability.

Land Acquisition, Planning and Management: Ways to improve planning and management of district-owned lands, including the potential for reorganizing and integrating the programs and responsibilities of the districts, regional planning councils, the FDEP and the Department Community Affairs, federal and local governments, and not-for-profit entities.

The subcommittees addressed issues within the particular subject areas assigned and made preliminary recommendations that were considered by the full Commission during its final deliberative meetings. As a whole, the Commission reviewed district governance issues, such as whether to appoint or elect governing board members, or perhaps select candidates in a manner similar to that of the judicial nominating commission or the Public Service Commission Nominating Council. In addition, Commission members studied the feasibility of creating new committees, subcommittees or joint committees of the Senate and House of Representatives to provide active legislative oversight of water resource management in Florida.

Chapter 94-270, Laws of Florida,[3] directed the Commission to report its recommendations to the Governor, President of the Senate and Speaker of the House of Representatives by January 1, 1996. At the discretion of Commissioners, this report has been designed to provide the Governor and Legislature with specific, detailed recommendations for improving water management districts. Historical perspectives on state water resource regulation, descriptions of the various proposals discussed, and explanations of Commission members' reasoning have been intentionally omitted from the report to facilitate efficient review. By February, 1996,

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[3] As amended by Chapter 95-171, Laws of Florida (1995).

the Commission will draft legislation to implement its recommendations for consideration during the 1996 regular session of the Florida Legislature, following which a comprehensive report detailing the Commission's work and resulting legislative action will be published.

The Commission's recommendations resulted from consideration of testimony given by the general public during statewide hearings, written comments to the Commission from Florida citizens and interested parties, information received from state agencies and the water management districts, the personal expertise and experience of individual Commission members, and extensive discussions among the participants in proceedings of the subcommittees and full Commission. Unless otherwise indicated, the recommendations were adopted by unanimous agreement of the Commission.

In recognition of the precatory nature of Commission's report, the phrasing of some recommendations was edited to omit words of mandatory direction, *e.g.*, "shall" or "must." Where the language adopted by Commissioners was imperative, however, the words have been included in brackets [ ] to clarify the members' collective intent. Proposed statutory revisions include appropriately phrased directives.

The following six (6) items represent the Commissions' recommendations regarding mitigation banking:

1. Mitigation Banking

A. Establishment of Mitigation Banks.[4] The Legislature should require the water management districts and the FDEP to cooperate with private and nonprofit entities, local governments and other state agencies to establish mitigation banks and projects on district or state lands where possible, and on appropriate private lands voluntarily designated for mitigation banks or projects.

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[4] *See* Fla. Stat. §373.4135 (1995).

B. Identification of Mitigation Bank Lands. To provide advance notice of areas within each watershed or ecosystem management area that could be available to offset the loss of wetland function, the Legislature should direct the water management districts and FDEP to coordinate with appropriate local governments to identify all public lands within each watershed or ecosystem management area that may be eligible for mitigation projects or mitigation banks to be established by public, private, or not-for-profit entities. A similar determination of eligibility could be performed for any private land upon request of the property owner. The identification process should be accomplished by March 1, 1997.[5]

C. Annual Report of Eligible Mitigation Bank Lands. The water management districts and the FDEP should [shall] include in the annual state land acquisition programs reports published and submitted to the Governor, President of the Senate and Speaker of the House of Representatives, a list of acquired lands (or interests therein) eligible for district-endorsed mitigation projects or mitigation banks.[6]

D. Expedited Consideration of Mitigation Alternatives. The water management districts and the FDEP should [shall] establish procedures to assist permit applicants in addressing mitigation requirements for proposed wetland impacts. The procedures should [shall] expedite the agency's consideration of mitigation alternatives and increase the level of certainty for permit applicants.

E. Cost of Mitigation Activity. Where a water management district or the FDEP operates a mitigation bank or accepts cash contributions for district-endorsed mitigation projects, the amount charged

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[5] This recommendation was passed by a 16-to-2 vote of Commission members.

[6] The preliminary proposal adopted by the Commission included a January 15 deadline for submission of the list of mitigation-eligible lands. The date was omitted from the final recommendation, however, because submission deadlines were *not* included in Recommendations #58 and #66, which described other elements of the annual report on land acquisition programs.

to permit applicants who utilize such bank or project should [shall] be no less than the full cost of the mitigation bank activity or project to the district or Department. Costs associated with mitigation bank activities or project should [shall] include, but not be limited to, all direct and indirect expenses for land acquisition, land management, restoration and enhancement activities.

F. Mitigation Fund Accounts. The Legislature should direct the water management districts and the FDEP to establish separate accounts for mitigation bank or project funds received from permit applicants and utilize those funds for acquisition and restoration projects that will offset the impacts mitigated.

VI. FUTURE OF MITIGATION BANKING

1. General

The current ecosystem management initiative of the FDEP is an undertaking to analyze the existing environmental regulatory program and balance the natural systems protection agenda with sustainable development. Historically, Florida's extensive and complex set of environmental regulations designed to address pollution of the air and water have conflicted with the private sector agenda to realize fair market value or greater for private property. In addition, adversarial financial and legal relationships have developed between the government, the regulated entities, and the public. Ecosystem management provides a bipartisan methodology to address the identified conflicts and shortcomings of the existing regulatory program.

Ecosystem management embodies a holistic approach to environmental permitting programs. Upland and wetland impacts are evaluated in the permit process. Ecosystem management is designed to integrate upland and wetland ecologies through a watershed basis or other technically defensible management plan. The management plan also includes

identification, restoration and preservation of critical lands, greenways and wildlife corridors as discussed above. By injecting balance and common sense between environmental and development objectives, ecosystem management serves as the framework for recognition of the interrelationship of Florida's wetland and upland habitats.

Private mitigation banking will be a strong partner in the implementation of the ecosystem management program. Private mitigation banks must incorporate ecosystem management in the design of each of its regionally significant banks, including upland and wetland components which increase the biodiversity and sustainability of the overall ecosystem. Further, this sustainable ecosystem management approach allows for significant benefit to the overall environment, including habitat and aquifer protection.

Private mitigation projects, due to state and federal budget restraints, will augment, and in some instances replace, existing government acquisition and management programs. The private mitigation banker through a fully funded long term maintenance trust as required by regulation will serve as perpetual steward of the upland and wetland ecosystem. Further, through the purchase of credits at a private mitigation bank, the marginal benefit for the funds expended on mitigation historically will now more significantly equal or exceed the ecological impact for which the mitigation is required for a specific project. Finally, through incorporating government involvement in the regulation of private mitigation banks, a public/private partnership is created to cooperatively implement ecosystem management while providing requisite economic compensation to the private corporation charged with acquiring, restoring and protecting natural areas for benefit of the public.

2. Ecosystem Credits

As the private mitigation bank program evolves, a significant concern focuses upon the methodology to determine the number of credits available for sale to third parties from a private mitigation bank. More specifically,

upland restoration and preservation which must be included as part of a sustainable ecosystem approach and is not subject to a consensus methodology for determination of the number of upland credits and also a release schedule for upland credits based on success criteria. If the holistic approach of ecosystem management is viewed as the end goal over the next several years by the Florida legislature and the implementing regulatory agencies, it is proposed that the existing mitigation credit terminology be amended to adopt the concept of ecosystem credits.

Under the auspices of the regulatory agencies, a proposed private mitigation bank would be evaluated in terms of the upland and wetland activities which in combination would recognize the ecosystem benefit as a result of incorporating sustainable ecosystem management concepts. The determination of ecosystem credits may be based upon the increase in functionality of the ecosystem recognizing the enhanced biodiversity attributable to restoration and preservation of both upland and wetland habitat.

The incorporation of the concept of ecosystem credits also would address the federal concern regarding upland restoration as compensation for wetland impacts. Further, ecosystem credits would minimize to some extent the existing dichotomy in the evaluation of upland and wetland habitat by requiring evaluation of the ecosystem in its entirety.

3. Land Acquisition Initiative

It is recognized that current public funding levels are insufficient to acquire and manage all of the lands identified and sought by governmental entities for restoration and preservation. In the current political climate, it is not anticipated that a significant source of additional funding will become available during this decade. As such, private mitigation banking affords an excellent opportunity for a significant public/private partnership to increase the rate at which environmentally sensitive lands are acquired, restored and preserved.

Private mitigation banks must be located in environmentally sensitive areas as determined by federal, state or local governments. A significant factor in the success of a mitigation bank is the proximity of the intended bank to state or regional parks or public lands. Regionally significant mitigation banks must have natural ecosystem characteristics which have been altered. It is axiomatic that the public agencies would like to expand the size of current public holdings.

One approach is to set aside a portion of gross revenues in a special trust fund for acquisition purposes. Property in the proximity of the mitigation bank or other environmentally sensitive properties which the government would like to acquire but for which funding is not available will be acquired by the private banker as credits are sold to third parties. These acquired lands can then be restored and maintained as part of the regional ecosystem.

4. Conclusion

Private mitigation banking establishes a financial and technical vehicle to accelerate environmental protection and balance private property rights. As the program evolves, the environmental benefit to the public will increase without significant tax burden imposed on the government. The private sector is poised to become a full fledged partner with the government as private mitigation banking progresses through the end of the decade and beyond.

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WETENUFF -- A SIMPLE SPREADSHEET TOOL FOR THE ASSESSMENT OF HYDROLOGIC CONDITIONS OF SMALL, ISOLATED WETLANDS IMPACTED BY DEVELOPMENT ACTIVITIES

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ABSTRACT

Proposed land development activities frequently occur in areas that include jurisdictional wetlands. Wetland regulators may require that wetland areas be excluded from the area to be disturbed by construction activities. The applicant may be required to demonstrate that the proposed development activities will not alter the hydrologic conditions to the point where the wetlands may not remain viable. In other situations, sufficient amounts of reliable hydrologic records may not be available, and it may not be known if the hydrologic conditions at the site meet the requirements set forth in Florida's unified wetland delineation methodology.

Where wetlands would be preserved and enhanced to offset the impacts, the applicant typically proposed directing as much runoff as practical from the proposed development activity to the wetland area. The WETENUFF procedure was developed as a spreadsheet tool to estimate if the post-development hydrologic conditions will meet the jurisdictional definition.

The WETENUFF procedure utilizes a simplified water budget approach on a daily time step. Factors included in the procedure are rainfall, initial abstraction, average evapotranspiration, soil storage, exfiltration, and surface water control elevation. The resulting average daily water surface

elevations are compared to ground surface elevations to estimate inundation and saturation conditions.

1. INTRODUCTION

Proposed developments that impact jurisdictional wetlands in Broward County must obtain an Environmental Resource License from the Biological Resources Division (BRD) of the Department of Natural Resource Protection (DNRP)(Broward County Board of County Commissioners, 1994). When preservation is warranted, license applicants or their consultants frequently reach consensus with the wetlands staff of the BRD that the ground surface of the areas determined to be jurisdictional wetlands will not be disturbed by the development activities.

Although the ground surface of the wetlands may not be disturbed, the hydrologic conditions of the wetland may be altered by excessive drainage of the development surrounding the wetlands. In the past, the applicant would attempt to mitigate the hydrologic impacts by routing as much storm drainage as practical from the development to the wetland. Where such negative alteration of the hydrologic conditions is likely to occur, DNRP now requires the applicant to provide evidence that the proposed development will still support or improve the hydrologic conditions of the wetland.

Other situations arise where the applicant and DNRP disagree as to whether or not existing wetlands are jurisdictional wetlands. DEP Rule 62-340.550 (Florida Department of Environmental Protection, 1994) indicates that when sufficient monitoring data are not available to determine if the area in dispute is inundated for at least seven consecutive days or is saturated for at least twenty consecutive days during conditions that represent long-term hydrologic conditions, a numerical model may be used to assist in the determination of jurisdictional wetlands.

BRD contacted the Water Resources Division (WRD) of DNRP to collaborate on the development of a numerical method to estimate the hydrologic conditions for relatively small wetland areas.

2. Development of Procedure

The estimation of precise surface and ground water surface elevations is a very complex science in south Florida. Estimates are complicated by high and variable transmissivity values in the surficial aquifer and the rapid reaction of ground water elevations to changes in surface water elevations and vice versa. Three dimensional integrated surface and ground water programs exist but their utilization in the assessment of relatively small developments is not practical at the present time.

It was believed that a relatively simple spreadsheet procedure could be developed that would account for the major water budget factors and provide the accuracy required for most of the situations in question (Maidment, 1993). The development and acceptance of the utilization of the procedure by DNRP would produce a standard that would facilitate the staff's review of the application package. The applicant would still have the flexibility to utilize a more detailed and complex approach where conditions warranted a more accurate answer.

2.1 Assumptions

In order to fulfill the intention to develop a simple procedure, a number of assumptions had to be made. The key assumptions of the procedure are indicated as follows. A sketch of the inflow and outflow mechanisms is presented on Figure 1. Figure 2 shows the change in water surface elevation based on inflow and outflow and whether or not the water surface elevation crosses the wetland ground surface.

- A simulation period of one year which encompasses both wet and dry seasons in south Florida is sufficient.

- Inflow and outflow mechanisms that determine the wetland water surface elevation are functions of rainfall, evapotranspiration, and exfiltration from the wetland area plus runoff from adjoining areas. A minimum or maximum wetland water surface elevation controls where the computed water surface elevation exceeds the logical range of values.

- Long-term hydrologic conditions are represented by rainfall for a "typical" year that approximates average monthly and yearly rainfall amounts.

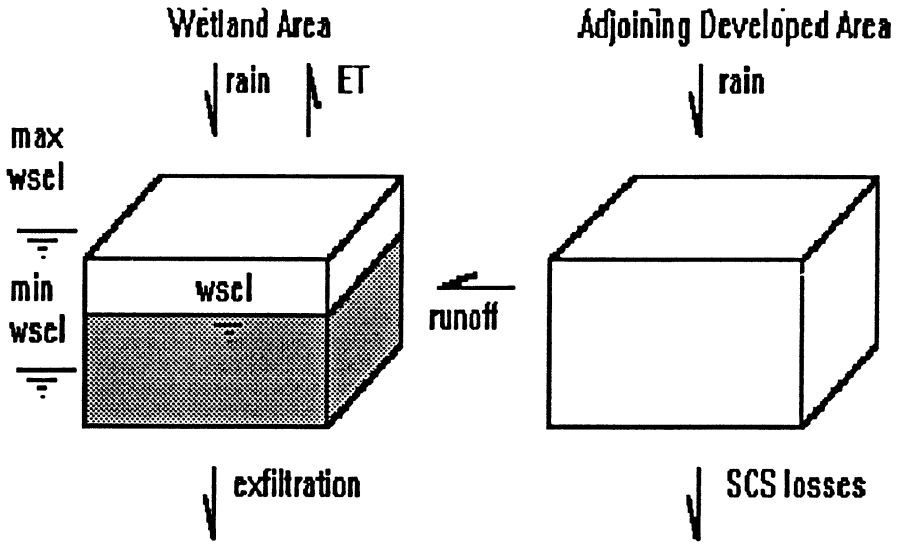

Figure 1. Wetland Inflows/Outflows

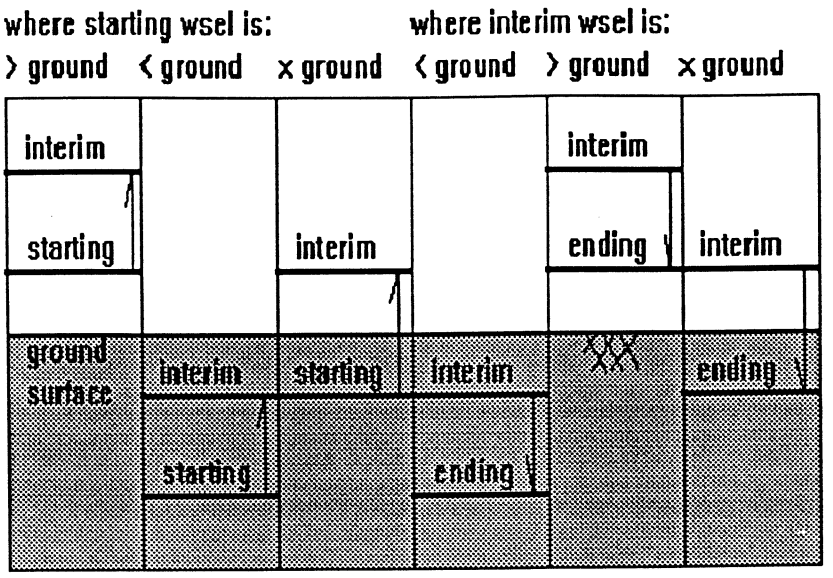

Figure 2. Water Surface Elevation Changes

- A one-day time step facilitates the approximation of the wetland water surface elevations without being data intensive or producing lengthy output.

- Wetland ground surface and water surface elevations may each be represented by one value throughout the site.

- Broward County may be divided into rainfall zones based on the location of rain gages maintained by Broward County (Broward County Office of Environmental Services, 1991).

- The "typical" years for rainfall in rainfall zones 1A, 2A, CS, and HW are 1987, 1990, 1990, and 1990, respectively.

- Wetland evapotranspiration is constant throughout the year and equals 119 centimeters per year.

- The exfiltration rate through the wetland bottom and sides is constant with head and time.

- Runoff from adjoining upland areas can be routed into the wetland area to supplement hydrologic conditions of the wetland.

- Control elevation and/or ground water elevation for the adjoining area may be represented by one value.

- The adjoining area soil storage, S, is based on SFWMD methodology (South Florida Water Management District, 1994).

- Adjoining area soil storage occurs over pervious area only.

- Adjoining area runoff is based on the SCS runoff equation (Soil Conservation Service, 1972).

- The wetland ground water elevation cannot fall below a minimum elevation.

- The wetland water surface elevation cannot exceed a maximum elevation. This simplification eliminates the need to reservoir route ponded water through the control structure. Many surface water control structures have wide weirs that can discharge a significant volume of water with a minor increase in head on the weir.

- Ponding occurs when the wetland water surface elevation exceeds the wetland ground surface elevation. The ponding area is constant.

- Saturation occurs in soils with a permeability less than 15 centimeters per hour if the wetland water surface elevation is less than 30 centimeters below the wetland ground surface elevation. Saturation occurs in soils with a permeability greater than 15 centimeters per hour

if the wetland water surface elevation is less than 15 centimeters below the wetland ground surface elevation.

2.2 Algorithms

Algorithms were developed based on the above assumptions to enable the computation of daily wetland water surface elevations. The principal algorithms utilized in the procedure are as follows:

Initial Abstraction, Ia = 0.2*S

Adjoining Runoff Depth, ARD = (P-0.2*S)^2/(P+0.8*S)

Adjoining Runoff Volume, ARV = ARD*Total Adjoining Area

Wetland Runoff Volume, WRV = P*Wetland Area

Total Inflow Volume, TIV = ARV+WRV

Average Loss Through Wetland Bottom = Wetland Area Exfiltration Rate/Wetland Area

Total Wetland Losses, TWL = Average Wetland ET+Average Loss Through Wetland Bottom

Change in Wetland Volume = TIV+TWL

Interim Wetland WSEL 1, IWW1 = Starting Wetland WSEL+(TIV/Wetland Area),
where WSEL>Wetland Ground Surface Elevation, and
= Starting Wetland WSEL+TIV/Wetland Area/Wetland Soil Water Storage Rate,
where WSEL<Wetland Ground Surface Elevation

Interim Wetland WSEL 2, IWW2 = Starting Wetland WSEL+(TWL/Wetland Area), where WSEL>Wetland Ground Surface Elevation, and = Starting Wetland WSEL+TWL/Wetland Area/Wetland Soil Water Storage Rate, where WSEL<Wetland Ground Surface Elevation; also, IWW2 = Minimum Wetland Elevation if IWW2<Minimum Wetland Elevation

Ending Wetland WSEL, EWW = Maximum Wetland Elevation, if IWW2>Maximum Wetland Elevation

Ponded? = YES, if EWW>Wetland Ground Surface Elevation; else = NO

Saturated? = NO, if EWW<Saturation Definition Elevation; else = YES

3. Application of Procedure

The user can generate daily wetland water surface elevations for a one year period by inputting a total of thirteen values in the "Parameters Used in Computations" section of the spreadsheet. The spreadsheet cells for which variables must be entered are shaded in the "Parameters" section and in the description below. The user may also enter the name of the project, the consultant firm name, the computation date, the consultant contact, the consultant telephone number, and special assumptions or other comments in the remainder of the labeled shaded cells under Assumption 10. The user should not change the values or formulas in any other cells without discussing the proposed changes with DNRP and/or noting the changes in the extra assumption cells on the spreadsheet. Additional information about the parameters is shown in Table 1.

Table 1. WETENUFF Parameters

Number	Name	Description (units)
1	wetland area	area of wetland (ha)
2	rainfall zone	BCOES rainfall publication
3	wetland ET	other than default (cm/yr)
4	wetland ground elev.	typ. ground surface (m)
5	wetland soil stor rate	computed by spreadsheet
6	wetland area exf. rate	thru bottom/sides (cm/day)
7	adj. pervious area	veg./bare areas that drain into wet. area (ha)
8	adj. impervious area	roof/road/parking/lake areas that drain to wet. area (ha)
9	total adj. area	computed by spreadsheet
10	avg. adj. ground elev.	representative adjoining ground elev. (m)
11	adj. control/GW elev.	lower elevation of soil storage area (m)
12	adj. area depth to GW	computed by spreadsheet
13	adj. compact soil stor	computed by spreadsheet
14	adjoining soil storage	computed by spreadsheet
15	minimum wetland depth	lowest expected wetland elevation (m)
16	maximum wetland depth	highest expected wetland elevation (m)
17	assumed wetland WSEL	initial wetland WSEL on January 1 (m)
18	saturation depth	depth of water needed for saturation (m)

Most of the site-specific shaded values are simple to estimate. The most difficult parameters to estimate values for are No. 6, the wetland area exfiltration rate; No. 15, minimum wetland depth factor; and No. 16, maximum wetland depth factor. The wetland area exfiltration value may best be estimated by installing monitoring wells and recording the water levels and times of reading on the recession side of storm events. Time

may limit the amount of data that can be obtained from the monitoring well. However, a limited amount of site specific data is almost always better than no site specific data.

Considerable judgment is required to estimate the minimum and maximum wetland depth factors. Setting the minimum elevation too low will not accurately represent the situation where the wetland water surface elevation is augmented by ground water flow from the surrounding area. Setting the minimum elevation too high may indicate that the wetland area is continuously saturated, especially in the cases where the saturation definition depth is 30 centimeters. Setting the maximum elevation too low will indicate that too much of the ponded water is discharged into the receiving surface water. Setting the maximum elevation too high will not accurately reflect the water lost through control structure discharges to the receiving surface water.

After entering all of the variables required in the "Parameters" section, the user should review the data presented in the spreadsheet for reasonableness and adjust the variables using professional judgment. Particular care should be given to the need to modify the wetland area exfiltration rate, the minimum wetland depth factor, and the maximum wetland depth factor. The starting wetland WSEL in the "Parameters" section should be revised (if necessary) to equal the ending wetland WSEL for December 31. The starting wetland WSEL may need to be adjusted several times until the ending wetland WSEL on December 31 equals the starting wetland WSEL. The columns with the titles "Ponded?" and "Saturated?" may be scanned to determine if ponding and/or saturation is indicated on a frequency that would support the determination of jurisdictional wetlands after the user has reviewed the parameter values and verified the starting wetland WSEL.

It is suggested that the spreadsheet be printed on legal size paper using the landscape orientation. Setting the scaling to fit to one page wide by eight pages tall produces a document at a 60 percent reduction that is suitable for submission to BRD.

4. Examples

Two examples with two scenarios each were prepared to illustrate the utilization of the WETENUFF procedure. The examples are described below, and the parameters for each scenario are shown in Table 2. Selected results for each scenario are shown in Table 3. The shaded lines indicate the parameters and the results that were varied to obtain the stated objectives of each example.

4.1 Example 1

In example 1, the applicant for a development is attempting to determine how much of the runoff from the area to be developed should be routed into the existing wetland area in order to maintain the viability of the existing wetland. The applicant is required to show that either the wetland is inundated for at least seven consecutive days or that the wetland is saturated for at least twenty consecutive days. A monitoring well was installed on the site. A wetland area exfiltration rate of -1.65 centimeters per day was computed using the average recession rate of three storms. Initially, 0.57 hectare of adjoining developed area were routed into the wetland, as indicated in scenario 1. The WETENUFF procedure results indicated that the maximum periods of ponding and saturation would be five and sixteen days, respectively.

Because the wetland hydrologic requirements were not met, an additional 0.08 hectare of development drainage was added to the wetland, as indicated in scenario 2. The parameters used in scenario 2 resulted in maximum periods of ponding for eight days and saturation for seventeen days. Because these periods exceed the wetland hydrologic requirements, the design using scenario 2 was found to be acceptable.

4.2 Example 2

In example 2, a wetland mitigation area is being designed. The species of vegetation to be established in the proposed wetland area require continuous inundation for at least eight months. The plants also require year around saturation, with the exception of a period of about four to six weeks. The initial design set the wetland ground surface elevation at 4.27 meters, as shown in scenario 3. The WETENUFF procedure results indicated that continuous inundation would occur for less than seven months and that a period of about eleven weeks without saturation would occur as initially designed. The wetland ground surface elevation was lowered to 3.96 meters, and the WETENUFF parameter was modified to reflect the design change. The minimum and maximum wetland depth factors were adjusted to produce elevations consistent with scenario 3. The results of the design change are indicated in scenario 4. Because continuous inundation was indicated for over eight months and saturation was indicated to occur year around except 36 days in February to April, the revised design was found to be acceptable.

Table 2. Example Parameters

	Example 1		Example 2	
Parameter	Sc. 1	Sc. 2	Sc. 3	Sc. 4
wetland area (ha)	0.405	0.405	0.100	0.100
rainfall zone	3	3	4	4
wetland ET (cm/year)	119	119	119	119
wet. gr. surf. elev. (m)	2.13	2.13	4.27	3.96
wet. soil stor. rt.(cm/m)	14.2	14.2	14.2	8.06
wet. exf. rate (cm/day)	-1.65	-1.65	-0.51	-0.51
adj. pervious area (ha)	0.16	0.16	0.30	0.30
adj. impervious area (ha)	0.40	0.49	1.32	1.32
total adjoining area (ha)	0.57	0.65	1.62	1.62
average adj. gr. elev.(m)	2.74	2.74	4.88	4.88
adj. control/GW elev. (m)	1.22	1.22	3.35	3.35
adj. area depth to GW (m)	1.52	1.52	1.52	1.52

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adj. comp. soil stor. (cm)	20.8	20.8	20.8	20.8
adj. soil storage (cm)	5.94	5.21	3.84	3.84
min. wetland depth (m)	0.34	0.34	1.22	0.91
max. wetland depth (m)	0.30	0.3	0.3	0.61
assumed wet. WSEL (m)	1.80	1.80	4.34	4.34
saturation depth (m)	0.30	0.30	0.30	0.30

Table 3. Selected Results

	Example 1		Example 2	
Selected Results	Sc. 1	Sc. 2	Sc. 3	Sc. 4
31 Dec ending WSEL (m)	1.80	1.80	4.34	4.34
number days of ponding	21	29	258	319
number days of saturation	75	82	N/A	N/A
number days without sat.	N/A	N/A	90	36
longest per. of pond, day	5	8	202	268
longest per. of sat., day	16	17	N/A	N/A
longest per. w/o sat., day	N/A	N/A	82	33

5. Improvements

Comments and experiences pertaining to the usage of the procedure will be welcomed by the authors. Suggested modifications may result in improvements to the procedure.

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