Proceedings of the 12th Annual Conference on Wetlands Restoration and Creation, May 16-17, 1985. 1985

Proceedings of The 12th Annual Conference on Wetlands Restoration and Creation

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Hillsborough Community College

PROCEEDINGS OF THE TWELFTH ANNUAL CONFERENCE ON WETLANDS RESTORATION AND CREATION

May 16-17, 1985

Sponsored by
Hillsborough Community College
Environmental Studies Center

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INTRODUCTION

The Annual Conference on Wetlands Restoration and Creation provides a forum for the nationwide exchange of results of scientific research in the restoration, creation, and management of freshwater and coastal systems. The conference is designed to be of particular benefit to governmental agencies, planning organizations, colleges and universities, corporations, and environmental groups with an interest in wetlands. These proceedings are a compilation of papers and addresses presented at the Twelfth Annual Conference.

This year's conference would not have been possible without the assistance and cooperation of Mr. Roy R. Lewis, III. We are grateful for his untiring help and participation, as well as Charles Deusner for providing administrative support for the conference.

The following people also deserve thanks for contributing to the conference and assisting in the preparation of the proceedings for publication: Fay Crowe, Jackie Watford, Cinthia Fields, Gary Lytton, Leslie McFetridge, and David Walker.

Thanks are also extended to Richard Callahan, Colleen O'Sullivan, and David Robertson for arranging and conducting field trips to wetland restoration/creation sites.

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

To all these people, thank you.

KEYNOTE ADDRESS

COMMENTS ON THE KISSIMMEE RIVER RESTORATION PROJECT

By:

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INTRODUCTION

I am here today at the invitation of our very able conference organizer, Fred Webb. He extended the invitation several months ago on the strength of an article I had done in last July's issue of Florida Environmental and Urban Issues. That article, entitled "The Kissimmee: From There to Here," was a historical review of the Kissimmee River as an environmental and political issue over the past century, from the Disston era to the decision by the Coordinating Council to recommend eventual dechannelization.

That article was the first of a two-part series. The second article, to be entitled "The Kissimmee: From Here to There," is nearly completed and may appear later this year.

Today, I'll sort of preview that second article. I feel obligated to do that since that's how I came to be here today, and because that's how my colleagues on the panel also came to be here. But if time permits, I also want to explore some larger issues which have captured my interest as an outgrowth of my work on the Kissimmee. Conducting, or at least trying to conduct, a program of objective evaluation and ancillary research in an emotionally charged adversarial atmosphere has been one hell of an interesting and educational experience, with many lessons learned. I have to confess to you that when I suggested the names of possible panelists to Fred Webb, I truthfully did not really expect to see so many of them actually here. I am delighted that they were able to be here, though. Partly because of their various involvements in the Kissimmee River issue, but more so because of their breadth of experience in multiple aspects of governmental science.

Let's use the Kissimmee as a point of departure and go where we may.

KISSIMMEE SYSTEM

Current System

The Kissimmee River Basin is located in central Florida and encompasses a drainage area of approximately 1,400,000 acres. There are two major sub-basins: the upper "Chain-of-Lakes" basin covering a poorly defined depressional watershed of a little less than a million acres, and the lower Kissimmee River Valley which drains an area of about 437,000 acres.

Upper-basin lakes are regulated on controlled schedules and drain to the lower basin through Lake Kissimmee.

The lower basin was channelized between 1962 and 1971 at a cost of nearly 24 million dollars. The channelization transformed the system from a meandering river of a little over 100 miles in length to a canal stretching nearly 52 miles from Lake Kissimmee to Lake Okeechobee, covering a straight line distance of a little more than 35 miles.

The canal system consists of five stair-step pools of varying lengths controlled by a series of six structures.

The canal averages 30 feet in depth throughout its length, but its length, width and water level vary in each pool. The head, or difference in water level above and below each structure, varies from structure to structure and with rate of discharge, but is typically about six feet.

The system is designed to transport the mythical "Ten Year Flood Event" in bank, without flooding.

The existing system has approximately 20,000 acres of wetlands, as compared to the prechannelization dry season—wet season range of from 23 to 38 thousand acres. Much of the current acreage of wetlands, however, is qualitatively different than that which existed before, primarily as a result of the more static water levels in the existing system.

Recommended Plan of Alteration

The alteration scheme recommended by the Coordinating Council is known as the Partial Backfilling Alternative. This plan envisions creation of a presumably more natural appearance and hydrologic functioning by backfilling about the middle half of the canal system while preserving some degree of water management capability by retaining the upper 15 miles of canal to provide get-away capacity to the upper basin. It would also retain the lower dozen or so miles to allow for collection and passage of peak flows from the river to Lake Okeechobee. This alternative would presumably recreate a riverine system in about two-thirds of the flood-plain. Under this alternative, approximately 20.5 miles of C-38 would be backfilled, restoring flow to approximately 41.4 miles of remnant river channel. This is equivalent to about 53 percent of the original length between Lake Kissimmee and S-65E.

Demonstration Project

At the same time that the Coordinating Council recommended the Partial Backfilling Alternative, it directed the South Florida Water Management District to develop plans for a proposed demonstration project to actually field test key elements of the PBF alternative.

As ultimately permitted, the demonstration project included creation of a flow-through marsh on the east side of the canal, partial destruction of an existing flow-through marsh on the west side, recreation of flow under certain limiting conditions in three segments of remnant channel, and pool stage manipulation or fluctuation over a three foot range.

There are critical differences between the system configurations of the PBF and the demonstration project which make it difficult to assess the potential value of the latter in terms of its ability to predict effects of the former. Most importantly, the PBF envisions a sloped water surface (when there is water) while the demonstration project will occur in a controlled and essentially flat water situation. At this point, the Governor, with an entourage of a thousand and one press people, has planted a cypress tree, 2½ of the sheet-pile weirs have been built, and a pool stage fluctuation schedule has been adopted for probable implementation this fall.

I have to chuckle at a certain irony here. We wanted to plant trees there eight years ago but were blocked by a so-called environmentalist. The Water Management District first proposed the pool stage fluctuation in 1972, and actually did it back in 1973 and 1974, or thereabouts, with the complete cooperation of affected landowners, but it was rejected as a restoration option. And one of the great environmental success stories of the state—The Boney Marsh Complex—is being largely destroyed in the name of restoration.

Oh well.

But we're a little ahead of the story here. Let's back up and take a look at a complex of related events leading to the Coordinating Council's decision to recommend Partial Backfilling of the Kissimmee System.

RUSH TO DECISION

Preliminary Results

The Corps began its survey review of the Kissimmee River in November, 1978, and was finally able to present preliminary results in July, 1983. The key questions of interest to the Coordinating Council were:

1. Whether dechannelization would cause additional flooding in the more heavily populated (and hence, more politically potent) upper basin,

2. How much would restoration cost (and who would pay for it), and

3. How great would the environmental benefits be.

Flooding

The critical finding to emerge from the Corps' hydrology and hydraulics (H and H) analyses was that the Partial Backfilling (PBF) alternative would not cause increased flooding or damages in the upper basin—the chain of lakes extending north from the State Road 60 to the State Road 50. The underlying reasons for this surprising finding are that the existing system of works in the upper basin is unable to move water south fast enough to effectively use the Canal's hydraulic potential, and that the conveyance capacity of the Kissimmee River, as restored by the PBF, would be sufficient to accommodate discharges from the existing upper basin system. The practical effect of the finding was to remove a huge potential political impediment to dechannelization. The Corps' findings did predict extensive flooding, and consequent damages of about a million dollars a year in the lower basin, between State Road 60 and Lake Okeechobee. However, there are few people in that area and cows still do not have the vote.

Cost

Cost estimates generated a good deal of confusion, partly due to the segmented fashion in which they were presented, but mostly due to smokescreen criticism from dechannelization advocates who feared the costs were so large as to threaten the marketability of dechannelization. In round numbers, the Corps was projecting construction (or de-construction) costs of about $50 million and spoil purchase costs of about $20 million, for a sub-total of about $70 million in upfront, direct costs.

The other major cost factor which the Corps had not attempted to compute directly was the potential land acquisition cost. There are about 41,000 acres within the 10 year flood profile and only 14,000 are publicly owned. Well, there used to be about 41,000 acres, but it has apparently grown some. I believe the Water Management District now says that there are about 52,000 acres. Thus, a dechannelization project is faced with the possibility of having to acquire either the land itself or flowage easements. Estimating the potential cost of such acquisition is impossible, in a practical sense, because the market value of included land varies from as low as $500-$600 per acre to as much as $15,000 per acre, and there is no way of knowing how much would be acquired in fee simple, versus suitable easements or trades. I understand that recent purchases in the area by the WMD have averaged about $700 per acre. There never was a consensus estimate of potential land acquisition costs, but objective observers generally believed that those costs could range from $25 million to $50 million in upfront dollars.

Altogether, the necessary front costs to dechannelize the river/canal could easily reach $100,000,000. That is undeniably a substantial sum, but to keep a sense of perspective, we should note that it is less than the cost of a single B-1 bomber, and roughly equivalent to building one mile of the Port Everglades Expressway (I-595) or 30 miles of rural interstate highway (asphalt with no interchanges and bridges).

Environmental Benefits

The question of environmental benefits didn't fare too well in the preliminary results. Such intangible, though nonetheless real, benefits have always been—and still are—extremely difficult to quantify in any obviously rational way. At the strong urging of the Florida Game and Fresh Water Fish Commission and the U.S. Fish and Wildlife Service, the Corps approached the wildlife element of environmental benefits by use of a numerical method known as Habitat Evaluation Procedure—the HEP analysis. To the surprise of many, and the great consternation of several, the analysis indicated that the principal wildlife species to benefit from the proposed dechannelization would be alligators and water rats. There were, of course, other wildlife which would likely be adversely affected, such as deer, turkey and the panthers that the local people swear are there. However, habitat impacts on these species (except deer), as well as livestock populations in the valley, were not evaluated in the HEP analysis.

Some of these preliminary findings may have changed some or been refined in the Corps' final report, but don't lose sight of the fact that it was these preliminary findings, not the final report, that were used to make the decision. Well, some were used, some ignored, and some abused.

Public Meetings

With the Corps' preliminary findings available, especially the basic "no impact" conclusion regarding upper basin flooding and damages, the Council opted to press forward to a decision as soon as possible. Accordingly, a series of public meetings was scheduled and announced for the second week in August, 1983. In preparation for these meetings, an issue discussion paper and a set of proposed findings and recommendations were prepared, generally under the aegis of the Coordinating Council but with substantive control of the product vested in philosophically committed agency staff. That eminently predictable point in the process had arrived when deliberate objectivity became less appropriate to the cause. Objectivity gave way to subjectivity, and evaluation was supplanted by justification. Such transformations (some would say subversions), are frustrating and exasperating, but nonetheless inevitable and, indeed, necessary. The entire sequence of events, after all, is known, by accurate euphemism, as the decision making process.

Out of the swirling techno-political fog, the discussion paper finally focused on eleven relatively distinct issues. Not so surprisingly, the issues were selected and framed in such a way as to bolster proponent concerns and undercut opponent arguments, with a smattering of objective issues for the impression of deliberation. With but slight liberty, the eleven issues can be reasonably categorized as flooding (3), costs (4), and technical (4). The "flooding" issues were (1) limitations of the existing systems, (2) potential effects on future development, and (3) flood damages. The first one, extolling the sophistication of contemporary hydrologic methods, reassured upper basin folks that they would not be adversely affected and held out the possibility of even improving the upper basins by noting that preservation of the northern 15 or so miles of the canal—which was intended—would allow for some increase in discharge from the upper basin. The second flooding issue argued that preserving the canal was simply anticipating future flood control needs which would stimulate development in flood prone areas and that removing the canal would remove the need for it, besides being consistent with the new policy of non-structural flood control. The flood damage issue basically argued that the estimated million dollars a year in flood damages to agricultural producers in the lower valley was just a smokescreen by the Corps to make dechannelization look bad.

The "cost" issues were (1) federal money, (2) construction estimates, (3) spoil acquisition, and (4) land acquisition. The first issue argued that cost sharing policies of the Reagan Administration (remember, this was during Reagan's first term) could be overriden by Congress, and that a change in administration could bring back the good old days (as under the Carter Administration). Besides, the Federal Government has a moral obligation to help the state restore Kissimmee. The construction estimate issue argued that the Corps over-estimated the Holey Land Reservoir Project by a factor of two and, therefore, their estimates for dechannelization were probably also too high—at least by a factor of two. Besides, the "very preliminary nature" of the estimates limited their usefulness in decision making; i.e., it was okay to ignore them. The spoil acquisition issue raised a valid question by suggesting it might be less costly to buy the land with the spoil on it than to buy the spoil off of it. The land acquisition issue revisited the long-standing question of whether or not reflooding would be a "taking," and asked the Corps to compute land costs for both "yes" and "no" scenario. It did not, however, note the equally longstanding answer that a court would ultimately have to decide the issue.

The technical issues raised were (1) water quality, (2) difference between "wetted lands" and "wetlands," (3) water levels in Lake Kissimmee, and (4) data adequacy. The water quality issue argued that restoration was necessary to prevent development in the basin and thereby prevent future water quality degradation. The "wetted" versus "wetlands" issue noted factors involved in development of "wetlands," and went on to downplay impounded wetlands in favor of "wetlands associated with a restored riverine system." The discussion ignored the fact that while the state was badmouthing impounded wetlands as an option for the Kissimmee, it was actively promoting construction of a huge impounded wetland system on the Holey Land tract in the Everglades Agricultural Area. The lake level issue noted errors in levels entered into the computer data base, and their potential impacts on flooding and damage estimates, and went on to argue that those estimates were sure to be affected by ongoing efforts to determine the Ordinary High Water Line (OHWL) which is critical in determination of private/public ownership. The final issue of data adequacy acknowledged that there were still some uncompleted studies, but said, "So what? There is no other river basin in Florida which has been more exhaustively studied."

COLLATERAL EFFORTS

While the Coordinating Council was preparing to make an official decision, the Governor's office was putting the finishing touches on two important related items: appointments to the Governing Board of the South Florida Water Management District; and the Governor's environmental capstone, Save Our Everglades.

The Water Management District Board (WMD) had a majority of its nine members up for reappointment or replacement for terms beginning in July, 1983. Various environmental groups had been lobbying the Governor for nearly a year to change the Board by appointing members committed to their general view of the world, but especially so as regarded the Kissimmee River issue. Their strategy was well laid, for it was clear that the WMD would be a key actor as the Kissimmee saga continued to unfold. Four of the appointments were made in early August, to the elation of environmental interests and to the great consternation of many others. The rumor mill was rife with allegations that the appointees had "taken the pledge" to support dechannelization of the River—and all this before the Council had made its determination. The glee and gloom engendered by the appointments were two sides of the same coin—deliberate politicization of the Board and the resultant compromised professionalism of the staff. The Governor didn't resolve any of these concerns when he confirmed to a group of businessmen and agricultural producers in November that he had indeed "assured myself that the appointees were supportive of major policy directions of my administration."

In addition to assuring support of the WMD, the appointments were also widely seen as converting a presumed "no" and an announced "undecided" vote on the Council to confirmed "yes," thus assuring at least a 4:1 vote when the Council did make a decision. No one would argue—and none should expect—that the Council could make a totally objective, apolitical decision, but individually and collectively the Council members had consistently struggled to make a responsible decision. Whether or not the decision ultimately made by the Council was a responsible one is a judgment best left to some future student of history, because the contemporary effect of the Board appointments was to cast a mantle of suspicion over the Council's action.

The other collateral effort by the Governor's office was the development and packaging of the "Save Our Everglades" program. The program collects under a single banner several projects already underway in some form or other and thereby creates a composite of seeming greater impact than the sum of its parts. Inclusion of the Kissimmee issue no doubt garnered additional visibility for the overall program. It certainly infused some apparent gubernatorial interest and resultant publicity in the Kissimmee question. The grand unveiling of the master plan was timed to coincide with the Coordinating Council's public meetings and scheduled float trip on the river/canal. The result was an armada of vessels and a squadron of helicopters such as was probably never seen before, nor likely ever again, on the used-to-be-and-quite-forgotten Kissimmee River, and prime-time television coverage over most of peninsular Florida for a couple of days.

A tangible effect of the inclusion was the delayed but eventual appointment of a Resource Planning and Management Committee under the provisions of Chapter 380, Florida Statutes. Appointed exactly a year and a day after the Council reached its decision, the Committee has the task of developing recommendations for institutional mechanisms to control development which might adversely affect the river/canal, restored or not.

The overall effect of all the publicity on the river is difficult to gauge. Certainly, the folks at center stage benefited greatly, at least in the short run. In the longer run, the resource, or its rational use will likely also benefit, for increased visibility results in increased scrutiny and, to a Madisonian democrat, that means the public is better able to make a well-informed decision.

Well, we've talked about where the system stands now, and taken a look inside the decision making process to understand better how we got to wherever it is that we are, or think we are.

CURRENT QUESTIONS AND POLITICAL CONCERNS

There are many questions we could consider here, but there are three that will, I think, capture the essence of the issue.

First, is it necessary to do anything to the Kissimmee River/Canal system. The answer is NO! I know of no compelling technical reason to do anything to the system, including just leaving it alone. What that really means is that we can do anything so long as we are willing to foot the bill and live with the consequences.

The second question is whether or not it is possible to restore the

Kissimmee River, and here again, the answer is NO. The river can not be restored. There are several reasons for this inescapable conclusion, but they can be summed up by population, passage of time, and recognizing that the river/canal system is in the middle of a highly structured, carefully controlled plumbing system.

The third question is whether or not the existing system can be altered in positive ways, and the answer here is an emphatic YES! There are many options and opportunities to alter the system in ways and to degrees that the great majority of taxpayers would be able to enthusiastically support.

There is a serious problem, though, in that adversarial politics have thus far prevented full development and objective evaluation of many viable improvement options. This supressive strategy has led to a situation in which the state leaders have committed to an impossible objective. Being, however, that they are astute followers of the people and agile politicians, we can reasonably assume the existence of an escape option somewhere.

SPECULATIONS AND PROGNOSTICATIONS

There is an obvious end point to this discussion, but not to the issue of the Kissimmee River/Canal system. Let me, therefore, offer some speculations and prognostications as to the incipient meanders of the issue. I base these projections on three beliefs. First, as a staunch Madisonian democrat (democrat with a small d), I place great faith and confidence in the public's ability to make a well-considered judgment when provided with all the pertinent information. Second, I recognize that there are many sane and rational people yet involved in the continuing process of charting a future for the system. And third, political postures tend to be ephemeral and astute politicians follow the people.

I suggest to you that the public's perception of restoration will eventually focus on realistic and affordable improvements to the system, and that the king will eventually learn about his new suit of clothes, and in the process, an irrationally constructed win-lose confrontation may be resolved with a win-win solution.

CLOSE

That completes what I wanted to say about the Kissimmee. So, I will close with a question. How long has it been since you thought about "wetlands" as two words?

RESPONSE

John B. Cruce
U.S. Army Corps of Engineers
Jacksonville District

GENERAL

The Kissimmee River Basin occupies 3,018 square miles within the Central and Southern Florida Flood Control Project. Nine water control structures regulate water levels and flow in the upper Kissimmee Basin. A channel, Canal 38 (C-38) with six water control structures regulates water and flow in the lower basin between Lake Kissimmee and Lake Okeechobee. Taylor Creek and Nubbin Slough are smaller basins to the east of the Kissimmee River and comprise about 200 square miles. The puported problems are those associated with the channelization of the Kissimmee River and the possible eutrophication of Lake Okeechobee.

The purpose of the Corps' study is to evaluate the feasibility of modifying the existing flood control project for purposes of improving water quality and enhancing fish and wildlife resources in the study area. Other areas of concern include restoration of wetlands, maintaining flood control and navigation capabilities, providing water supply, and increasing recreational opportunity. Potential modification alternatives include the possible restoration of all or parts of the Kissimmee River below Lake Kissimmee.

SITUATION

In 1954, Congress authorized improvements to the Kissimmee River in response to the State of Florida's request for improved flood protection within the Kissimmee River Basin. Construction of the canal which channelized the Kissimmee River, Canal 38 (C-38), began in 1962 and was completed in 1971 as part of the Central and Southern Florida (C&SF) Flood Control Project.

Following completion of construction, there was concern that the widening and deepening of the river along with the adjoining basins of Taylor Creek and Nubbin Slough were contributing to the eutrophication of Lake Okeechobee which is the principal water storage area for South Florida. Canal construction also had a number of perceived detrimental environmental effects such as a loss of fish and wildlife habitat associated with the basin's wetlands, and the loss of the esthetic quality inherent in a natural meandering Kissimmee River.

Several state studies led the 1976 Florida legislature to create the Coordinating Council on the Restoration of the Kissimmee River Valley and Taylor Creek-Nubbin Slough Basin. This State Council proposed major modifications to the existing project to the extent that project purposes would be changed. It was determined that Congressional approval would be required for implementation of such modification.

In response to a request by the 1977 session of the Florida legislature, Congress authorized the U.S. Army Corps of Engineers in 1978 to review the existing flood control project to determine whether modification of those works was advisable. The study authorization stated that these modifications shall include, but not be limited to, consideration of restoration of all or parts of the Kissimmee River below Lake Kissimmee.

The current study which began in November, 1978, has been conducted by the Corps in close coordination with both the Kissimmee Coordinating Council and the local study sponsor, the South Florida Water Management District (SFWMD). In August, 1983, the Council made its recommendation for project modification as generally proposed in the Partial Backfilling alternative. The Council also recommended that the state of Florida assume primacy in restoration of the Kissimmee River with instructions that state agencies proceed as rapidly as possible with backfilling operations.

The South Florida Water Management District was requested to begin immediate development of a program to accomplish the backfilling objective. The Council endorsed the Water Management District's proposal of a demonstration backfilling project. This demonstration project would divert water into meanders adjacent to the canal in Pool B, effectively eliminating canal conveyance.

The Jacksonville District of the Army Corps of Engineers received a permit application from the state for a "Demonstration Project" for Phase I of the project to restore the Kissimmee River. Phase I generally consists of the placement of steel weirs within the channel to divert flows into the river oxbows. Phase II of this state project refers to the actual filling or plugging of the canal. This latter action has been determined to require an Environmental Impact Statement (EIS) as project purposes will be modified, and is addressed in the Kissimmee River Study.

Upon evaluation of findings to date and after careful review of current policies and guidelines concerning Federal water resource planning, it appears that there is no basis for Federal implementation of modifications to the existing project within the Kissimmee River Basin. However, the report nearing completion is expected to provide information which could be used by local interests in determining long-term solutions to water and related land resource objectives within the basin. If implemented, these measures are expected to result in increased wetland acreages and associated wetland habitat units, improved water quality, the maintenance of flood protection, as well as navigational and recreational benefits.

During the past 15 years, an extremely high degree of interest and concern has been maintained over the possible river restoration within the Kissimmee River Basin. The intense interest expressed locally, statewide, and within the national media further heightened interest in the formulation, evaluation, and recommendations contained within the Corps' report. The restoration of the Kissimmee River is part of Governor Graham's "Save Our Everglades" Program.

QUESTIONS AND ANSWERS

A. QUESTION: What have local interests accomplished to date toward filling in the channelized Kissimmee River project?

ANSWER: To date, local interests have constructed two weirs under a Corps' permit across the channelized river (C-38). These steel, sheet pile weirs are expected to divert canal flows into portions of the old river meanders. The weirs also contain a notch for navigation. The success of the weir is to be assessed toward the restoration of wetland acreages, as well as the possible enhancement of the basin's fish and wildlife resources. An additional weir is to be constructed this year by the South Florida Water Management District.

Concurrently, under Federal local cost-sharing arrangements, the existing control structures S-65 (A, B, C, and D) are being upgraded to original design specifications by adding baffle block on the discharge apron. This modification will provide the water management capability to control flood waters as authorized.

The actual plugging or filling of the canal (C-38) as proposed in the state's Phase II demonstration project is expected to be reassessed after the evaluation of the Phase I monitoring program currently scheduled for a three-year time period. Further action by state interests toward actual plugging of the canal is expected to require Congressional approval.

B. QUESTION: What would be involved in restoring the Kissimmee River as requested by the state of Florida?

ANSWER: The Kissimmee River restoration strategy as currently proposed by the state of Florida calls for a phased program to restore the values of the Kissimmee River. Further evaluation of various "mix and match" scenarios is proposed within the Corps' report, as are other plans such as possibly altering existing regulation schedules within the upper chain of lakes. Acquisition of the Kissimmee River floodplain consisting of some 50,000 acres, and the implementation of Best Management Practices to improve basin water quality are included within the state's program. Further evaluation of the ongoing Phase I Demonstration Project is a prerequisite to the implementation of Phase II, the actual plugging of the existing canal.

C. QUESTION: What other options are feasible to restore or enhance the Kissimmee River Basin's natural resources?

ANSWER: Because of the current hydrologic conditions within the basin, certain plans as formulated and as evaluated do not appear to meet the restoration objective. Findings of the Corps' report indicate that water volumes needed for wetland restoration are not available under current basin conditions. Options are available, however, such as possible schedule modifications in the upper basin which, through evaluation, may provide additional water supply sources.

Because of the water management controls within the upper Kissimmee Basin which are expected to remain in place and other factors of local control, the measures evaluated in this report which offer the greatest potential for meeting study objectives include a fluctuation of water levels (Pool Stage Manipulation), providing increased surface flows into drained areas, such as Paradise Run, and the implementation of Best Management Practices (BMP's) for water quality benefits. The cost of this scenario is approximately $10 million. This does not restore the river per se, but does restore some of the qualities associated with a riverine system. These qualities include increased wetlands and improved fish and wildlife habitat. Futher evaluation of the "mix and match" scenarios could also provide insight into measures which enhance the basin's resources. Implementation of some of these conceptual plans could be accomplished in a relatively fast time period under discretionary authority of the Chief of Engineers.

D. QUESTION: Do you foresee Federal involvement in further Kissimmee River restoration efforts?

ANSWER: Based on the Corps' report which will incorporate recently received public review and comments, there appears to be no National Economic Development (NED) benefit with which to base a recommendation for Federal implementation of a plan to modify the existing project as specified under current guidelines. Tools are available, however, for the Corps working with others to evaluate additional alternative restoration plans as may be formulated in conjunction with Federal, State, local, and individual interests to protect the human and natural resources of the Kissimmee River Basin.

E. QUESTION: What is the Federal (Corps') interest in the state of Florida's Kissimmee River restoration efforts?

ANSWER: The Federal interest is basically twofold: (1) flood control or flood damage prevention, and (2) navigation. The former may be negated if assurance can be made that current flood protective works in the lower Kissimmee River basin would not be needed. This could occur because of actions currently under way such as state acquisition of those privately owned lands now afforded water management control.

Navigation within the Kissimmee Basin remains a Federal interest. Any action which would impede or restrict navigational usage of the system as authorized would require Congressional approval to permit a reduction of those navigational benefits.

F. QUESTION: The Corps' Kissimmee River Feasibility Report states that hydrologic condition (water availability) within the basin may restrict the development of a riverine or wetland restoration effort. Could the regulation schedule now in effect within the Upper Chain of Lakes be modified or altered to provide additional volumes of flow to the Lower Basin?

ANSWER: This plan of action was proposed in the early phase of the study. This concept as originally proposed, however, was to mitigate the potential loss of flood control capability resulting from implementation of certain restoration alternatives within the lower Kissimmee River Basin.

Early in the study process, the premise was made that the lower basin alternatives would not negatively impact the upper basin. Regarding schedule modification, no management scheme will increase the amount of water available to the lower basin, however, the possibility does exist that flood waters, normally passed through the system, could be stored in Lakes Kissimmee, Cypress, and Hatchineha and released at a reduced rate, thus increasing the hydroperiod. Storage of these flood waters within the Upper Basin would be in the zone now reserved for flood control, therefore, analysis of the impacts on affected landowners and water management authorities in that basin would be necessary.

G. QUESTION: What is the current schedule for submittal of the Corps' final Kissimmee River Feasibility Report?

ANSWER: The draft report is currently undergoing review, having received comments from Federal, State, and local interests. The concerns, issues, and other comments received during the report review period are now being incorporated into the final report. This report is expect to be available to the public in the summer of 1985.

H. QUESTION: Wouldn't restoration of wetlands within the lower Kissimmee River Basin improve the water quality of Lake Okeechobee?

ANSWER: It is generally agreed that wetlands provide an improvement in water quality. Those marshes and wetland areas associated with peat and muck accumulations are essentially functioning as net accumulators of nutrients. Riverine marshes such as those within the Kissimmee River Basin are seasonal repositories for nutrients, accumulating nutrients during high runoff and flood conditions. While some water quality improvement is expected within C-38 below the structure S-65E, there is no indication that any of the canal modification alternatives will significantly improve water quality in Lake Okeechobee.

I. QUESTION: What impact does the restoration/nonrestoration of the Kissimmee River have on the Governor's proposal plan to "Save the Everglades?"

ANSWER: The Kissimmee River, along with basin rainfall, provides the major water supply source to Lake Okeechobee, the lake that has been described as the liquid heart of Florida. At the turn of the century, surface outflow from the lake occurred primarily during wetter periods. This surface release manifested itself as sheet flow through the sawgrass Everglades south of the lake.

Lake Okeechobee is now completely regulated by water control structures, and enclosed by a dike up to 25 feet in height. Water is stored in the lake for delivery to the lower east coast, the Everglades agricultural area, and to the Everglades National Park during periods of drought. Because of these water management controls and current operational criteria, modifications addressed in our study for the lower Kissimmee River Basin are not expected to significantly impact the volume nor quality of water which is delivered to the Everglades.

RESPONSE

Jeanne Hall
South Florida Water Management District
West Palm Beach, Florida

The core mission of the South Florida Water Management District is to manage water and related resources for the benefit of the public and in keeping with the needs of the region for the purposes of providing: environmental protection and enhancement, water supply, flood protection, and water quality protection.

Kissimmee River restoration is being undertaken as part of an overall plan to protect and manage the natural resources of the Kissimmee River-Lake Okeechobee-Everglades ecosystems. By Exeuctive Order, issued November 4, 1983, Governor Bob Graham created the Kissimmee River-Lake Okeechobee-Everglades Coordinating Council (KOECC), and set forth the following objectives:

Avoid further destruction or degradation of these natural systems.

Reestablish the natural ecological functions of these natural systems in areas where these functions have been damaged.

Improve the overall management of water, fish and wildlife, and recreation.

Successfully restore and preserve these unique areas.

The Council's responsibilities in the Kissimmee River Valley specifically require it to oversee restoration efforts. Each member must also actively work to implement the restoration program by making full use of his respective regulatory, planning, monitoring, and enforcement powers within the project area.

The Council must also adhere to a Memorandum of Agreement that spells out each agency's responsibilities with regard to the restoration project. Under the Memorandum of Agreement, executed October 28, 1983, the South Florida Water Management District agreed to develop, design, construct, operate, and maintain a demonstration program to dechannelize the Kissimmee River.

Phase 1

The District developed Phase 1, a demonstration project to field test methods of reestablishing a more natural water regime in the Kissimmee River Valley. While the channelization that took place from 1963 to 1971 enhanced flood protection in the region, some environmental degradation resulted when the area's wetlands were diminished.

The structural aspects of the Demonstration Project are designed to divert water back into historic oxbows and formerly existing marsh lands.

Another component of Phase 1 will entail the fluctuation of water levels to more closely correspond to natural wet and dry cycles typical of an Everglades hydroperiod. The levels are currently held at a stable 40 feet mean sea level (m.s.l.) throughout the year. Under pool stage manipulation, water levels would range between 39 feet and 42 feet m.s.l. The lower stages will provide drier conditions that promote plant germination and growth. The higher water stage will induce an aquatic habitat by inundating marsh lands long enough to reestablish wetland ecosystems. It is hoped that, in this way, as many as 1,300 acres of wetlands will become integrated into the riverine system.

The Demonstration Project will take place in Pool B, a 12-mile reach between structures S-65A and S-65B in the C-38 canal. The project lies within the boundaries of Polk, Osceola, Okeechobee and Highlands counties.

The first item to be put in place will be the northernmost weir, weir #3, then weir #2, and weir #1 will be constructed. The culverts and berms will be constructed after the last of the weirs is in place. The total construction timetable spans about 10 months.

Monitoring Regime

The Momorandum of Agreement divides the task of monitoring the project among several public agencies. Each agency will determine whether the project has successfully reestablished natural values in these areas:

Department of Environmental Regulation will design the water quality monitoring program.

Game and Fresh Water Fish Commission and the DER will assess the project's impact on fish and wildlife and vegetation.

Effects on hydrology and hydraulics will be monitored by the South Florida Water Management District. The District will use this data to evaluate the Demonstration Project's impact on flood protection. The District will also be testing the feasibility and cost effectiveness of construction methods used.

The monitoring program will begin during the construction phase, and will evaluate the project's performance through two wet cycles. The KOECC will evaluate the data collected, then address further restoration of the Kissimmee River.

Benefits

Wetlands ecosystems have always been an important component of water management. Marshes and swamps are natural reservoirs where potential flood waters can be diverted, to be stored and conserved for future use. Some of the water captured in wetlands recharges the aquifer, the underground source of most of our water supplies. The slow movement of water that is characteristic of wetlands improves water quality by allowing impurities to be filtered out.

The riverine marsh system that is to be reestablished by the Demonstration Project will restore some of these wetland functions by renewing water flow in natural oxbows and by creating overland flow across marsh areas.

These environmental enhancements will be perused by the District, but not to the detriment of flood control and navigation capabilities that now exist in the area. Phase 1 will help the District find effective methods of meeting all our water management objectives in the Kissimmee River Valley: environmental enhancement, as well as flood control; water quality protection, as well as recreation.

RESPONSE

Mollie Palmer
Florida Department of
Environmental Regulation
Tallahassee, Florida

I would like to say a few words about the environmental benefits Pat mentioned and the recent Corps study and how it might affect the state's restoration efforts for the Kissimmee.

The draft Corps study completed last fall did not recommend a dechannelization alternative for the Kissimmee. Rather, it concluded that a combination of the following measures offers the greatest achievement of the objectives and the greatest benefit at the lowest cost: pool stage manipulation, implementation of BMP's and restoration of Paradise Run. They concluded that the river restoration alternative, partial backfill, did not restore as many acres of wetlands as other alternatives and that it might even result in a reduction of wetlands.

The state disagrees with many of the environmental conclusions in the Corps study, and we feel the book is not closed on the dechannelization partial backfill alternative.

I would like to touch on a few of the major issues in the Corps study which DER and other state agencies feel need to be reevaluated or require additional study for the partial backfill restoration alternative. These issues must be resolved for the state to proceed with restoration of the Kissimmee. Resolution of these issues is important not only for the state's development of its restoration strategy, but also to ensure that nothing in the Corps' report will inhibit the state from proceeding with restoration of the Kissimmee River system. Corps permits will be required for implementing the partial backfill alternative and Congressional approval may also be required. Following are the major areas on which there is disagreement between the Corps and the state.

Of significant concern to us is the Corps' conclusion pertaining to wetlands that result from the partial backfill alternative. This includes a reduction of wetlands by 3,300 acres. The Corps used three different assumptions to develop different estimates for the acres of wetlands recreated by the partial backfill alternative. The estimate of 14,700 acres (according to the Corps actually a reduction of 3,300 acres of wetlands) was based on the assumption that wetlands are areas that would be wet 50 percent of the time. The middle figure of 24,600 acres, an increase of 6,600 acres of wetlands, is based on the long-term average discharge. The highest figure of 43,200 acres, an increase of 25,200 acres, was based on the assumption that water would be available as it was before the project.

The Corps' study rejected the conclusion that 25,200 acres of wetlands could be recreated with the partial backfill alternative based on preproject hydrologic conditions. This conclusion was rejected for several reasons: (a) water entering C-38 is now controlled by management of the upper Kissimmee chain of lakes; (b) there has been less rainfall over the past several years; (c) the flow of Lake Istokpoga has been diverted from the Kissimmee River directly into Lake Okeechobee; and (d) upland drainage practices have altered natural storage capacity of the floodplain and uplands.

While we realize these variables have changed the hydrologic conditions in the Kissimmee basin, we do not assume that current conditions must remain unchanged. We cannot accept without further evaluation the study conclusions based on these assumptions for the following reasons:

A. Management of upper Kissimmee chain of lakes: Management schedules may be changed, and evaluations need to be made based on modeling of alternative management schedules for the different options before a final judgment is made on river restoration.

B. Decreased rainfall: Although we have had less rainfall in the last 18 years (postproject), there is no valid reason for considering only the data from the drier, more recent history and projecting those conditions to the future. If climatic conditions return to normal, the Corps' assumptions and conclusions will be invalid. A longer period of record which included both the preproject wetter years and the post-project drier years should have been used for the primary conclusion on wetlands recreation.

C. Lake Istokpoga diversion: The diversion of Lake Istokpoga, affects only the lower one-third of the Kissimmee River basin, since the lake originally drained into the river two-thirds of the way down. It, therefore, has had no effect on the floodplain of the upper river. There is also the possibility of restoring Lake Istokpoga's flow back into the river, which should be evaluated.

D. Upland drainage practices: We agree that upland drainage practices can alter the natural storage capacity of the floodplain and uplands; however, there has been no modeling done to show how these practices have affected gross wetland acreage in the floodplain. Furthermore, upland drainage and flood control should be included in best management practices, and if implemented, these should allay potential overdrainage problems.

A fair evaluation of the partial backfill alternative must include assumptions that include changing some or all of these current hydrologic conditions. We also feel the 50 percent inundation criteria is inappropriate for defining recreated wetlands.

In conclusion, we feel that the full range of wetlands developed by the Corps, with 43,200 acres (an increase of 25,200 acres) being the upper end of the range, should be considered in evaluating the partial backfill alternative. We also believe that the partial backfill alternative would not result in a reduction of wetlands, but would, at least, recreate somewhere between 6,600 and 25,200 acres of wetlands; this estimate would fall between the middle and upper range of the Corps' evaluation results.

RESPONSE

Larry Perrin
Kissimmee River Restoration Project
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KISSIMMEE RIVER RESTORATION PROJECT
LARRY PERRIN
SECTION LEADER

The Florida Game and Fresh Water Fish Commission (GFC) has been conducting fish and wildlife resources investigations within the Kissimmee River Basin since 1978. These studies indicate that substantial adverse impacts have occurred as a result of channelization of the Kissimmee River. Based on these findings, and an evaluation of the proposed restoration alternatives, the GFC has recommended that restoration be accomplished through implementation of dechannelization options. Only dechannelization would restore both wetlands and riverine functions, i.e., water level fluctuation and flow.

Dr. McCaffrey stated in his presentation that it is not necessary to do anything to the Kissimmee River. The GFC strongly disagrees with this position. Without any action or corrective measures the Kissimmee River would continue to deteriorate. Urban development would gradually encroach into and along its floodplain, further reducing water quality, and eliminating any future chance for adequate restoration. Consideration must also be given to anticipated impacts to Lake Okeechobee if the "no action" course is opted. Because the Kissimmee River is the largest tributary of Lake Okeechobee, its fate will have a significant influence on the future environmental quality of south Florida.

Many restoration advocates feel that restoration would be worthwhile just to correct a serious mistake and attempt to put back a unique river system. However, a more objective approach requires a holistic view of wetland trends within the state of Florida. At the turn of the century, Florida had approximately 20 million acres of wetlands. Currently, there are less than 12 million acres of wetlands remaining and most of these have been extremely altered. Such changes have had a dramatic effect on the fish and wildlife that these wetlands traditionally supported. As a result, the state has emphasized the need for protection of both wetlands and wetland wildlife values.

Wetlands which were once considered wastelands are now known to be some of the most productive natural systems in the world. They provide feeding and nursery areas for fish and wildlife, serve as natural filtering and cleansing systems, and give numerous opportunities for recreation, education, aesthetics, etc. Therefore, the people of the state of Florida stand to gain substantial long-term economical and ecological benefits through restoration of floodplain wetland acreage and riverine functions.

Initially, costs associated with implementation of a dechannelization option will be high. However, the long-term fish and wildlife values, water quality, and other benefits should far outweigh these costs, particularly when considering the anticipated adverse impacts to the river and Lake Okeechobee if no action is taken.

On another issue, Dr. McCaffrey made reference to a Habitat Evaluation Procedure study in which he stated that this investigation found restoration would benefit alligators and water rats. While this may sound amusing to some, the intent of this statement was to cast doubt upon and make light of the expected fish and wildlife benefits from river restoration. The fact is, 23 of 25 wildlife species evaluated during the course of this study (including the alligator and Florida water rat) would benefit from restoration. This study also illustrated the severe loss of habitat values associated with the channelization of the Kissimmee River.

In conclusion, the GFC supports restoration through dechannelization measures, as these are the only alternatives capable of restoring riverine functions. At the same time, the GFC also recognizes that if, for engineering resasons, these options cannot be implemented, then other measures should be incorporated. The GFC is, as previously mentioned, strongly against a "no action" option. Alterations could be made to this river/canal system that would be a definite improvement over its present condition. The incorporation of these measures are less desirable from an environmental standpoint than dechannelization, but, would be a much preferred course over that of a "no action" scenario.

THE *IN VITRO* PROPAGATION OF SEASIDE GOLDENROD *SOLIDAGO SEMPERVIRENS*

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ABSTRACT

Seaside goldenrod (*Solidago sempervirens*) is a common pioneer species on the primary dunes of the barrier islands of Virginia's eastern shore north of the Chesapeake Bay. It serves as a windbreak and an initiator of dunes which provide habitat for several animal species. Large numbers of individuals, suitable for use in dune stabilization projects, have been obtained by *in vitro* methods of micropropagation. Explants obtained from terminal and subterminal nodes of vegetative stems were cultured on various media combinations. Adventitious shoots were initiated within three weeks on Murashige's Minimal Organic Medium supplemented with 1-3 mg/1 kinetin. These shoots, when separated and placed on similar media with and without auxins, formed roots within five days and could be transferred to peat pots for greenhouse culture in 10 days. The use of *in vitro* micropropagation to produce numerous plants, which can be selected to enhance survival on the primary dunes, is discussed.

INTRODUCTION

Atlantic coastal barrier islands are an important resource. As a physical buffer between the ocean and the mainland, they protect coastal wetlands from flooding, erosion, salt intrusion, sand burial, and the generalized effects of storm surges. These islands also serve as the basis for a highly productive ecosystem capable of supporting a diverse array of both terrestrial and aquatic life. The physical and ecological integrity of these islands is maintained by a highly structured series of plant communities. The island floral complex is roughly divided into three zones, which, in order of increasing distance from the ocean, are: the pioneer or dune zone, the intermediate or shrub zone, and the back dune or forest zone. Fauna, including crustaceans, amphibians, reptiles, birds, and mammals are distributed among these zones according to the availability of suitable food and habitat (Woodhouse, 1982).

Despite the inherent value of these island chains and their associated salt marshes, this resource has been steadily shrinking. It has been estimated that 90 percent of the tidewater dune/marsh ecosystems between Maine and Virginia have been modified by human activity (Shaw & Fredine, 1956). Fortunately, the increasing recognition of the ecological and commercial value of these areas has promoted considerable effort toward the preservation and/or restoration of this resource. The plants used in most dune restoration projects have been selected for their tolerance of the pioneer zone environment, their ability to trap and stabilize sand, and their ease of propagation. The plants used most frequently in these projects are six grass species: American beach grass (Ammophila breviligulata), European beach grass (Ammophila areenaria), sea oats (Uniola paniculata), bitter panicum (Panicum amarum), saltmeadow cordgrass (Spartina patens), and American dunegrass (Elymus mollis).

The devlopment of additional species, especially dictoyledons, for use in dune revegetation is desirable in order to achieve maximal ecological diversity. For the Atlantic barrier islands south of Chesapeake Bay, seashore elder, yellow sand verbena, pennywort, and railroad vine have been suggested as possible candidates. The successful inclusion of these plants and others will require the selection of appropriate ecotypes and a rapid cost-effective method of producing vigorous planting stock (Lewis, 1982).

This research utilizes seaside goldenrod (Solidago sempervirens), a common dune species of Virginia's nothernmost barrier islands, as a model plant for developing propagation techniques appropriate for dicot dune species. In particular, this paper describes the technique of in vitro micropropagation which offers significant advantages for the production of alternative dicot species useful for dune revegetation. Appropriate ecotypes for a given region can be propagated at rates which are often higher than those obtained with other methods of propagation. Equally important, the production of vigorous planting stock can be adjusted to the sporadic demand for plants which is often associated with exceptional storm damage. This can be accomplished through the maintenance of small numbers of stock cultures, which can be subdivided as required, thereby minimizing the space requirements and loss of vigor associated with plants held in traditional nurseries.

MATERIALS AND METHODS

Seaside goldenrod, Solidago sempervirens, is a common dune inhabitant of Virginia's barrier islands north of the Chesapeake Bay. The distribution of this perennial subshrub, which is highly tolerant of salt spray, is limited to regions of minimal sand accumulation (Van Der Valk, 1974). On Virginia's Assawoman and Metompkin Islands, where primary dunes are less than 2m in height and few in number, Solidago is common in all zones and serves as a dune initiator through the trapping of debris and sand. These miniature windbreaks promote dune initiation by stabilizing sand movement and are often used as nesting sites by several species of migratory sea and shore birds. The secondary establishment of Salsosla kali, Euphorbia polygonifolia and Cakile edentula at these sites is common. On Assateague Island, which has a well-developed series of primary dunes, Solidago is restricted to the tops and backfaces of the dunes, which are also regions with minimal sand accumulation. Under these conditions, it serves primarily as a sand stabilizer and, to a lesser extent, secondary dune initiator. The plants are occasionally used as forage for island herbivores including rabbits, Sitka deer, and wild ponies.

Stock plants of seaside goldenrod, evaluated for maximum size, vigor, and branch pattern, were collected for greenhouse culture from Assateauge, Assawoman, and Metompkin Islands. These were maintained as sources of explant units. Standard procedures for the micropropagation of numerous species (Murashige, 1973) were adopted for the culture of Solidago. In brief, the tissues from the stock plants, the explants, which includes both axillary and terminal buds, were sterilized by soaking for five minutes in a solution of 0.525 percent sodium hypochoorite (10% Chlorox) containing 0.01 percent Triton as a wetting agent. The sterilized buds were rinsed three times with sterile distilled water. All manipulation of the explants was performed using aseptic technique in a sterile laminar flow hood. The buds were cultured on Murashige's Minimal Organic Medium. The pH was adjusted to a 5.6 with 1M KOH, and hormonal supplements were added prior to autoclaving the media for 20 minutes at 121° C. Cultures were maintained in growth chambers at 27° C under continuous illumination with cool, white, fluorescent lights at 150 $\mu E/m^2 S9/s$.

RESULTS

Shoot Formation

Excised terminal and axillary buds were cultured on the basal media supplemented with serial dilutions (0-4 mg/1) of the cytokinins benzyladenine (BA), 2-indole purine (2ip) and kinetin. On all media, initial bud expansion leading to leaf emergence was rapid, occuring within seven days. No additional growth was observed on the media supplemented with either BA or 2ip; however, on media containing kinetin, leaf expansion continued and stem callus was formed at the cut end of the explant within one week. Seven to ten days later, adventitious shoots arose from the stem callus as well as surfaces of the expanded leaves and petioles (Fig. 1A). Results for shoot initiation on the various concentrations of kinetin tested are given in Figure 2. By the end of the sixth week of culture, individual shoots were separated and subcultured on fresh kinetin media for additional shoot proliferation (Fig. 1B) or transferred directly to media containing auxins for rooting (Fig. 1C). In all tests, no differences with respect to shoot formation were observed between explants obtained from individuals collected from the different islands.

Figure 1. Adventitious shoot and root initiation on nodal explants of Solidago sempervirens. (A) Ten day old explants showing multiple shoot primordia. (B) Shoot proliferation on media containing 0.1 mg/l kinetin. (C) Root formation on shoot subcultured for 7 days on 0.1 mg/l NAA.

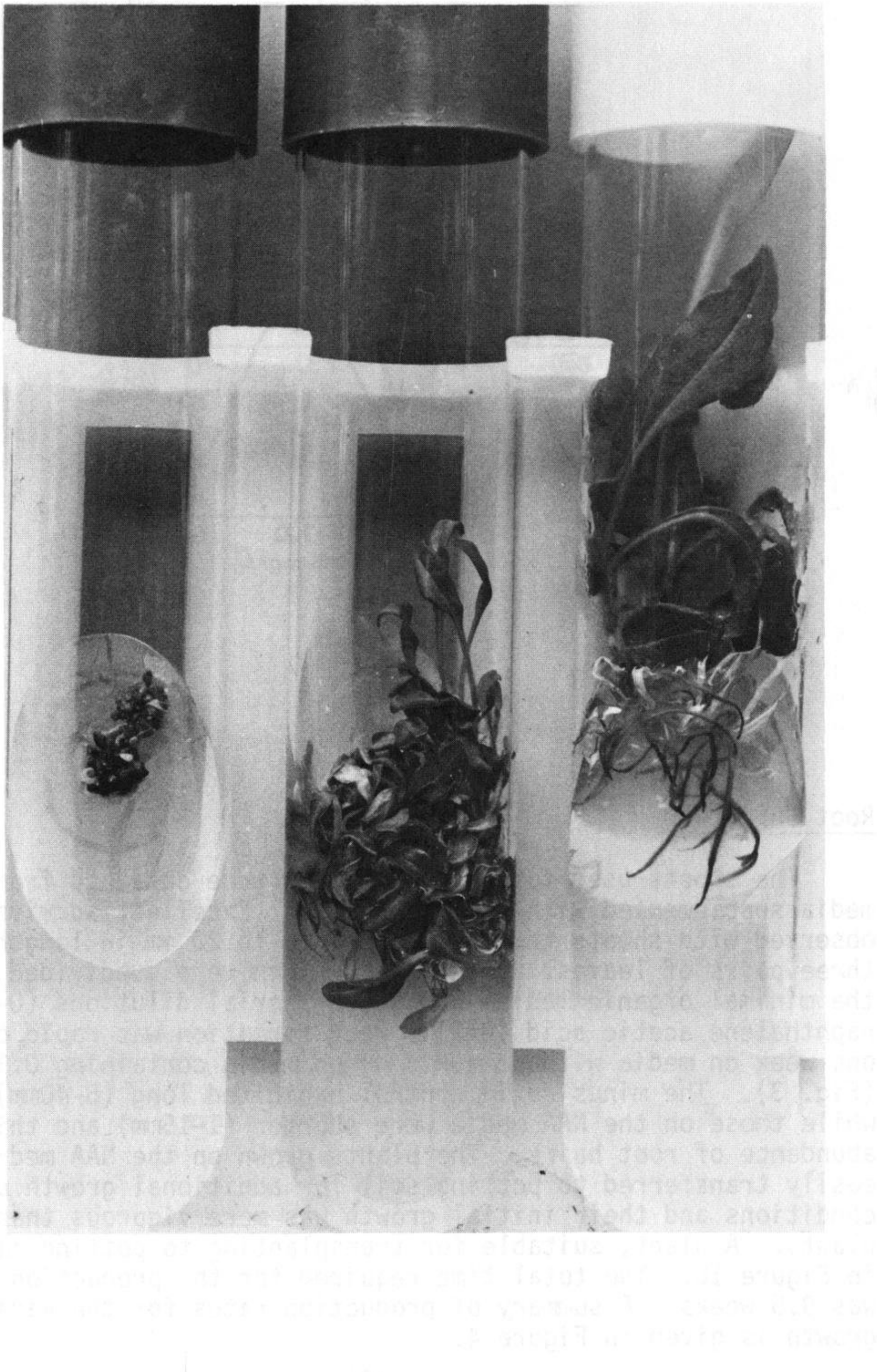
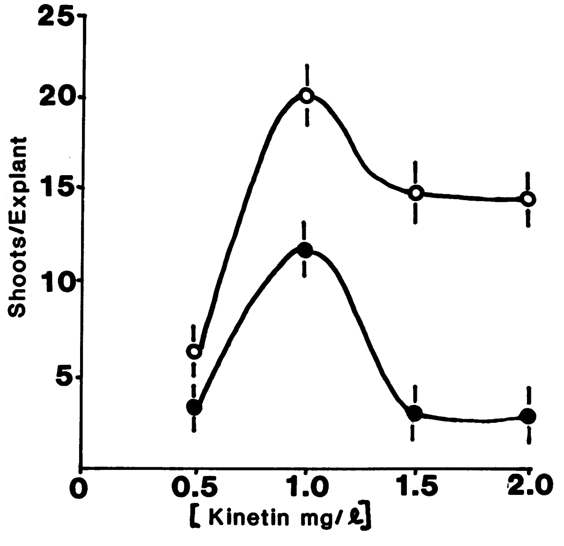


Figure 2. Adventitious shoot formation on explants grown on the basal media supplemented with 0-2.0 mg/1 kinetin.

● 26 days in culture
o 52 days in culture



Root Formation

The shoots used for root initiation were obtained from explants on media supplemented with 1 mg/1 kinetin. Excellent survival rates were observed with shoots that had reached 5 to 20 mm in length with one to three pairs of leaves. These shoot clumps were subdivided and grown on the minimal organic medium containing serial dilutions (0-1.0 mg.1) of naphthalene acetic acid (NAA). Root formation was rapid, occuring within one week on media without auxin or on media containing 0.1 mg/1 NAA (Fig. 3). The minus auxin control exhibited long (5-40mm) and thin roots while those on the NAA media were shorter (1-15mm) and thicker with an abundance of root hairs. The plants grown on the NAA media were more easily transferred to potting soil for additional growth under greenhouse conditions and their initial growth was more vigorous than the control plants. A plant, suitable for transplanting to potting soil is shown in Figure 1C. The total time required for the production of this plant was 9.5 weeks. A summary of production rates for the various stages of growth is given in Figure 4.

Figure 3. Root formation on shoots subcultured for 10 days on the basal media supplemented with 0-0.7 mg/1 NAA.

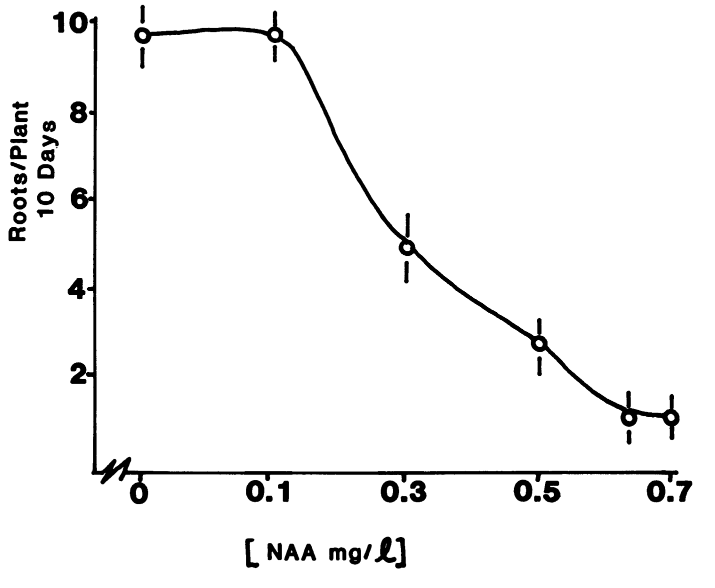
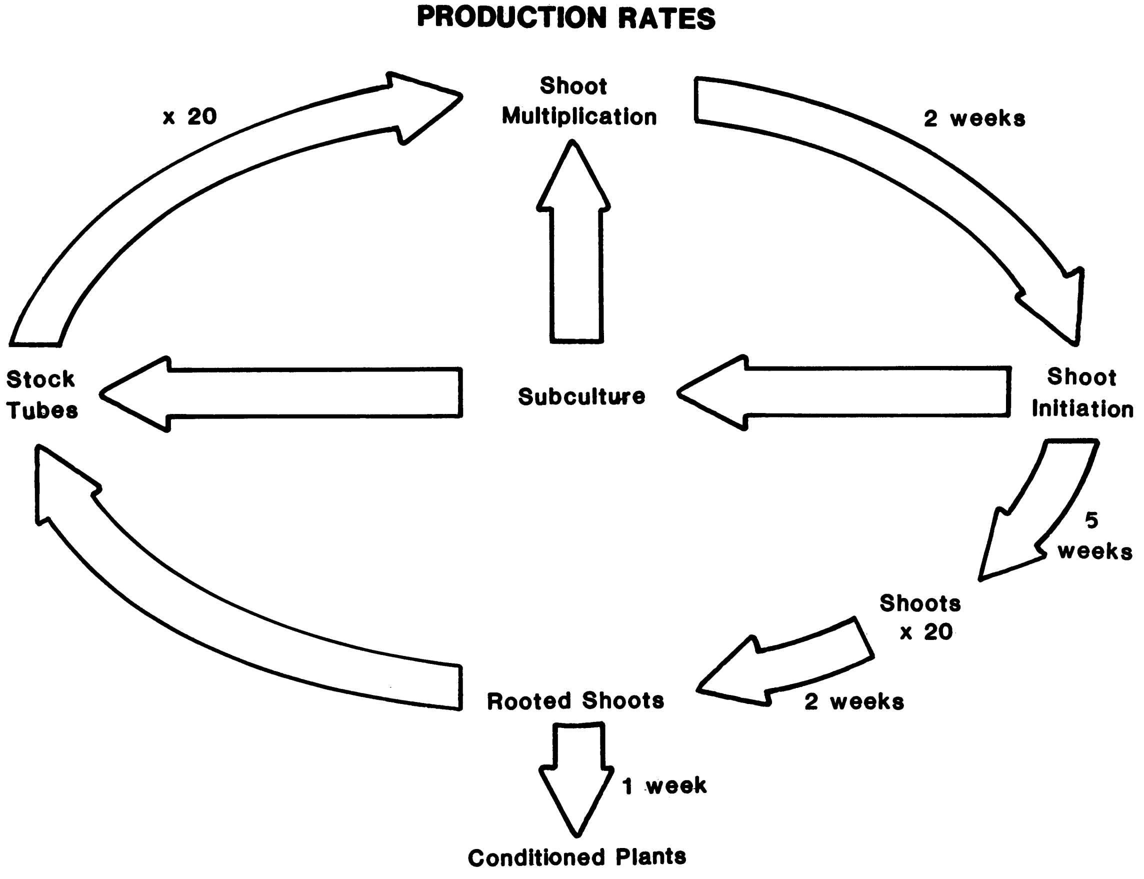


Figure 4. Summary of production stages and rates for the micropropagation of *Solidago sempervirens*.



DISCUSSION

Seaside goldenrod illustrates the utility of micropropagation for the production of plants used for dune restoration. The eight-week regeneration time for young plants from original explanted stock material is more rapid than seed propagation when one considers the time requirements for stratification or germination (Van Der Valk, 1974). The vigorous plants obtained are uniform clones of regional ecotypes which would presumably offer superior characteristics for surviving the environmental stresses specific for each region.

Traditional methods of plant propagation may not offer a cost-effective means for producing individual plants in dune restoration projects. Seed propagation requires large uniform stands of plants for economical seed harvest. Seedlings, when obtained, will often exhibit a high degree of variability with respect to stress tolerance; thus, most of the planting may be unable to survive the harsh conditions of the dune environment (Barbour, 1976). Vegetative propagation, to establish a population of genetically superior plants, is time consuming and labor-intensive; moreover, many native plant species are not amenable to the standard methods of vegetative propagation (Hartman & Kester, 1983).

In vitro micropropagation, which has been successfully applied to large numbers of agricultural and horticultural crops, has several advantages over conventional methods of plant propagation (Conger, 1982). A single stock plant may produce thousands, even millions, of geneticably identical plants depending upon the capability of the culture system. The rate of production is often faster than other means of vegetative propagation. This method circumvents in-field seed harvesting or the alternative maintenance of extensive nursery stands. Undesirable developmental stages, e.g. seed and vegetable dormancy, can often be avoided. In addition, production is easily adjusted to seasonal demand. The spacial requirements for stock maintenance in our laboratory is a section of growth chamber; yet, 6,000 plants suitable for transplanting can easily be produced from the 30 tubes of maintenance stock within a four to five month period.

I am currently examining the possibility of using *in vitro* techniques to select native plant species for increased tolerances to various environmental stresses such as high and low temperature, limited water availability, and high salinity. Such selection procedures, which are routinely used for crop improvement in agricultural sciences (Chaleff, 1983), promises to make plants more able to withstand the rigors of the dune environment. It is hoped that the inherent advantages of micropropagation, plus the ability to select far more tolerant plants less likely to need replanting, will eventually help reduce the substantial costs involved in dune restoration.

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MONITORING OF TWO ARTIFICIAL WATERWAY SYSTEMS IN JUPITER, FLORIDA

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ABSTRACT

Two artificial waterway systems were constructed in Jupiter, Florida, between 1979 and 1981. The shorelines of the two systems were planted with Spartina alterniflora and Rhizophora mangle. During 1983 and 1984 a monitoring program was conducted to measure vegetative growth rates and water quality parameters. Quarterly measurements for Spartina and Rhizaphora growth and percent cover were collected at 25 stations. Faunal samples were also collected on a quarterly basis.

Water quality parameters that were measured on a quarterly basis include dissolved oxygen, temperature, salinity, nutrients and pH. Heavy metals and protein content within the Spartina were measured annually, as were permeability and ion exchange capacity within the sediments.

In general, the Spartina densities increased by an average factor of 3.4, while Rhizophora numbers remained constant, althought the distribution shifted in favor of the higher elevations within the intertidal zone. Water quality parameters were generally within state standard levels. Benthic diversity was low and plankton numbers and diversity were high. Colonization by other intertidal plant species is also evident and the marshes and waterway host a variety of fish and invertebrates at different times of the year.

INTRODUCTION

Between 1979 and 1981, two artificial waterway systems were constructed in Jupiter, Florida, and vegatated with smooth cordgrass (Spartina alterniflora) and red mangroves (Rhizophora mangle). A two year monitoring study was conducted between February, 1983, and December, 1984, to evaluate the resultant communities and determine utilization by marine animals. Water quality samples were also analyzed during the same period to monitor chemical parameters of the water and marshes. This paper represents a brief synopsis of the study results. The complete report is on file in the offices of the Department of Environmental Regulation in Tallahassee, including sampling data and collection details.

STUDY SITES

The study sites are artificial waterway systems which are connected to the Intracoastal Waterway in Jupiter, Florida, called Admiral's Cove (Fig. 1) and Frenchman's Creek (Fig. 2).

MATERIALS AND METHODS

Eighteen study sites were selected within the Admiral's Cove project, six within the Frenchman's Creek waterways, and one in the Intracoastal Waterway as a control. A series of one-square-meter plots were studied along the centerline of each test site, between mean high water and mean low water. Spartina and Rhizophora plants were counted and measured in each plot and the percentage of cover was estimated, as well as the percentage of cover by other intertidal plant species.

Dissolved oxygen, temperature, salinity, nutrients, and pH were measured on a quarterly basis using field monitoring instruments. Benthic invertebrates and plankton were collected quarterly as well. Heavy metals and protein content within the Spartina and permeability and ion exchange capacity of the sediments were measured by Dr. Eugene Corcoran at the University of Miami.

Faunal samples within the waterways and marshes were collected with gill net, bag seine, and dragnet.

RESULTS

The original planting densities for Spartina averaged 27 plants per square meter at both locations. At the beginning of the study, the average density at both projects was about 96 Spartina plants per square meter with about 37 percent of the total coverage consisting of several species of volunteer intertidal plants. By the end of the study, the average density of the Spartina was 75 plants per square meter while the coverage by other species had increased to 45 percent. The density of Spartina at the control site remained at about 75 plants per square meter throughout the study period.

The initial planting density for Rhizophora was approximately 2.4 plants per square meter. At the beginning of the monitoring period, the density had increased to 4.1 plants per square meter due to colonization of the upper intertidal zone by propagules. By the end of the study period, however, the density was at a level similar to the original planting (2.8 plants per square meter) as the red mangroves that had been planted in the lower intertidal zone died out.

Approximately 750,000 Spartina plants were installed throughout these two projects. By the end of the study period (three years following revegetation) there were about 2,500,000 Spartina plants. Although the number of red mangroves was the same three years following planting (approximately 250,000), the distribution was markedly different from the original installation pattern.

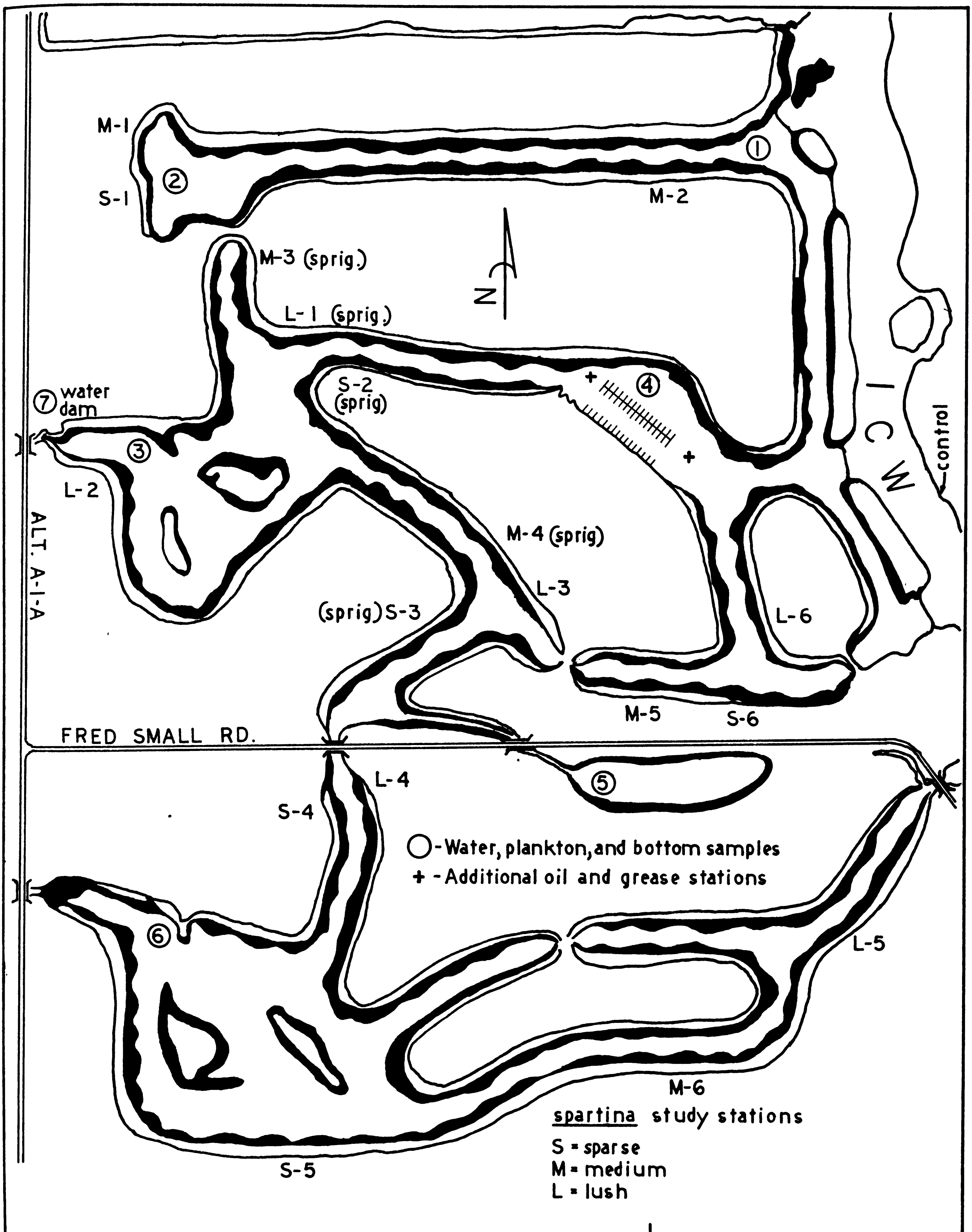

FIG. 1 ADMIRAL'S COVE

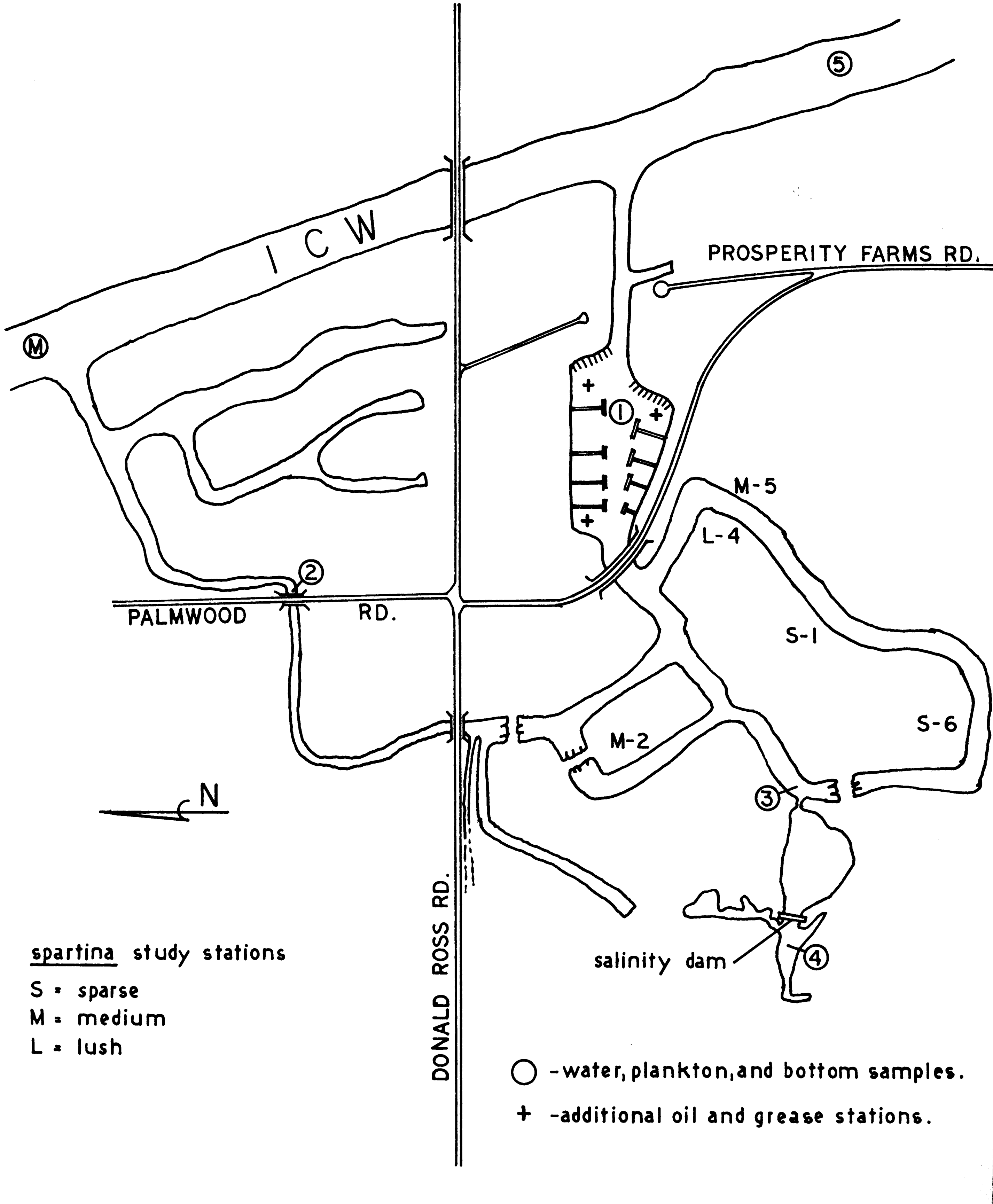

FIG. 2 FRENCHMAN'S CREEK

Although the number of Spartina plants increased by a factor of three and the number of red mangroves remained relatively constant, those two species only comprise 55 percent of the total intertidal community. The volunteer species are listed below, in order of abundance:

White mangrove—Laguncularia racemosa
Sea purselane—Sesuvium portulacastrum
Seashore saltgrass—Distichilis spicata
Salt jointgrass—Paspallum vaginatum
Glasswort—Salicornia virginica
Black mangrove—Avicennia germinans
Saltwort—Batis maritima
Sea daisy—Borrichia frutescens
Cattail—Typha spp.
Needlerush—Juncus roemerianus

The nekton sampling within the water column of the artifical systems indicated the following utilization patterns:

Snook—Centropomus undecimalis C. parallelus C. ensiferus C. pectinatus	Summer, fall, and winter
Anchovy—Anchoa mitchelli A. hepsetus	Summer and fall
Spadefish—Chaetodipterus faber	Summer and fall
Sandbar Shark—Carcharhinus milberti	Summer
Menhaden—Brevoortia tyrannus	Summer
Barracuda—Syphyre ana barracuda	Summer
Mojarra—Eucinostomus gula E. argenteus	Winter and spring
Silversides—Menidia berylina	Fall
Lookdown—Selene vomer	Winter
Pinfish—Lagodon rhomboides	Spring

Shrimp (Penaeus aztecus and P. setiferus), blue crabs (Callinectes sapidus), mullet (Mugil curema and M. cephalus), sea catfish (Arius felis), and croakers (Micropogon undulatus) appear to occur in significant numbers throughout the year.

In the Spartina marshes, anchovies, grass shrimp (Palaemonetes spp.), amphipods (Gammaropsis sp.), and salt marsh crabs (Sesarma reticulata) appear to occur throught the year, while juvenile blue crabs (Callinectes sapidus) and shrimp (Penaeus setiferus) occur during the fall and winter.

The results of the water quality monitoring for selected areas within the system are delineated in Tables 1 and 2. The benthic invertebrates and phytoplankton collections are also summarized in Tables 3 and 4, respectively.

DISCUSSIONS AND CONCLUSIONS

It appears that the planted Spartina and Rhizophora exhibit an initial proliferation followed by a reduction to a stable population, colonizing the appropriate elevation zone for each species. Other areas within the intertidal zone are eventually colonized by other intertidal plants, creating a more diverse system than that which was originally installed. A consistent factor in determining the relative health of Spartina growth seems to be the depth of the root hairs. Root hairs at or near the surface of the substrate correlate with a healthy growth pattern while exposed or buried root hairs indicate stressed conditions due to erosion or accretion with resultant density decreases.

The waterways and marshes within the artificial system appear to support the same faunal populations as nearby natural areas sampled, benthic faunal communities and phytoplankton also are similar to adjacent Intracoastal populations.

The most important factor relative to successful wetland establishment seems to be the creation of proper elevations relative to tidal fluctuations. Nearby intertidal marshes should be surveyed to determine the proper elevations for each species that is included in the revegetation program. The substrate type (with the exception of hardpan clay materials) does not seem to be a major factor in the successful establishment of a saltmarsh community.

ACKNOWLEDGMENTS

The authors wish to express their appreciation to Robert Snyder of Snyder Oceanographic Services, Inc., who designed these two systems, for his support and helpful suggestions throughout the study, and to Russell Beilenberg and Don Shackelford of the Bankers Land Company for their cooperation and consideration in funding this study and supporting the advancement of our knowledge relative to the creation of functional wetland systems.

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TABLE I

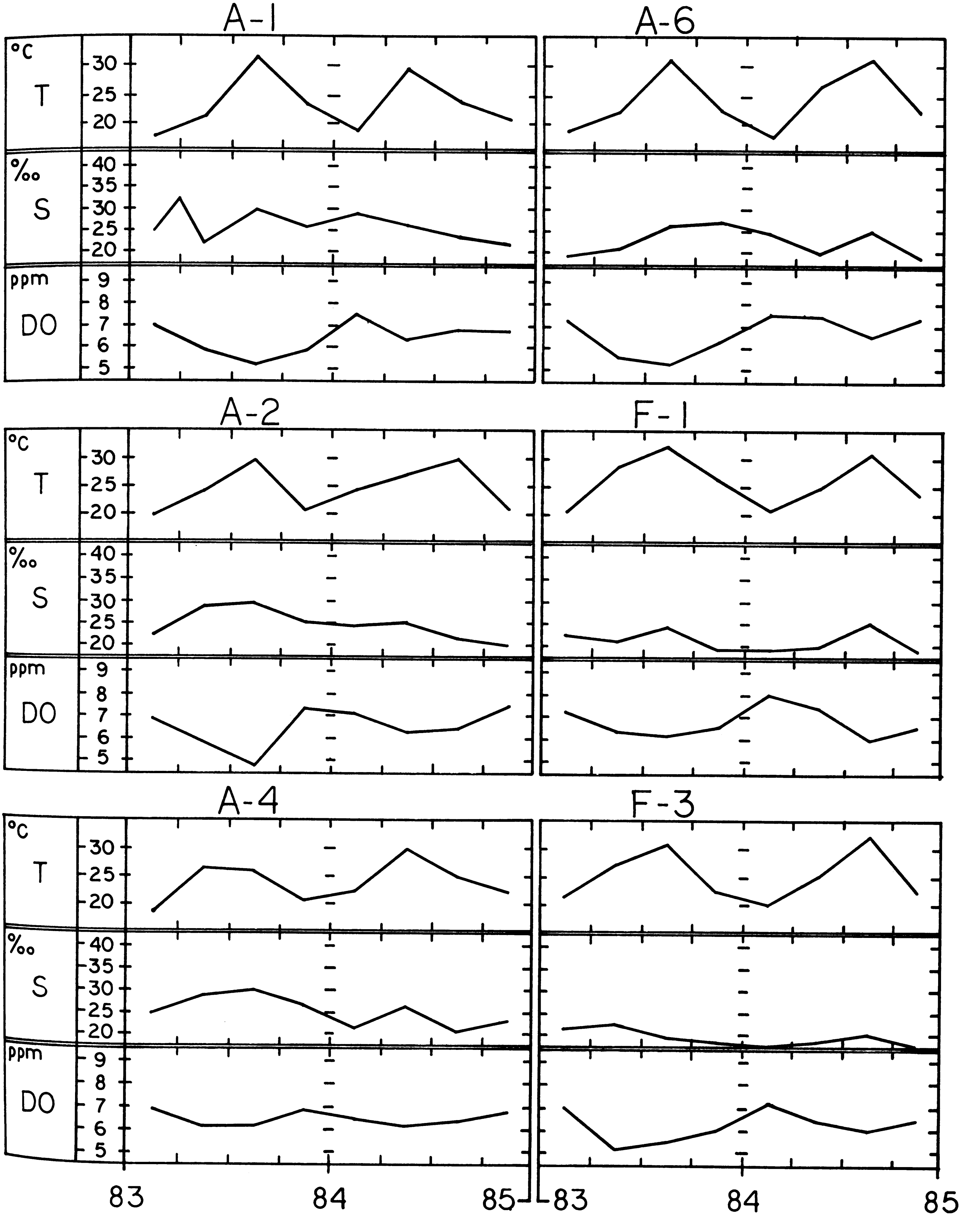

TABLE 2

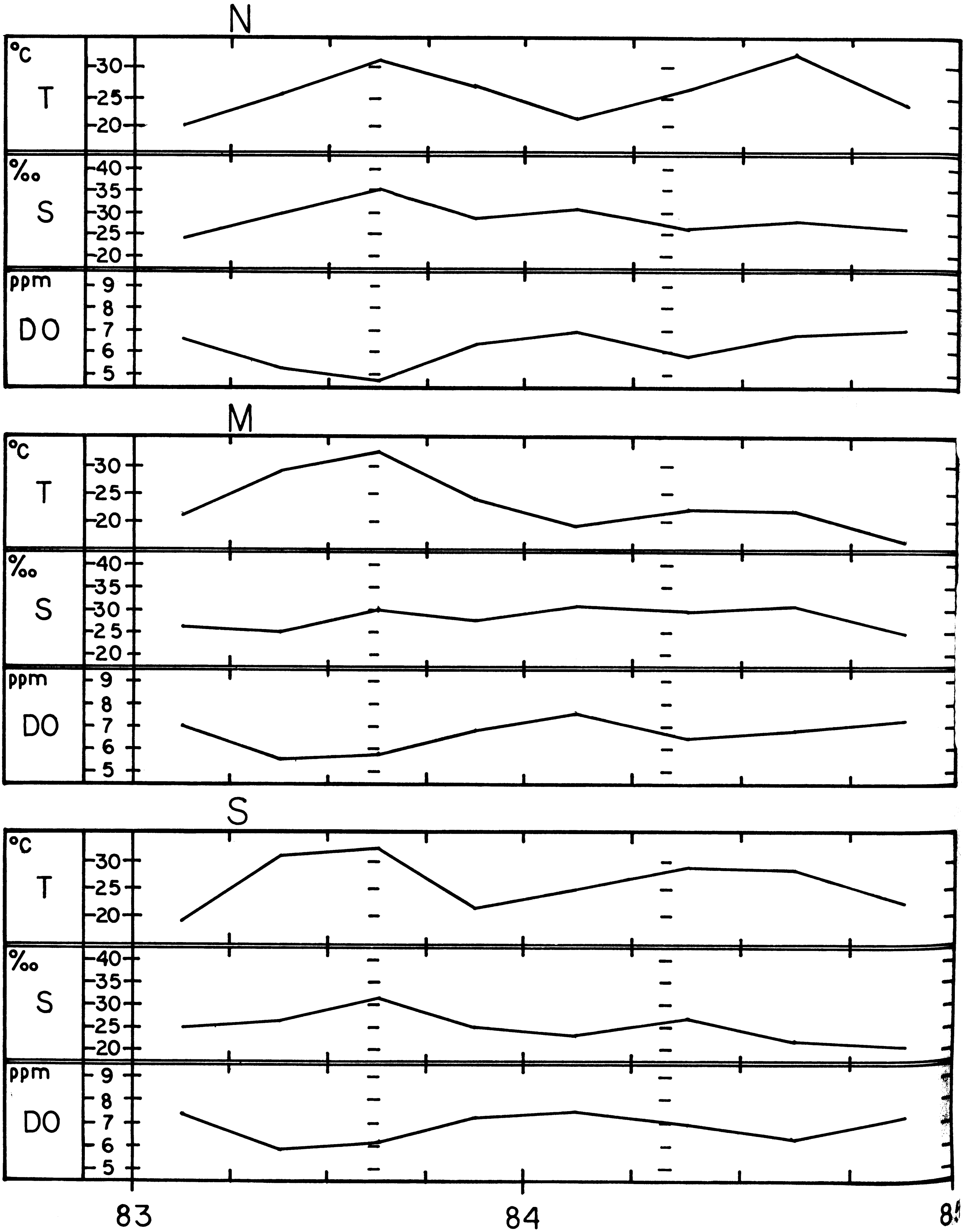
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Table 3

Benthic Invertebrate

Diversity Indices

Date	A-1	A-2	A-4	A-6	F-1	F-3	N	M	S
2/83	2.61	3.03	2.47	-	-	1.99	3.03	3.47	3.54
4/83	-	-	2.20	2.77	2.41	2.07	3.14	-	2.86
6/83	3.04	3.01	2.80	2.61	3.27	1.58	3.15	2.73	2.16
10/83	3.21	2.50	1.94	3.44	-	1.43	1.88	2.28	3.55
1/25	3.99	3.68	3.74	2.94	2.38	-	4.12	4.01	-
5/84	2.73	1.71	3.69	2.32	3.11	1.64	3.37	1.45	3.07
8/84	1.72	3.19	2.27	2.66	2.10	2.73	3.19	3.24	3.07
12/84	2.23	3.42	2.99	1.93	1.22	2.69	1.87	1.91	-
average no. of species	8	10	12	8	9	7	12	9	6

Table 4

Total Phytoplankton

At Noon mm^3/l

Date	A-1	A-2	A-4	A-6	F.1	F.3	N	M	S
2/83	39.1	60.4	93.8	51.0	14.2	28.9	26.8	23.1	23.5
4/83	2.94	9.0	16.3	41.2	16.4	8.3	23.3	14.8	7.0
6/83	17.3	14.7	8.1	17.3	21.3	3.9	12.0	44.4	53.8
10/83	23.6	16.8	19.1	32.7	12.7	4.4	32.9	27.3	20.4
1/25	38.4	62.9	36.0	49.1	30.3	13.0	52.0	40.9	87.9
5/84	46.6	78.3	112.1	50.8	42.7	21.6	73.4	44.1	39.1
8/84	22.9	12.5	8.1	5.8	8.7	29.8	26.9	28.3	20.0
12/84	91.9	83.0	109.2	54.7	94.1	34.8	97.2	134.0	130.7

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SALT MARSH IMPOUNDMENT MANAGEMENT ON FLORIDA'S CENTRAL EAST COAST: REINTEGRATING ISOLATED HIGH MARSHES TO THE ESTUARY

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ABSTRACT

In Florida, management methods in salt marsh impoundments which benefit mosquito control, fish and wildlife resources, and water quality enhancement are encouraged. Therefore, to provide quantitative management information, the water management regime of (1) opening a 20.2 ha. southeast Florida impoundment to the adjacent estuary with culverts through the dike, then (2) passively retaining water with flapgate risers has been studied to determine the effects on marsh flooding, vegatation, fish, macro-crustaceans, zooplankton, water quality, and mosquitoes.

The common mosquito control schedule of closing culverts in the early spring, retaining water during the spring and summer, then reopening the marsh on the fall high tides appears basically compatible with the major periods of fish ingress and egress. Significant regrowth of high marsh vegetation has occurred after reopening the impoundment to the estuary. The impoundment plankton fauna appears similar to that of other shallow water systems in Florida with copepods being the numerically-dominant group. With the present water-control-structure configuration (i.e., two 45.7 cm diameter culverts), passive water retention with flapgate risers to 1.0 feet NGVD has, so far, not been permanently detrimental to existing vegetation, but large mosquito broods were produced from rainfall and tidal flooding. Supplemental pumping will be necessary to provide adequate "source reduction" mosquito control benefits.

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[1]Authorship sequence determined by alphabetical order.

INTRODUCTION

Salt-marsh impoundments in Florida's central east coast are high marshes which were surrounded by dikes in the 1950's and 1960's and which are artificially flooded by pumping water from the adjacent estuary or with artesian wells during the mosquito producing season (approximately May-October). The salt-marsh mosquitoes (Aedes taeniorhynchus (Wiedemann) and Ae. sollicitans (Walker) lay their eggs in moist substrates, but will not oviposit upon standing water; the eggs hatch when they are flooded by tidal waters or by rainfall. Impounding, therefore, prevents these species from ovipositing in what otherwise would be highly attractive sites. This "source reduction" technique is both effective and economical in reducing populations of these mosquitoes (Clements & Rogers, 1964, Provost, 1977).

When most Florida impoundments were constructed, they were managed primarily for mosquito control and in some locations for waterfowl enhancement. Since then, research has now shown that impounding can interrupt the marsh-estuary exchange of organisms and detritus and kill high marsh vegetation if excessively flooded for prolonged periods (Gilmore et al., 1982).

Now that the importance of the high marsh in the lagoon system is recognized, organizations responsible for wetlands resources are encouraging management of these habitats in ways that minimize deleterious effects to wildlife and water quality while controlling salt-marsh mosquitoes. The ultimate goal is to enhance conditions for the former, without compromising the effectiveness of the latter. However, until very recently, studies on the effects of different management strategies on salt marsh flora, fauna, and physical conditions have been almost non-existent. Thus, management plans for impounded marshes have had to be developed with litte concrete information on their possible results.

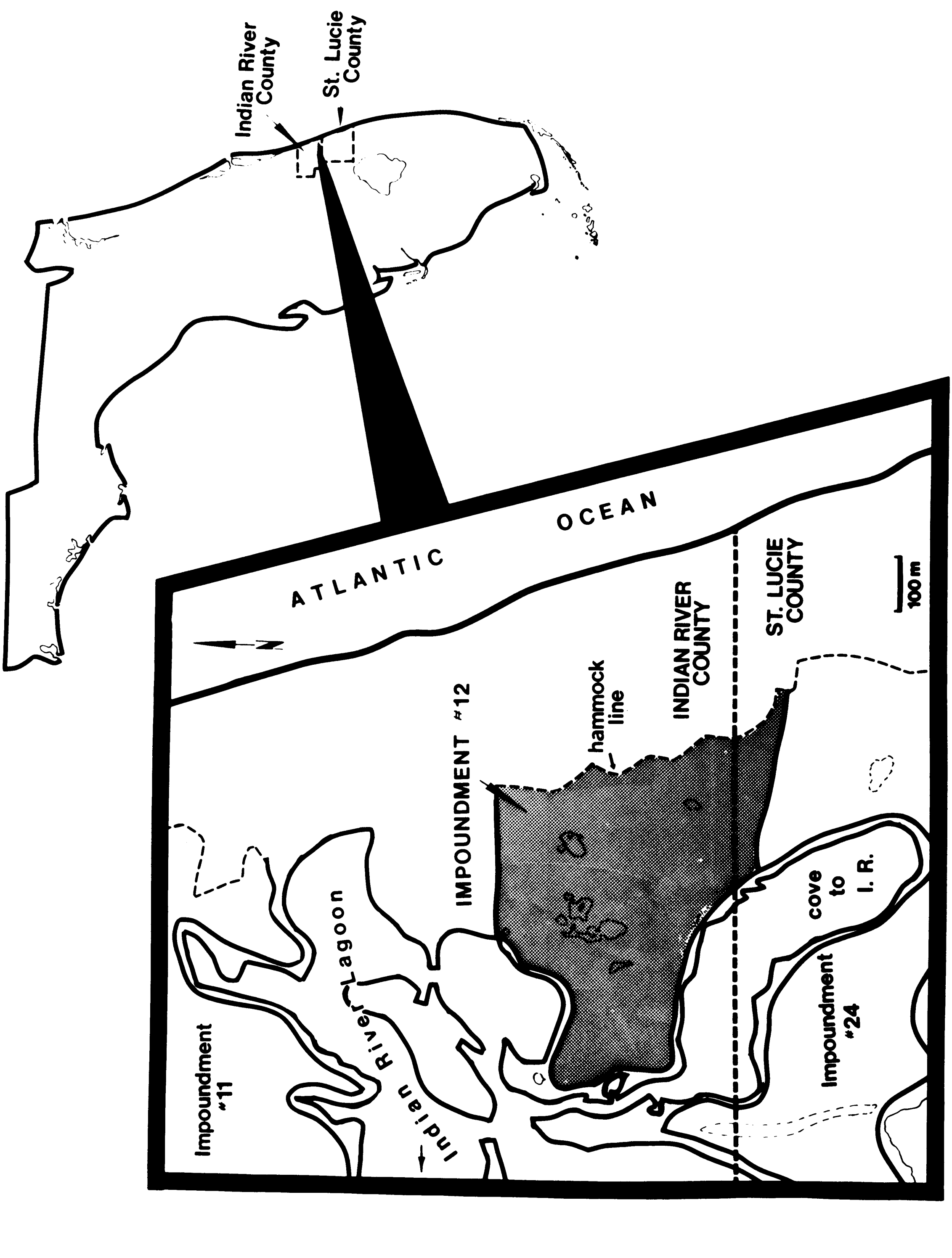
In an attempt to provide information that would be applicable to salt marsh impoundment management, a cooperative research project was started in 1982 between the Indian River Mosquito Control District, The Florida Medical Entomology Laboratory (University of Florida—IFAS), and the Harbor Branch Foundation, Inc. The project is investigating the effects on vegetation, fish and macrocrustaceans, zooplankton, mosquitoes, and water quality of reintegrating an isolated impounded high marsh to the Indian River lagoon system by first establishing connections through culverts through the dikes, and then passively retaining water with flapgate risers.

SITE DESCRIPTION

The salt-marsh mosquito control impoundment used for this study is Indian River Impoundment #12 (Bidlingmayer & McCoy, 1978). This 20.2 ha. high marsh is located on the barrier island at the boundary between Indian River and St. Lucie counties. A shallow cove, which is part of the Indian River lagoon, lies southwest of the site (Fig. 1).

The impoundment was constructed in March, 1966, and was seasonally

Figure 1. Study site location.



flooded (approximately May-October) with water pumped from the Indian River lagoon until 1978 when pumping was discontinued at a property owner's insistence. From 1978 to 1982 water levels were allowed to fluctuate depending upon rainfall, evaporation, and percolation.

Most of the north and east sides of the marsh are bounded by an undiked upland edge, whereas the remainder of the impoundment is delimited by a man-made dike and perimeter ditch. The perimeter ditch ranges in width from one to eight meters, and is up to two meters deep, but many portions are filled with mud and organic debris. Well-defined drainage patterns are evident from the interior of the impoundment to the perimeter ditch. Numerous large depressions occur over the marsh surface, many of which retain water even during extremely dry periods thus forming permanent or semi-permanent ponds. Elevations of the marsh surface (excluding the perimeter ditch which is up to two meters deep) range from -0.35 to 1.80 feet (NGVD) (National Geodetic Vertical Datum) but the majority of the marsh ranges from 0.40 to 0.90 feet NGVD.

Presently, the most common plant species on the marsh are saltwort (*Batis maritima* L.), annual glasswort (*Salicornia bigelovii* Torr.), and perennial glassworth (*S. virginica* L.). Black mangroves (*Avicennia germinans* (L.)), red mangroves (*Rhizophora mangle* L.), and white mangroves (*Laguncularia racemosa* Gaertn.) are widely dispersed with the greatest regrowth along the perimeter ditch. *Ruppia maritima* L. (widgeongrass) is often very abundant in the permanent and semi-permanent ponds in the interior of the marsh.

An adjoining impoundment, St. Lucie County Impoundment #24, was used as a control for the zooplankton, vegetation, and water quality portions of this study. It is similar in nature to the experimental impoundment, but it has remained isolated from the Indian River throughout the study period.

METHODS

Scope

One of the major objectives of this study was to simultaneously obtain and integrate information on a variety of components of the marsh-lagoon system. To this end, there has been close cooperation during the design and implementation of the project between the three organizations involved. Nevertheless, the separate portions of this study have been carried in a semi-independent manner by the entities involved. The specific areas investigated are: mosquito production—Indian River Mosquito Control District; zooplankton, water quality, and vegetation—Florida Medical Entomology Laboratory; fish and macrocrustaceans—The Harbor Branch Foundation, Inc. In addition, a number of physical parameters have been monitored throughout the study by the three organizations.

Water Management

This study commenced in February, 1982, at which time a 45.7 cm (18 in.) culvert was opened to allow free exchange of water with the Indian River lagoon. In July, 1983, a flapgate riser was attached to the culvert and set at 1.0 feet NGVD (Fig. 2). The function of the riser is to trap rainfall and tidal waters within the marsh up to the set level, but allow any excess to spill over the riser and escape into the lagoon. In late September, 1983, an additional 45.7 cm culvert was placed to enhance marsh-lagoon interchange. In late January the flapgate risers were removed and unrestricted flow through the culverts was re-established.

Water Levels

Rain data were collected twice weekly with a tube range gauge. Bi-weekly maximum and minimum water levels at several locations within the marsh and in the Indian River were recorded with a greased staff-float apparatus.

Ichthyofauna and Macrocrustaceans

To obtain qualitative and quantitative information on the impoundment ichthyofauna, eight different types of sampling gear were used every two weeks over a 24 hour period to obtain a complete tidal and diel analysis. Different gear types were necessary to appropriately sample different microhabitats within the impoundment and because the organisms sampled are highly mobile and easily conditioned. The gear types included three static traps and five mobile traps. They were:

Static traps.

1. Heart trap—an aluminum frame adjustable aperture (to 35 mm), 3.2 mm ace weave mesh 0.62 X 0.78 m, 0.63 deep heart trap was used to capture fishes and macrocrustaceans moving through shallow areas extending from the marsh interior to the perimeter ditch such as tidal creeks or rivulets.

2. Culvert trap—this 1.52 m long, 44 cm diameter trap was specifically designed to trap organisms passing through the culvert.

3. Culvert net—a 1.7 X 1.0 X 1.3 m, 3.2 mm mesh bait box net was modified to fish the water exiting the culvert.

Mobile traps.

1. Throw net—a 1.0 m^2 throw net was used to determine density and biomass samples in upper marsh ponds.

2. & 3. Seine nets—both 3.08 m, 3.2 mm and 15.2 m, 3.2 mm ace mesh bag seines were pulled over measured distances at various impoundment and adjacent cove locations.

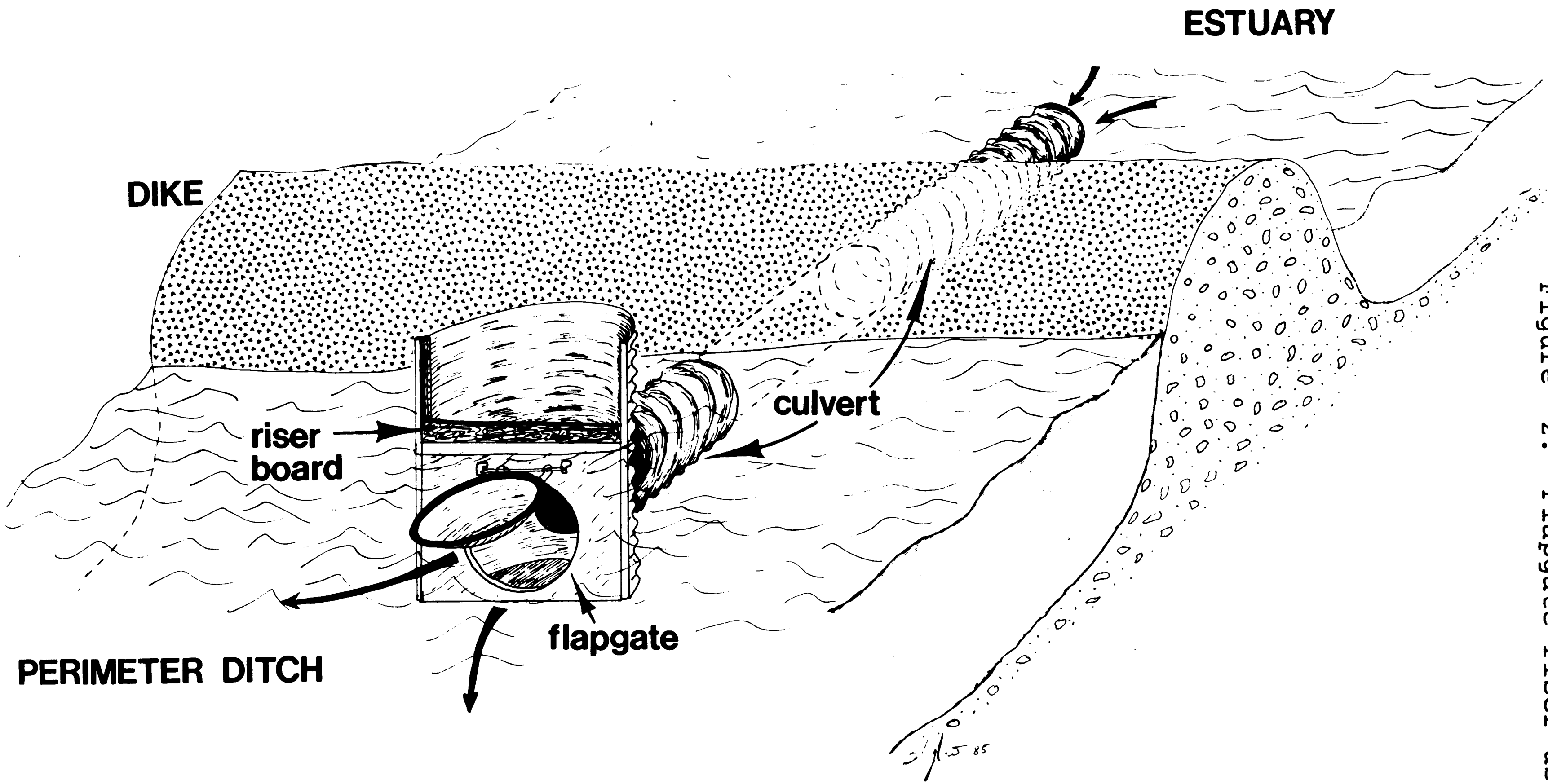


Figure 2. Flapgate riser used in study.

4. Pull net—a 2 m X 5.65 m, 3.2 mm ace weave pull net was used to sample the perimeter ditch by pulling it over a measured distance.

5. Cast net—a 2.8 m radius, 2.5 mm mesh cast net was used to sample deep interior marsh locations.

Mosquitoes

The entire marsh surface was divided into 12 quadrats and at least twice weekly, mosquitoes were sought out in all quadrats. When immature mosquitoes (larvae or pupae) were found, mosquito broods (a group of immature mosquitoes in a sampling area which hatch and mature concurrently) were randomly sampled by taking 5-350 ml dips per quadrat. A mean size over the brood duration was determined.

Vegetation

Vegetation in the experimental and control impoundments was monitored along 1200 foot transects established across the marsh. Each transect was subdivided into 100 foot sections and five quadrat locations were established at random points within each section. At each location, the percent cover by each species was measured at quarterly intervals using a half-meter-square portable grid. The growth and survival of mangroves at the experimental and control impoundments were also monitored at quarterly intervals. One hundred seedlings were marked at each location and their growth and viability were recorded at each sampling.

In January, 1984, the Indian River Mosquito Control District conducted a visual survey of the vegetative cover of the experimental impoundment and recorded the qualitative frequency and abundance of the various species.

Zooplankton

Sampling for plankton in shallow waters is difficult because standard circular nets usually drag the bottom quickly clogging the meshes and contaminating the samples. Pump sampling in such habitats also has been generally ineffective because standard sieves become clogged before adequate samples can be collected and filtered.

We developed several techniques for sampling plankton in such marsh habitats that allowed us to obtain adequate sample sizes with a minimum of bottom contamination. These techniques included floating, rectangular nets, a large-volume filtering apparatus for pump samples, and hand nets. A total of 603 samples were taken at 4 stations, using these methods.

Water Quality

Twelve water quality sampling stations were established at strategic locations within the experimental and control impoundments, and in the Indian River. Sampling was started on September 25, 1984, and is being conducted on a 24-hour cycle at bi-weekly intervals. During each sampling, the following variables are measured and recorded at each site: dissolved oxygen, pH, water temperature, and salinity. A water sample is also collected at each site and taken back to the laboratory for further analysis. In the laboratory, nutrient analyses are carried out using a Technicon II Autoanalyzer system, and dissolved solids and tannin-lignin determinations with a Myron-L 512T3 DS meter, and a HACH TA-3 Test Kit, respectively. Results of the water quality sampling and analyses are still too preliminary to yield meaningful conclusions although some interesting patterns and differences between sites are already beginning to emerge.

RESULTS

Field work for this study is still under way; below we report on some of our preliminary findings and their possible management implications. A comprehensive report of our results will be produced upon completion of the project.

Marsh Water Level

Seasonal tidal effects played the greates role in setting the marsh water levels, with rainfall playing a smaller, but significant role. During both years the fall increase in water levels caused corresponding fluctuations within the marsh. Total coverage of the marsh usually occurred in early September, and persisted for several months. Water level measurements in the impoundment and also in the lagoon indicate that a standing water head was maintained within the marsh; that is, marsh water levels exceeded lagoon levels, and daily fluctuations outside the impoundment were usually greater than those inside.

Since the general marsh countour consists of gradual increase in elevations from the perimeter ditch (apprxoimately 0.40 ft. NGVD) to the upland edge (approximately 0.90 ft. NGVD) water levels below 0.45 feet NGVD generally flood only the perimeter ditch. However, rainfall can form isolated pockets of water away from the ditch which are not detected by the water level recorders (pers. obs.). Water levels of 0.60 feet NGVD flood as much of the western half of the impoundment and a flooding level of 0.90 feet NGVD or greater inundate the marsh to the eastern edge.

Ichthyofauna and Macrocrustaceans

1. General collection information. Marsh fish research originally began at the Impoundment #12 study site in January, 1979, when the impoundment was not open to the estuary nor managed for mosquito control. The fish fauna was compared to the fauna of another impoundment (Impoundment #23, St. Lucie County) which was open to tidal influence through a single culvert. The open impoundment (#23) was found to possess a far richer ichthyofauna (i.e., at least 30 additional species, Gilmore et al., 1982). Subsequent to this initial study, we demonstrated that when the closed impoundment site (#12) was reopened to tidal influence, considerable faunal changes occured with a major increase in species richness, i.e.,

from 12 to 45 species of macrocrustaceans and fishes.

Over 2,006 collections made from February, 1982, to April, 1985, in Impoundment #12 and two additional study sites have captured 465,310 individuals (449.5 kg—wet weight) of macrocrustaceans and fish representing 103 species (Table 1). Only 21 of these species, 41,215 individuals (7.3 kg), were crustaceans, contributing only 9 percent and 2 percent of the total number and weight of organisms collected, respectively. Therefore, the 82 fish species collected make up 91 percent of the total numerical catch and 98 percent of the sample weight, demonstrating the major contribution of the ichthyofauna to the marsh faunal biomass. It should be noted that the major marsh macrocrustaceans missed by these gear types are the burrowing crabs, Uca spp and Cardisoma guanhumi which add considerable biomass if sampled.

Fourteen fish species (totalling 95% of the total catch) are marsh residents which can reproduce within the marsh. Of the 89 transient species encountered, 74 are considered ephemeral migrators. Fourteen transients that depend on the marsh for a portion of their life cycle were collected. Twenty of the transient species captured are of commercial or sport fishery value and all of these spawn in open estuarine, neritic, or pelagic habitats.

2. Microhabitat faunal comparisons. For comparison purposes, the impoundment habitat was classified into (1) the lower marsh (=perimeter ditch) with the rest of the marsh being defined (2) upper marsh. The resident and transient fauna was more speciose in the lower versus upper marsh. The resident fauna was richer than the transient fauna on the upper marsh. More species of residents and transients occur outside the impoundment than on the upper marsh.

The transient fauna is richer than the resident fauna in the lower marsh from July to late November with this trend reversing from March to early July. Typically the largest catch of transient species took place in the culvert trap as migration into or out of the marsh required passage through this water control structure. Major transient species were recruited around adult spawning seasons. Primary immigration into the impoundment occurred with periods of sea level rise (May-June, August-October). Although some emigration occurred in June, most emigration of transient species takes place during the late fall and winter months as sea levels fall.

Mosquitoes

Mosquito sampling demonstrated that explosive, synchronous mosquito production triggered by rainfall and tidal flooding is possible in revegetating impoundments. The vast majority of mosquito broods (65 out of 75) were triggered by rainfall in the spring and summer, with as many as 1,444 larvae collected in one 350 ml dip. Mosquito production differed between some sampling quadrats and occurred at elevations ranging from 0.25-0.90 feet NGVD. Mosquito production during the first two years of study ranged from highs of 17 broods (with a mean brood size of 65.9

Table 1. Fish and macrocrustaceans collected from three impounded marsh sites (including Impoundment No. 12) from February 1982 to May 1985. Species are ranked by numerical abundance. Also given are weight (g) and specific relative occurrence (i.e., number of occurrences out of 2,006 samples taken). The numbers in parentheses indicate the percentage of total number or weight represented by the specific species.

Common Name	Species	Total Number	Total Weight	Specific Relative Occurrence
Sheepshead minnow	Cyprinodon variegatus	188480 (40.51)	129579.22 (28.83)	33.10%
Mosquitofish	Gambusia affinis	141237 (30.35)	24539.99 (5.46)	43.27%
Sailfin molly	Poecilia latipinna	63642 (13.68)	41760.42 (9.29)	36.24%
Grass shrimp	Palaemonetes spp	38566 (8.29)	4818.27 (1.07)	39.63%
Striped mullet	Mugil cephalus	5451 (1.17)	120929.49 (26.90)	23.53%
Ladyfish	Elops saurus	5238 (1.13)	9418.07 (2.10)	15.80%
Snook	Centropomus undecimalis	4752 (1.08)	13918.96 (3.11)	16.00%
White mullet	Mugil curema	2665 (0.57)	36427.69 (8.10)	12.91%
Marsh killifish	Fundulus confluentus	2499 (0.54)	2796.00 (0.62)	10.47%
Blue Crab	Callinectes sapidus	1364 (0.29)	25668.89 (6.00)	13.00%
Inland silverside	Menidia beryllina	1253 (0.27)	890.54 (0.20)	2.39%
Shrimp	Penaeus spp	1247 (0.27)	2439.89 (0.54)	13.75%
Irish pompano	Diapterus auratus	1054 (0.23)	1684.36 (0.38)	8.90%
Rainwater killifish	Lucania parva	951 (0.20%)	287.60 (0.06)	3.29%
Yellowfin mojarra	Gerres cinereus	740 (0.16)	650.71 (0.14)	6.68%
Tidewater mojarra	Eucinostomus harengulus	721 (0.15)	862.58 (0.16)	3.90%
Bay anchovy	Anchoa mitchilli	707 (0.15)	1711.91 (0.38)	2.29%
Silverside	Menidia spp	393 (0.08)	159.13 (0.04)	3.49%
Spot	Leiostomus xanthurus	383 (0.08)	721.80 (0.16)	2.24%
Silver jenny	Eucinostomus gula	350 (0.08)	1345.22 (0.30)	0.65%
Tarpon	Megalops atlanticus	343 (0.07)	6283.00 (1.40)	2.79%
Tidewater silverside	Menidia peninsulae	333 (0.07)	273.27 (0.06)	1.35%
Gulf killifish	Fundulus grandis	325 (0.07)	756.80 (0.17)	3.49%
Fat sleeper	Dormitator maculatus	321 (0.07)	2203.85 (0.49)	6.53%
Clown goby	Microgobius gulosus	258 (0.06)	46.12 (0.01)	1.45%
Code goby	Gobiosoma robustum	248 (0.05)	40.58 (0.01)	1.94%
Mojarras	Eucinostomus spp	233 (0.05)	17.88 (0.00)	1.60%
Pinfish	Lagodon rhomboides	212 (0.05)	4457.03 (0.99)	1.30%
Lined sole	Achirus lineatus	184 (0.04)	70.84 (0.02)	2.96%

Common Name	Species	Total Number	Total Weight	Specific Relative Occurrence
Black drum	Pogonias cromis	172 (0.04)	77.81 (0.02)	0.85%
Menhaden	Brevoortia spp	160 (0.03)	42.35 (0.01)	1.20%
Croaker	Micropogonias undulatus	79 (0.02)	96.01 (0.02)	1.20%
Killifishes	Fundulus spp	77 (0.02)	5.41 (0.00)	1.30%
Gray snapper	Lutjanus griseus	73 (0.02)	4228.52 (0.94)	2.69%
Gulf pipefish	Syngnathus scovelli	68 (0.01)	14.87 (0.00)	1.50%
Frillfin goby	Bathygobius soporator	47 (0.01)	126.74 (0.03)	1.84%
Sheepshead	Archosargus probatocephalus	39 (0.01)	2746.01 (0.61)	1.25%
Mangrove crab	Aratus pisonii	38 (0.01)	13.76 (0.00)	1.50%
Lyre goby	Evorthodus lyricus	33 (0.01)	90.05 (0.02)	0.55%
Sailors choice	Haemulon parrai	33 (0.01)	149.46 (0.03)	0.40%
Great barracuda	Sphyraena barracuda	31 (0.01)	169.33 (0.04)	1.30%
67 additional species were collected		310 (<0.01)	7004.75 (<0.02)	-
Total		465310	449525.18	

larvae/dip) in an east quadrat and 15 broods (with a mean brood size of 34.1) in a west quadrat to 0 broods in the three south quadrats. Of the mosquitoes collected, 82 percent were Aedes taeniorhynchus with Ae. sollicitans comprising 16 percent of the total sample.

Trapping of rainfall and tides with flapgate risers aided in eliminating oviposition sites but still allowed mosquito production in some marsh locations. Tidal flooding permitted larvivorous fish access to mosquito larvae, but these fish were unable to provide adequate control benefits to eliminate larviciding (Carlson & Vigliano, 1985).

Vegetation

Considerable revegetation by B. maritima, S. bigelovii, and S. virginica has occurred since the marsh was opened to the Indian River (Fig. 3). All three species of mangroves are growing well in the closed impoundment (control—Impoundment #24) (Fig. 4); in fact, to a much greater extent than in the experimental impoundment (#12). Considerable mortality of seedlings was evident in the open impoundment. Results of the transect survey show small changes in percent cover by other plant species during the first year of the study.

Zooplankton

As expected, copepods were the numerically-dominant group of organisms collected in the plankton samples. On a coarse scale, the plankton fauna of these marshes appear to be similar to that of other shallow water systems in Florida except for a somewhat greater representation of primarily-benthic species, specially harpacticoid copepods, in the impoundment samples.

Preliminary analysis of the diversity and abundance data indicate that increased site isolation correlated with lower plankton diversity and higher individual abundances.

DISCUSSION

On Florida's central east coast, many impoundments have lacked tidal connection since they were constructed. However, concern over the possible deleterious effects of this marsh-lagoon isolation has recently resulted in an increased emphasis on developing multipurpose management strategies for these coastal impoundments. The creation of the Technical Subcommittee on Managed Salt Marshes, a subcommittee of the Governor's Working Group on Mosquito Control, is an example of this renewed emphasis on wise management of these valuable resources. The Technical Subcommittee, which was formed in 1983 to serve as a forum to integrate the numerous special management interests in impoundments, consists of 13 representatives from: (1) governmental agencies responsible for wetlands resources, (2) research institutions involved in salt marsh wetlands research, and (3) mosquito control agencies. It is responsible for reviewing the technical aspects of impoundment management plans, and of serving as an information gathering and dissemination source on the subject.

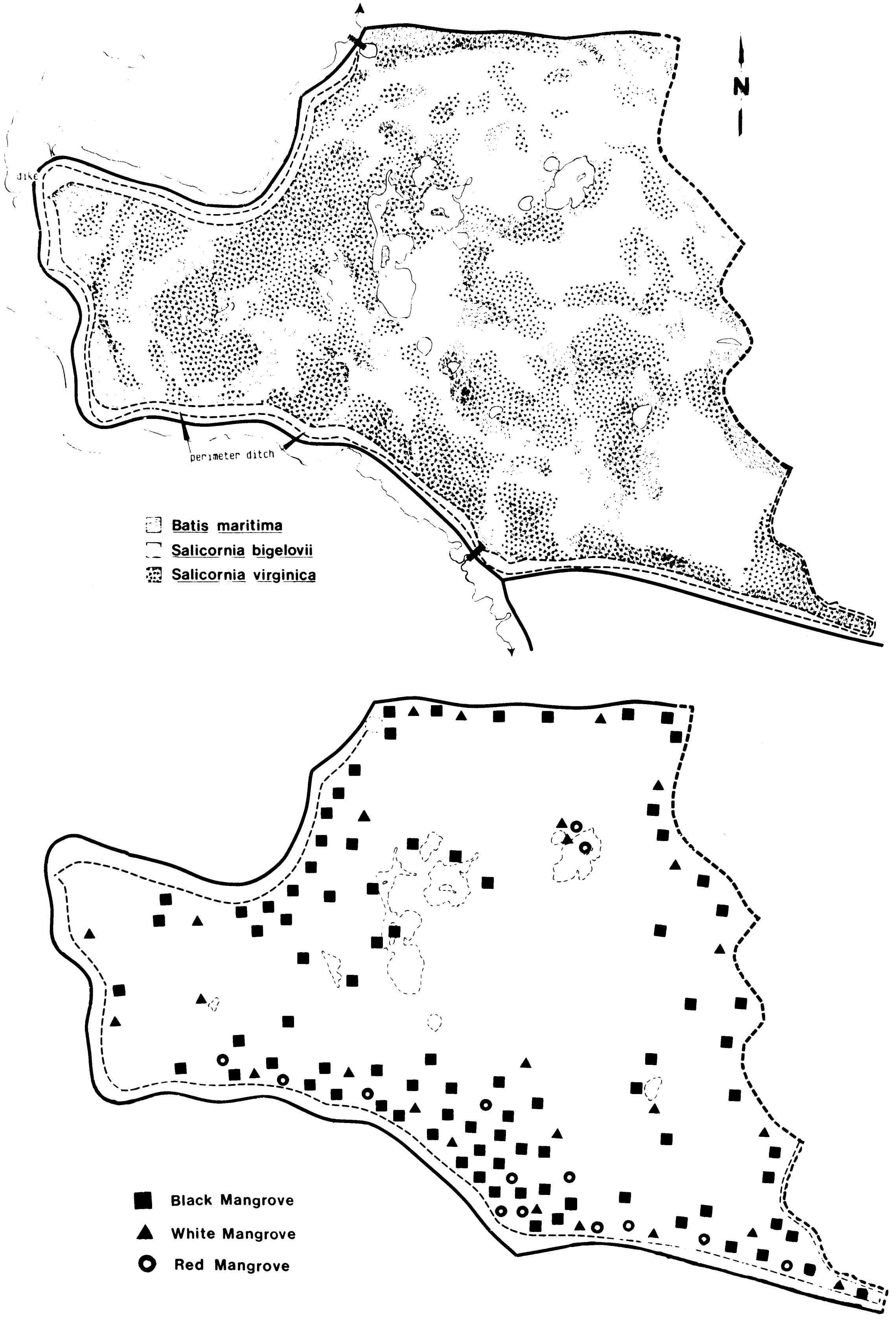


Figure 3. Approximate occurrence and location of major marsh vegetation (January 1984).

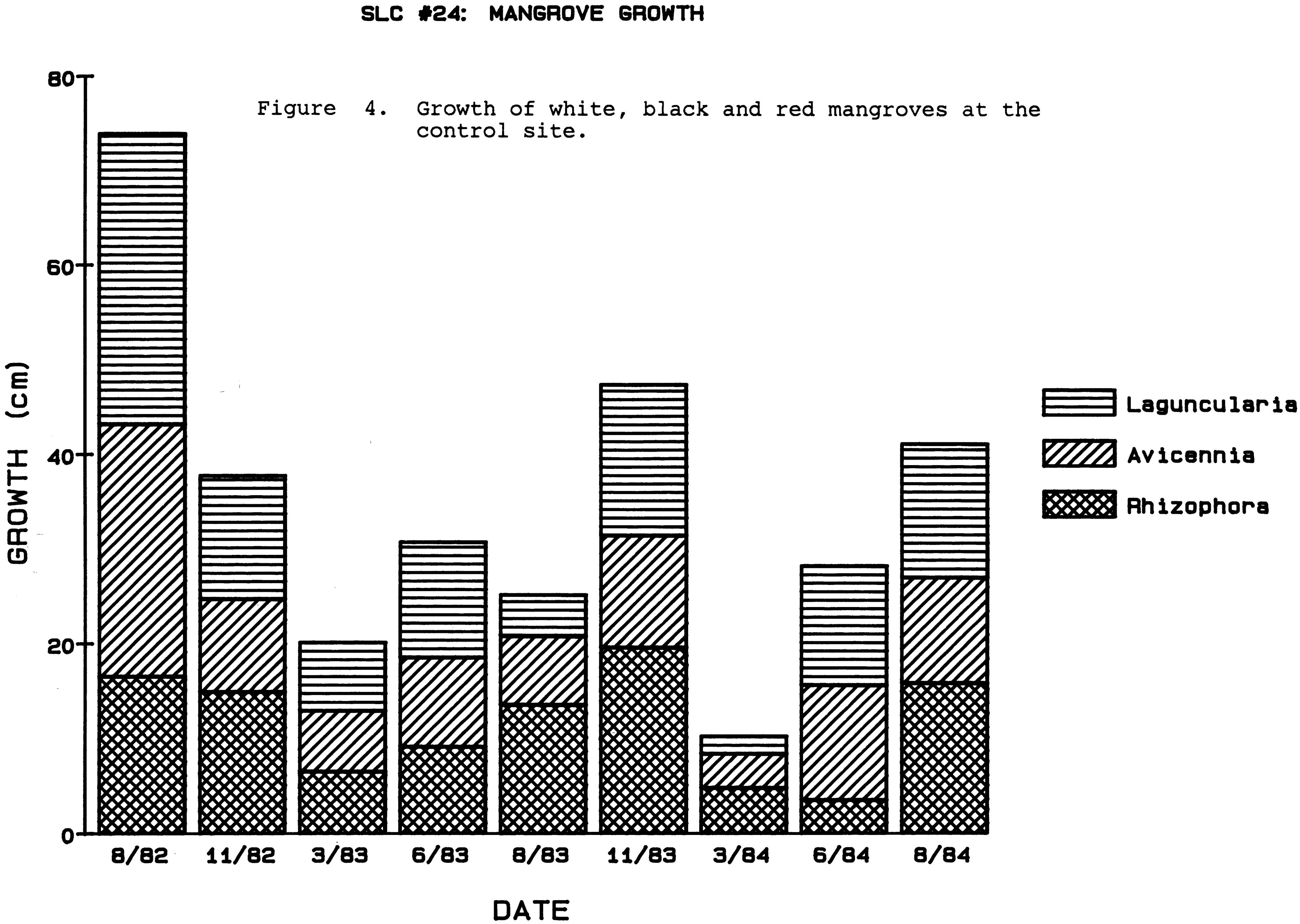


Figure 4. Growth of white, black and red mangroves at the control site.

Historical Effects of Impounding Salt Marshes

In central and south Florida, unimpounded marshes are generally vegetated with *Batis*, *Salicornia*, and black mangroves. In the late 1950's, Harrington and Harrington (1961) reported that 16 species of fish regularly use these high marshes. Unimpounded marshes also have been reported to produce large, synchronous mosquito broods which are hatched from rainfall and/or high tides (Harrington & Harrington, 1961). Impounding has been shown to cause significant mortality of marsh vegetation, and often results in the replacement of the typical *Batis*-*Salicornia* marsh with almost monospecific stands of red mangroves. A significant reduction of fish species diversity after impounding (16 to 5) has been demonstrated (Harrington & Harrington, 1982), but Gilmore et al. (1982) have shown that as many as 12 species of fish can regularly be found in isolated, unpumped impoundments regardless of the harsh environmental conditions of high salinities and low dissolved oxygen levels often occurring there.

Year-round or seasonal flooding can adequately control salt-marsh mosquitoes (Clements & Rogers, 1964). Also, if flooding is not excessive, vegetation can persist while controlling mosquitoes (Provost, 1974). Impounding also has been reported to change marshes barren of waterfowl to habitats very attractive for ducks and large wading birds (Provost, 1959).

Management Implications of Past and Current Research

The present study, although still in progress, has already produced important information that will be valuable when devising management strategies for salt marsh impoundments. This and other studies are showing that it may be possible to implement management methods which provide mosquito control benefits while minimizing many of the adverse effects usually associated with salt marsh impoundments and allowing the marshes to provide many of the benefits that they naturally produce.

Data from this study demonstrate that the normal mosquito control schedule of closing the impoundment in the early spring, retaining pumped estuarine water, rainfall and tidal waters within the impoundment, then reopening the marsh on the fall high tides with culverts placed through the dike is basically compatible with the major periods of fish ingress and egress. Both resident and transient fish species can use the culvert to pass between the marsh and the lagoon.

The vegetation data is encouraging. Although in 1979 the experimental impoundment was devoid of vegetation from past excessive flooding (Gilmore et al., 1982), it has experienced significant regrowth of the typical high marsh vegetation. The greater growth of all species of mangroves in the control cell (Fig. 4) is at first puzzling. Given the fact that the vegetation in most isolated impoundments consists of almost monospecific stands of red mangroves, one would expect that only this species would show greater growth in the control cell. It appears, however, that infrequent events, such as high rainfall, storms, and very high tides, may play a significant role in this phenomenon (Rey, in prep.).

With the present water-control-structure configuration (two 45.7 cm diameter culverts for a 20.2 ha impoundment), retaining water with flap-gate risers to the minimum elevation necessary for adequate mosquito control has, so far, not been permanently detrimental to succulent halophytic vegetation development. However, without the capabilities of augmenting the water levels established by rain waters and tidal intrusions by pumping lagoon water during the spring and summer months, unacceptably high numbers of mosquitoes were periodically produced in the experimental marsh. In another Indian River impoundment, Clements and Rogers (1964) showed that such supplemental pumping can provide adequate mosquito control benefits. This study has demonstrated that larvivorous fish are not able to control these synchronous mosquito broods even when they had access to the mosquito larvae. Fish gut analysis demonstrated that mosquito larvae were an insignificant part of the diet of marsh fish.

Research and experience are showing that many of the original functions of high marshes can be preserved while still maintaining multipurpose management, if the proper techniques are utilized. However, much work remains to be done to identify and "fine-tune" management strategies, and to determine appropriate variations when attempting to cope with different situations and/or management goals. Some goals may be attainable through generalized schemes that are compatible with a variety of objectives, while others may require very specific and idiosyncratic approaches. At the moment, the limitations to reintegrating these isolated high marshes are proving to be more political than scientific.

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EFFECTS OF *IN SITU* SHADING ON *THALASSIA TESTUDINUM*: PRELIMINARY EXPERIMENTS

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ABSTRACT

Three 0.92 m^2 plots of turtle grass, *Thalassia testudinum* Banks *ex* König were shaded for 68 days during the summer of 1984 to determine the effects of light stress on turle grass growth and sediment chemistry. The shaded plots, receiving only 25 percent of incident sunlight, were randomly interspersed with three control plots in a homogeneous turtle grass bed near the mouth of Tampa Bay, Florida. No significant treatment effects were observed in pore water sulfide concentrations, pH, or redox potentials. The total leaf area per short-shoot, the density of short-shoots, leaf blade length, leaf blade width, and turnover times for short shoot biomass all declined dramatically in the shaded *Thalassia* plants. We suggest that these parameters may be used in appropriate circumstances as indicators of light stress in *Thalassia*.

INTRODUCTION

Seagrasses are distributed worldwide in shallow subtidal environments along coasts and within estuaries. In virtually every place where they occur, seagrass communities support diverse food webs which ultimately support sport and commercial fish species (McRoy & Helfferich, 1979).

The mesohaline and euhaline portions of most Florida estuaries are characterized by vertical zonation of three seagrass species. *Halodul wrightii* Aschers., Cuban shoal grass, typically occurs in the low intertidal and shallow subtidal zone. *Thalassia testudinum* banks *ex* König, turtle grass, occurs from mean low water (MLW) to depths of 1 m below mean low water (MLW). *Syringodium filiforme* Kutz, manatee grass, occurs only in subtidal regions to 2 m below MLW (Phillips, 1960). Lower depth limits are extended in clear water, and light penetration is frequently cited as the single most important factor controlling vertical distribution of seagrasses (Bulthuis, 1983). Seagrasses in several estuaries are in peril as a growing body of evidence now links seagrass die-back to anthropogenic light stress.

Chesapeake Bay and Tampa Bay stand as landmarks where the correlative link between anthropogenic turbidity (due to channel dredging and nutrient-stimulated phytoplankton blooms) and declines in seagrass area is strongest. Burt (1955) documented the trend of increasing turbidity in the Chesapeake Bay from 1939 to 1954, the same time period in which a dramatic loss in submerged macrophyte cover occurred in the upper two-thirds of the Chesapeake Bay (Van Tine and Wetzel unpubl). Using hydrographic surveys, Lewis et al. (1985) determined that as much as 80 percent of the original seagrass present in the Tampa Bay system has been lost in the past 100 years while light transparency of the water column has consistently declined. If light stress (defined here as light attenuation) is responsible for seagrass die-back, investigation into the actual mechnisms and acommpanying funcitonal and structural symptoms of light stress may be very important for the diagnosis and prevention of seagrass die-back.

Our research was carried out with two objectives. The first was to identify early warning symptoms indicative of light stress in Thalassia testudinum. The second objective was to determine the potentially synergistic role of sulfide toxicity in light-stress-induced mortality. Sulfide toxicity is a recognized phenomenon in marsh plants and rice (Joshi et al, 1975; Vamos & Koves, 1972; Carlson & Forrest, 1982), but only one reference which we could find considered sulfide as a physiological stress in seagrasses (Penhale & Wetzel, 1983). If sulfide contributes to seagrass mortality, a whole new suite of parameters must be considered to anticipate the survival of an existing seagrass bed or the success of a transplant effort.

STUDY AREA

The study area was located on the north side of Mullet Key (lat. 27°38'N long. 82°42'W) at the mouth of Tampa Bay, Florida. Mullet Key is contained within Fort DeSoto Park operated by Pinellas County. The site was selected for protection from vandalism and generally good water quality. The Thalassia bed selected for study was located at an elevation of approximately 0.0 m relative to the 1929 mean low water datum of the National Geodetic Survey. At this elevation, water depths over the Thalassia ranged from approximately one meter at high tide to less than 20 cm on low tides for most of the experiment. Tides exposed the area several days near the end of the experiment, and the bed is frequently exposed during the fall and winter.

METHODS

This experiment was a pilot project, anticipating a larger scale effort in the summer of 1986. The study began August 3, 1984, and ended October 10, 1984. Six 0.96 m^2 plots, each measuring 71 cm X 122 cm were laid out in a 3 X 2 rectangular array. The long axis of the array was oriented north-south, and treatments were randomized to provide three shade plots and three control plots. The control plots were marked by a PVC frame anchored to the sediment surface. Experimental plots were shaded by three thicknesses of gray fiberglass window screen bonded to a PVC frame. The PVC frame was supported 38 cm above the sediment surface to prevent injury to the Thalassia leaf blades. Each thickness of window screen transmits approximately 62 percent of incident radiation, and light attenuation is approximately equal for wavelengths of light between 400 and 700 nm. Total light transmission through the shade screens, measured periodically in the field, was 25 percent of incident radiation. No fouling of the screens was encountered.

Pore water chemistry in the study plots was monitored using samplers designed by Montgomery et al. (1979). One sampler was inserted 10 cm into the sediments at the center of each plot. Pore water samples were collected every two weeks and analyzed for pH, redox potential, and sulfide concentrations using the procedures of Carlson et al. (1983). Redox potentials were expressed as raw potentials between a calomel half-cell and the platinum cathode of an Orion model 94 combination redox electrode. Values were not corrected for temperature or pH variations.

At the beginning of the experiment, a long knife was used to sever rhizome connections around the perimeter of all the plots to prevent translocation of materials from shoots outside the plot to shoots inside each plot. In each plot, ten short-shoots (sensu Tomlinson & Vargo, 1966) were labelled by plastic tags looped around the shoot base at the sediment surface. At seven to fourteen day intervals, the length of each leaf blade in the numbered short-shoots was measured to the nearest 0.1 cm by SCUBA divers. The oldest blade in each short-shoot was labelled blade A, and successively younger blades were labelled alphabetically.

The variable TOTLTH is the sum of the lengths of all the leaf blades present in a given short shoot. Leaf area index (LAI) is a "one-sided" measure calculated as the product of TOTLTH, mean blade width, and shoot density per square meter of bottom area. As shoot density was not measured at the beginning of the experiment, initial LAI values were calculated using the final shoot densities of control plots. The parameter called growth increment is the amount of TOTLTH in each short shoot replaced by daily growth and can be represented by the following expression:

$$\text{Growth increment} = \frac{\text{TOTLTH}_1 - \text{TOTLTH}_0}{(\text{TOTLTH}_1) \times (t_1 - t_0)}$$

Turnover time for the entire shoot is the reciprocal of growth increment. Statistical analyses (ANOVA, t-tests, and Duncan's multiple range tests) were carried out using SAS (SAS Institute, 1982).

RESULTS

Physical and Chemical Parameters

Water termperature dropped through the experiment from 30.8° C on August 3, 1984, to 24.9° C on October 10, 1984. The most rapid decline occurred between September 16 and September 22. Comparison of pore water pH, redox potentials and sulfide concentrations (Table 1) showed little difference between shade and control plots. pH showed an initial dip and then rose at the end of the experiment in both treatments. The range of pH values encountered in both treatments was 6.99 to 7.39. Redox potentials in both treatments showed a monotonic rise from near -390 mv at the beginning of the experiment to -180 mv at the end of the experiment. Pore water sulfide concentrations varied without pattern between 311 um and 599 um. There were no significant differences in sulfide concentrations

Table 1. Pore water chemistry of shade and control plots. All data presented as mean (standard deviation). N=3 for all parameters.

	Sulfide (um)	pH	Redox Potential (mv)
Control			
Initial	376 (96)	7.39 (0.01)	-389 (7)
Final	458 (285)	7.35 (0.15)	-177 (18.5)
Shade			
Initial	358 (124)	7.37 (0.04)	-385 (17.5)
Final	511 ---	7.39 (0.10)	-188 (13.3)

Table 2. Changes in structural and growth indices of *Thalassia testudinum* in response to shading. All data are presented as mean (one standard error). N for most parameters is based on 30 shoots. See methods for definitions.

Treatment/Time	TOTLTH (cm)	Blade Width (mm)	Max Blade Length (cm)	Blades per Shoot	Shoot density	Leaf Area Index	Growth Increment
Control							
Initial	94.7 (5.4)	7.10 (0.18)	38.6 (1.3)	3.00 (0.10)	--	2.95 (0.36)	1.21
Final	71.5 (3.7)	7.53 (0.12)	33.4 (1.2)	3.78 (0.52)	440 (12.8)	3.13 (0.18)	0.85
Shade							
Initial	78.1 (5.1)	7.65 (0.08)	37.6 (0.9)	2.60 (0.13)	--	2.61 (0.09)	1.04
Final	37.5 (3.4)	7.27 (0.15)	26.1 (0.6)	1.75 (0.11)	380 (19.2)	1.53 (0.24)	0.44

between treatments (a = 0.05), but sulfide concentrations may have covaried with pH values.

Growth Responses

The parameter TOTLTH integrated changes in both blade length and density. TOTLTH showed a significant decline (a = 0.01) in both control and shade plots throughout the experiment (Fig. 1). For the first 14 days of the experiment, TOTLTH's in control and shade plots (presented here as the mean ± 2 standard error) was not significantly different. From day 19 of the experiment to the end, shade plot TOTLTH remained significantly lower (a = 0.01) than TOTLTH in control plots. Between day 49 (September 22) and day 68 (October 10, 1984), both control and shade plot values dropped rapidly as temperature decreased. The treatment effect was preserved throughout the course of the experiment, however, and final values for control plots (71.5 cm per shoot) were nearly double values in shade plots (37.5 cm per shoot).

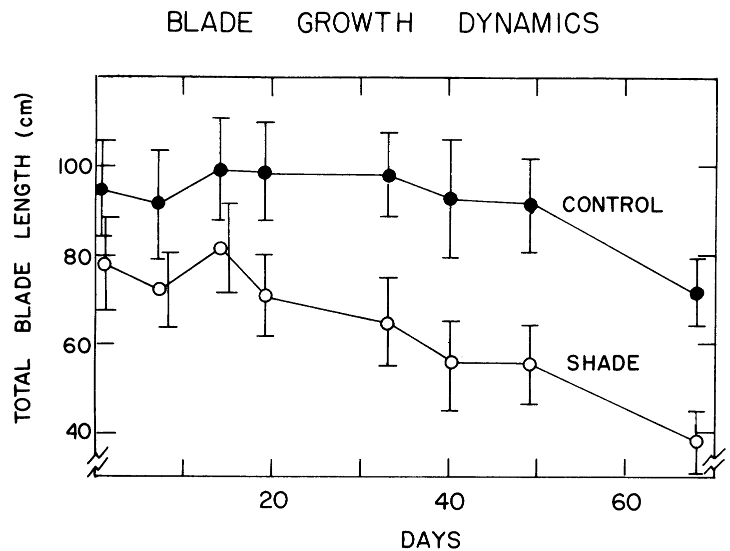
Blade width also showed a treatment effect over the course of the experiment (Table 2). The initial leaf width in shade plots was significantly greater than that of control plots (a = 0.01). By the end of the experiment, however, the pattern had reversed, and control plot blades were significantly wider (a = 0.01) than shaded blades.

Changes in the maximum length of leaf blades in each short-shoot (Table 2, Fig. 2) showed both treatment and time effects. Maximum blade length was shorter at the end of the experiment in both shade and control plots. Control plot values declined from 38.6 cm to 33.4 cm, while shade plot values dropped from 37.6 cm to 26.1 cm. While initial values in shade and control plots were not significantly different, the maximum blade length of shade plot short-shoots was significantly lower (a = 0.01) than control plots at the end of the experiment.

The observed decline in the parameter TOTLTH in the shade plots reflected, in part, the decline in maximum blade lengths in the shade plots. A major fraction of the TOTLTH decline was also due to a decline in the number of blades per shoot (Table 2). While the decline in the number of blades per shoot in control plots was not significant (a = 0.05), the decline in blades per shoot in shade plots was significant and dramatics. Not only did the shade plots shoots have fewer leaf blades, but the blades were sloughed at a lower rate (Fig. 2).

Shoot densities measured at the end of the experiment in shade plots were also significantly lower (a = 0.01) than in control plots. The combined effects of reduced leaf area per short-shoot and reduced shoot density in shade plots caused a marked reduction in leaf area index in shade plots from 2.61 m^2 leaf area per m^2 bottom area to 1.53 at the end of the experiment.

Growth increment (the fraction of total shoot leaf length replaced by daily growth) also showed a treatment effect. Not only was the standing crop in shade plots reduced, but the rate of biomass replacement was further slowed. While leaf blade elongation rates in shade plots



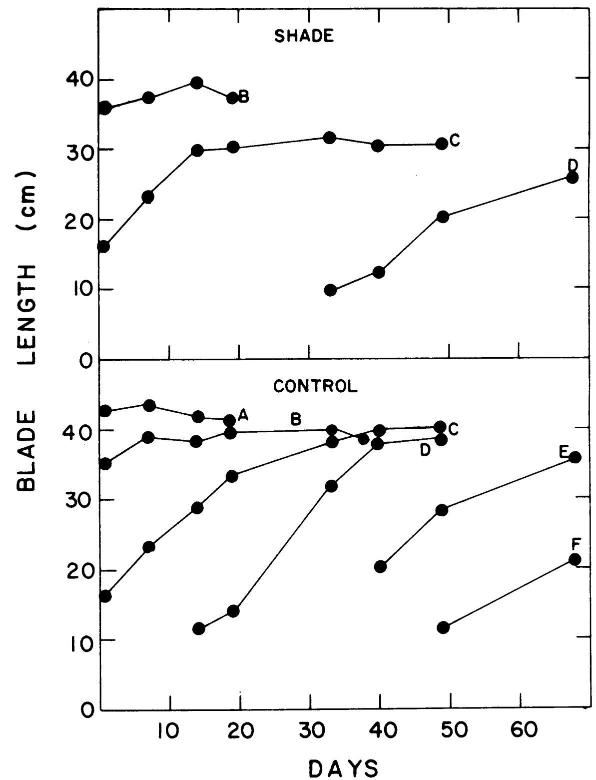


Figure 1 (above): Variations in total blade length per short shoot with time and treatment.

Figure 2 (right): Elongation and retention of leaf blades in shade and control plots.

were reduced (data not shown), individual blades in short-shoots within shade plots were retained longer than in control plots (Fig. 2).

For the most part, our data agree with other shading experiments. The decrease in leaf width that we observed in shaded plots agrees well with Burkholder and Doheny's (1968) observations on Zostera marina. Phillips and Lewis (1983) have also reported that leaf width in Thalassia varies with salinity and geographical distribution and may be a general stress indicator for Thalassia.

TOTLTH showed a marked treatment effect, but the covariance of TOTLTH in both treatments suggested that temperature exerts a strong influence on Thalassia blade length. This penomenon has been described in Tampa Bay by Taylor et al. (1973). They found that the mean blade length of Thalassia varied directly with water temperature from a low of 15 cm at 15° C in February to a maximum of 30 cm at 30° C in July and August. The covariance of control and shade plot values points up the need for carrying out shading experiments in a well-defined temperature regime.

Leaf area index and shoot density values were lower than those reported by other researchers. Reductions in short-shoot or "turion" densities also have been reported in other shading experiments (Backman & Barilotti, 1976, Barko & Smart, 1981, and Bulthuis, 1983, 1984). These studies suggest that short-shoot density would have continued to decline if the experiment had lasted longer.

The lack of a demonstrable rhizosphere effect on sediment sulfide concentrations or redox potentials was puzzling. Several authors have reported rhizospher effects on the sediment redox environment for seagrasses and aquatic macrophytes (Carpenter et al., 1983, Iizumi et al, 1980, Smith et al, 1984, Thursby, 1984, and Wetzel and Penhale, 1979). Anatomical studies of Thalassia roots and leaves (Tomlinson, 1969 and 1972) show well-developed aerenchyma which could easily transport large quantities of photosynthetic oxygen to the roots and sediments. Oremland and Taylor (1977) reported diurnal variations in the oxygen content of bubbles in the sediments of a Thalassia bed in the Florida Keys. The lack of a marked treatment effect on sediment chemistry at our study site may be due to the highly reduced nature of the sediments. In these sediments, the concentrations of sulfide (0.3-0.6 mM) may overwhelm any oxygen leakage from roots. Oremland and Taylor did not describe the redox status of the sediments which they studied, but there is considerable variation in the substrate texture of Thalassia beds in Tampa Bay (M. Durako & M. Moffler, personal communication). Demonstration of the role of sulfide as a synergistic stressor must await stable sulfur isotope ratio analyses of plant tissue from the experiment.

CONCLUSIONS

We have demonstrated that several structural parameters and growth indices of Thalassia testudinum respond to light stress. TOTLTH in particular responded to light stress within three weeks. We suggest that these parameters may be used to determine the extent of anthropogenic stress in Thalassia beds by comparing these structural indices between stressed beds and adjacent healthy *Thalassia* beds. While anthropogenic light stress may be less intense than the 75 percent reduction in incident light which we used, low intensity light stress may be offset in "natural" stress conditions by longer duration. Future studies must address the effects of longer and less intense shading on *Thalassia* and the relationship of sediment chemistry to seagrass die-back.

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WETLAND REGULATION: AN EFFECTIVE APPROACH

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ABSTRACT

Incomplete and, at times, inaccurate definition of wetland types has led to ineffective and inconsistent enforcement and administration of state wetland regulations. Four basic shortcomings of state wetland policies and some of the consequences of these shortcomings are discussed. The Rhode Island Fresh Water Wetlands Act, along with the acts of several other northeastern states, is critically reviewed. Special emphasis is played on the need to fine-tune the legislative definition of wetland types and to improve state mitigation policy. Examples from the wetlands acts of other states, particularly Florida's Henderson Wetland Protection Act of 1984, are used to show the biological and geological detail necessary for successful and consistent implementation of the Act. Necessary components of a successful mitigation policy are discussed.

INTRODUCTION

The work of Wetland Management Specialists, Inc., with developers, conservation groups, regulatory agencies, and towns has given the firm a unique vantage point from which to view the inadequacies of current wetland legislation. Wetland regulations enacted during the early 70's lack sufficient biological and geological detail upon which to base today's regulatory decisions. Incomplete and, at times, inaccurate legislative definitions of wetland types has led to ineffective and inconsistent enforcement and administration of state wetland acts. Most of the early wetland regulations were enacted in response to overwhelming public concern. Although well-intentioned, they were crisis oriented and based upon little biological detail. Initially, the legislation served the purpose of regulating and slowing the destruction of valuable natural resources. Today, however, wetland legislation enacted for the purpose of enhancing and protecting wetland values is being used to unnecessarily limit development of "marginal" wetland and mesic areas. This lack of consistency has been harmful to developers and environmentalists alike. Most of the existing state wetland acts make no attempt to incorporate emerging concepts such as mitigation in the form of wetland creation and habitat improvement. When properly implemented, such measures can allow economic development and habitat preservation to exists side by side.

In the past decade, the complexity of wetland regulation as well as our knowledge of wetland ecosystem functioning has grown exponentially. Knowledge gained from over a decade of experience must now be incorporated into the specifics of wetland regulation.

This paper discusses four basic shortcomings of state wetland regulation, and some of the consequences of these shortcomings.

METHODS

The wetland legislation and accompanying rules and regulations of several states, particularly Rhode Island, were critically reviewed. Special emphasis was placed on assessing the scientific detail of the legislative definitions of wetland types and on evaluating wetland mitigation policy. The legislative definitions of descriptive terms as they relate to wetland delineations were also analyzed. There is a need for phrases such as "significant portion," "duration of growing season," "at or near the surface," etc., to be precisely and reasonably defined.

RESULTS AND DISCUSSION

In analyzing state wetland legislation, the following four basic shortcomings were identified:

1. Incomplete and, at times, inaccurate definition of wetland types and boundaries.

2. No distinction between regulating wetlands from a hazard versus a value point of view.

3. Insufficient administrative and procedural requirements governing such items as exempt activities, criteria for judging significant impact and designation of unique or prime wetlands.

4. Little attempt to incorporate emerging concepts such as mitigation in the form of wetland creation and habitat improvement into state wetland policy.

The lack of detail in legislation is understandable. Most legislators lack the technical expertise to adequately address the issues. Additionally, enacting legislation involves compromise which often necessitates vague legislative detail. Through the rule-making procedure, however, it is the regulating agency's responsibility to fill in this missing detail. Failure to do so results in excessive agency discretion, applied on a case-by-case basis. This, in turn, leads to inefficient and inconsistent enforcement of state wetland legislation. Sanctions are selectively applied, and wetland boundary determinations are subjective to, and to a large degree dependent upon, which state biologist happens to visit the site. Findings of significant impact lack consistency from case to case, and permit processing deadlines are not adhered to. This places an undue burden upon developer and conservationist alike. Delays become the status quo and outcomes are rarely predictable. It challenged in Superior Court, states with a track record like that of Rhode Island stand in danger of having the entire wetlands act overturned on the basis of constitutionality.

The following definition of bog taken from the Rhode Island Fresh

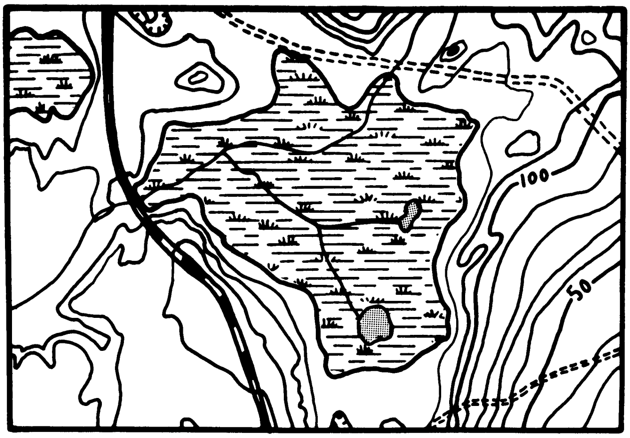
Water Wetlands Act typifies what is meant by lack of biological detail (State of Rhode Island 1971). "A place where standing or slowly running water shall be near or at the surface during the growing season and/or where the vegetational community shall have over 50% of the ground or water surface covered with sphagnum, and/or where the vegetational community shall be made up of one or more of, but not limited to, nor necessarily including all of the following . . ." The Act then lists some typical bog dwelling species, as well as some species common to both bogs and shrub swamps. No mention is made in either the Act or the accompanying Rules and Regulations regarding such items as characteristic bog microtopography, site geology, type of substrate, pH, and depth of peat. "Near or at the surface" as defined in the Act means within 36 inches of the ground's surface. As the definition now reads, it is possible to call a minerotropic shrub swamp with a 50 percent sphagnum cover, mound and pool topography, and a shrub community of blueberry (Vaccinium atrococcum) and red maple saplings (Acer rubrum) a bog. With no minimum size requirements on bogs and a three-acre minimum requirement on shrub swamps, such a tactic could legally be used to prohibit development in undersized shrub swamps.

If it is the state's intention to protect swamps under three acres, then the minimum size requirement for swamp should be reduced. Stretching the legal definition of bog so as to include swamps distorts biological reality and sets a dangerous legislative precedent.

Study sites mapped in Figure 1 are used to illustrate the ambiguity inherent in current legislation. Diagram 2 of the figure depicts a streamside shrub swamp, which the Rhode Island Department of Environmental Management (DEM) has erroneously from a biological point of view, but correctly from a legal point of view, classified as a bog. While there are some acid-tolerant species present such as leather-leaf (Chamaedaphne calyculata) and sphagnum moss (Sphagnum sp.), no other bog characteristics are evident. There is no floating mat, there is a muck rather than a peat substrate, and the geographical location is streamside rather than the usual restricted drainage.

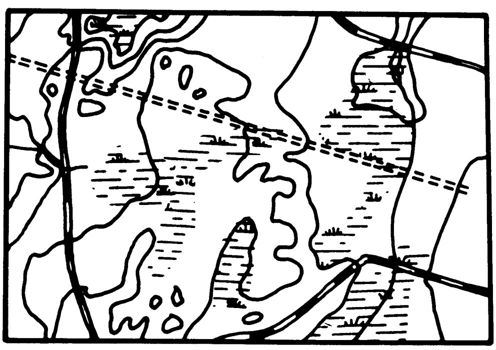
Figure 1 also illustrates the geographical location of a true biological bog. Diamond Bog, as this area is known, is characterized by a floating sphagnum mat rather than mound and pool topography. Peat depths generally exceed 15 feet and drainage is extremely restricted. Several characteristic bog species such as pitcher plant (Sarracenia purpurea), sundew (Drosera rotundifolia), bog spruce (Picea mariana), leather-leaf, and cranberry inhabit the area.

Similar problems exist with the term "swamp," as defined in the Rhode Island Wetlands Act. As written, this definition encompasses both swamps and marshes, and both wetland and mesic areas. Although the species listed are primarily swamp species, the last line of the definition reads "or species indicative of marsh." Furthermore, no distinction is made between wetland and transitional areas, or between obligate and facultative species. Under the Rhode Island Act, a red maple forest with an understory of sweet pepperbush (Clethra alnifolia) (both facultative species), a ground cover of false Canada mayflower (Maianthemem canadense)



True Biological Bog

Legal R.I. Bog



Mesic Forest (legal R.I. wetland) Contiguous With True Biological Bog

Figure 1: Geographical location of selected "legal" vs. biological wetlands in R.I.

(a mesic indicator), and a water table three feet below the surface could be called a swamp. This is ironic in that on-site sewage disposal systems in Rhode Island may be placed in areas (legal wetlands) where the seasonally high water table is only two feet below the surface.

Diagram 3 of Figure 1 depicts contiguous areas of mesic forest and wooded swamp. Swamp areas at this site are characterized by mound and pool topography, a ground cover of sphagnum and skunk cabbage (Symplocarpus foetidus), a shrub layer of sweet pepperbush, and an overstory of swamp white oak (Quercus bicolor) and red maple. This habitat, including transitional areas, roughly coincides with the swamp symbols present on the figure. Adjacent areas without swamp symbols meet the legal definition of swamp but are best characterized as mesic forest. They are vegetated with red maple, sweet pepperbush, and false Canada mayflower. The water table is approximately two feet below the surface.

"Marsh" under the Rhode Island Act and its accompanying Rules and Regulations is also broadly defined. The definition encompasses wet meadow, shallow marsh, deep marsh, and floating-leaved marsh.

Many of the freshwater wetland acts enacted during the 70's were modeled after the Rhode Island Act. As such, these acts are flawed by a similar lack of biological and geographical detail. Water regimes, geographical location, indicator species, and substrates are all poorly detailed. In contrast, Florida's Henderson Wetland Protection Act of 1984 and its accompanying Rules and Regulations provide substantial biological criteria upon which to base wetland determinations (State of Florida 1984). Some of the more important aspects of this detail are outlined below:

1. The Act contains extensive species lists subdivided into wetland indicators (submerged species), transitional indicators, and species not to be used as indicators. In some instances the species lists are specific to particular geographical areas. This represents a completely different approach from the "all inclusive" lists that appear in most state wetland legislation.

2. The Act relies on both soils and vegetation in determining wetland edge.

3. Wetland definitions such as swale and landward extent of surface waters are specific and detailed.

4. The law outlines very specific criteria for the determination of wetland edge. It instructs how to locate the edge of wet (landward extent of surface water) by site inspection or photointerpretation, and provides formulas for determining relative density and basal area. The Act and its accompanying regulations explain how to locate the landward extent of surface waters where visual inspection and photointerpretation are inadequate. It also describes the methodology to be applied in determining dominance and wetland edge if more than one strata is used.

5. The Act distinguishes between transitional and submerged (wetland) areas. Plant species are designated as transitional or submerged. Where

transitional species are dominant, they must accupy at least 80 percent of the stratum before the area may be classified as wetland. Projects in transitional areas requiring less than 25 cubic yards of fill are exempt from permitting.

6. This biological detail is futher supplemented with such administrative measures as the establishment of the Vegetative Index Review Committee, the creation of a Wetlands Indicator Index, and a Wetland Monitoring Program.

The Florida Act is new. As its implementation is tested, problems are likely to surface. The Act does, however, represent a significant advance over most of the wetland legislation enacted in the 70's. Similar detail incorporated into the wetland acts of other states would help to relieve ambiguity and hence expedite the permit process.

The second shortcoming evident in the wetland legislation surveyed is that no distinction is made between regulating wetlands from a value versus a hazards point of view. Wetland legislation originally enacted for the purpose of enhancing and protecting wetland values is being used to unnecessarily limit development of "marginal" wetland and mesic areas. According to Dr. Frank Golet, wetland specialist at the University of Rhode Island and a co-author of the National Wetlands Inventory, the values that we traditionally ascribe to wetlands (flood control, pollution abatement, wetland wildlife habitat, and certain forms of recreation and harvest) are associated primarily with areas where surface flooding occurs (Golet, 1984). Such wetlands are associated with very poorly drained, rather than poorly drained soils. Poorly drained sites, on the other hand, possess few wetland values but pose constraints for building development, the installation of septic systems, agriculture and the like. These are problems which can usually be overcome with proper site design and engineering. Much of the confusion regarding the definition and limits of wetlands stems from the inability to recognize and legislatively account for these two different viewpoints. Policy should be revised to account for the differences between wetland and wet land (McCaffrey, 1985). Dr. Golet's current research focuses on defining the biological and geological criteria necessary for determining wetland boundaries in"marginal" areas.

The third shortcoming common in state wetland legislation is insufficient administrative and procedural requirements for (1) evaluating wetlands, (2) designating unique or prime wetlands, (3) determining exempt activities, and (4) determining significant impact. By way of example, in R.I. Alterations in Areas subject to Flooding (ASF's) and Areas Subject to Storm Flowage (ASSF's) were originally intended, given adequate engineering, to be approved via a preliminary determination and exempt from the formal permitting process. They were, in effect, to be regulated from a development hazards point of view and not from the perspective of wetland values. Due to a lack of specific criteria, an inordinate amount of agency discretion and confusion surrounding the legislative definition of ASF's and ASSF's, these areas have been subjected to the formal approval process. Florida and New York, on the other hand, provide detailed categories of exempt activities (State of New York, 1981). Similarly, the New Hampshire Act defines major and minor projects on the basis of such tangible items as square feet of fill and size of subdivision, and on less tangible issues such as cumulative impact, by majority vote (State of New Hampshire, 1982).

In many of the state wetland acts, there also exists substantial administrative discretion in designating unique wetlands. In Rhode Island they are determined on a case-by-case basis by the biologist in charge of that particular application. There is no review board, so once a wetland is designated as valuable, it can be contested only through a public hearing or in Superior Court. This is costly both for the state and for the developer.

New Hampshire has taken a unique approach to this problem. The wetlands act details criteria that municipalities are to use in designating prime wetlands. Wetlands so designated are submitted to a review committee to be judged for compliance with the required form and content. This eliminates biologist bias and haphazard case-by-case approach. A developer knows beforehand that the wetland is designated as a prime wetland and that his options regarding the development of the property are limited.

The fourth and final legislative shortcoming I would like to address is the reluctance of states to incorporate mitigation in the form of wetland creation and habitat enhancement into their wetland regulations. Fifteen years ago when legislation was first enacted, there was insufficient information upon which to base a scientifically sound mitigation policy. Fifteen years of reasearch, however, has produced viable wetland creation and enhancement techniques. When properly implemented, such measures can allow economic development and habitat preservation to exist side by side.

As with the biological definitions of wetland types, mitigation policy, if present at all in state wetland legislation, lacks sufficient detail to be implemented effectively. In Rhode Island mitigation policy appears neither in the Fresh Water Wetland Act nor its Rules and Regulations, but only as a vague statement in an interagency memo. A conscientious applicant having reviewed the Act and its Rules and Regulations would have no idea that the State even had a mitigation policy. The interagency memo calls for a 1:1 value replacement in high and outstanding value wetlands as evaluated by the modified Golet Wetland Evaluation System (Golet, 1976). No additional biological detail or review criteria is provided.

Massachusetts has taken a step in the right direction in legislating a fairly specific mitigation policy (State of Massachusetts, 1983). The policy which is spelled out in the performance standards for vegetated wetlands provides some biological and geographical detail. The goal of the performance standards is to ensure that the replacement area will function in a manner similar to the area that will be lost. The biological and geographical detail includes specifics relating to (1) ground and surface water hydrology, (2) the configuration and location of the replacement wetland with respect to the bank, (3) surface area of the replacement wetland, (4) erosion control methods, and (5) use of indigenous species in vegetating replacement sites.

In New York mitigation can be used to increase the likelihood that a proposed project will meet the applicable standards for permit issuance. Mitigation must occur in immediate vicinity of the proposed project, must provide the same benefits that will be lost through the proposed activity, and must be subject to regulation under the Act. No substantial biological or geographical detail is provided. Florida's Henderson Wetland Protection Act addresses mitigation only as it relates to water quality. Wetland mitigation is addressed in the Rules Governing Mine Reclamation, but only as it relates to mining projects (State of Florida, 1980).

Mitigation policy, if it is to be successful, must be well-defined. I must draw upon the political experience and scientific achievements of the past 15 years of wetland protection efforts. Mitigation should be viewed as a tool that requires developers (1) to modify their proposals so as to cause the least environmental and/or social harm, and (2) (only after the first has been completed) to provide habitat of equal value and/or provide privileges which offset any unaviodable loss.

The following are some guidelines for states to follow when establishing mitigation policy:

1. Biological, geographical, and (where appropriate) geological criteria governing mitigation should be spelled out.

2. Using productivity as a value to be mitigated against is technically complex. What techniques are to be used in assessing productivity? How soon must the productivity of the replacement site equal the productivity of the replacement area? If using productivity in the strict sense, a sewage lagoon or fish hatchery could be substituted for a wetland (LaRoe, 1977). Oregon has addressed this problem by focusing on the potential of biological systems and not productivity. Guidelines have been developed to ensure that with time the area will develop a qualitatively and quantitatively similar flora and fauna (State of Oregon, 1977).

3. The mitigation plan must clearly prioritize certain values and the corresponding geographical locations that exhibit those values. It must be determined which values or set of values are being mitigated for in which geographical locations. Productivity or wildlife value, for instance, may not be considered important in a heavily industrialized area. Along an urban waterfront, perhaps recreation and access are the prime values. To select and mitigate against a single value with no consideration as to the geographical location of the wetland would not result in a functionally efficient mitigation policy.

4. Consideration must be given to the biological and physical interdependency of different habitat types. If this does not happen, there is a tendency to value one ecosystem type over the other, resulting in a net loss of the lesser valued option. Such a decision, if it is to be made, must be based upon adequate scientific, economic, and social information.

5. It should not be assumed that all areas of a particular habitat type have the same inherent value. A monotypic cat-tail marsh, for example, has a relatively low wildlife value but may be important as a water purification system. The state of Rhode Island ignores this fact in that wetland

evaluation scheme and mitigation policy are based solely on wildlife value.

6. Exact amount of mitigation should be determined on a case-by-case basis based upon such things as value of area displaced, size of project, purpose of project (public vs. private benefit, etc.).

7. Established guidelines should be functionally simple and should provide the public with a measure of predictability. This measure of predictability will be based upon the regulatory agency's ability to interpret guidelines consistently from case to case. The geographical specifics of a plan will not have direct applicability in ecologically dissimilar areas. If adhered to, however, general guidelines used in the development of a mitigation plan will have wide utility.

8. The institution of a mitigation bank for financing land acquisition, park development, waterfront beautification, and habitat restoration and management should be considered.

9. Mitigation must be supervised from project beginning to project end by an agency vested with the appropriate legal and political authority.

10. Ultimately, mitigation policy should reinforce the goals and priorities of the Wetlands Act.

CONCLUSION

With over a decade of implementation history and scientific research behind us, it is time that the individual states took a long, hard look at their wetland protection programs in light of the four shortcomings detailed in this paper. To do so would facilitate better management of our natural resources. A failure to do so will exacerbate existing problems and perpetuate public discontent with wetland regulation.

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WATER QUALITY WITHIN A CENTRAL FLORIDA PHOSPHATE SURFACE MINED RECLAIMED WETLAND

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ABSTRACT

The project (Agrico Swamp) site consists of 148 hectares of phosphate mined land of which 61 hectares of wetlands and 87 hectares of contiguous uplands were reclaimed in 1981/82. The wetlands were designed to create freshwater marsh, hardwood swamp, and open water habitats. After two complete growing seasons (two years) the comprehensive monitoring of ground and surface waters within and adjacent to the project, including the receiving waters of Payne Creek, shows that Agrico Swamp is stabilizing. Surface water within the project is comparable to open water areas within the natural wetlands of the region. The vegetation growth should buffer the high alkalinity and moderate other peak values recorded in Agrico Swamp. The most unexpected finding was that total N and total P were relatively low. The low P values may be partially due to the insolubility of cacium fluorapatite in the alkaline water within the project. The perterbation resulting from surface mining is probably responsible for peak values of pH, dissolved solids, specific conductance, alkalinity, and gross alpha in the project. Groundwater quality in Agrico Swamp is good considering the recent surface mining of the area. Payne Creek continues to meet Class III water quality standards with Agrico Swamp having no apparent adverse impacts to date.

INTRODUCTION

Agrico Mining Company proposed to reclaim 148 hectares of phosphate mined land in 1980 to comply with Florida's mine reclamation rules. The company proposed to reclaim 61 hectares of wetland (Agrico Swamp) and 87 hectares contiguous uplands. The reclamation site was originally pine flatwoods and rangeland with some mixed forest before it was mined in 1978 and 1979. This plan was approved in 1980. Construction was completed on all but 8 hectares by December, 1981. The remaining acreage was completed in May, 1982.

The reclamation project was designed and constructed to create open water, fresh water marsh, hardwood swamp, and upland habitats. The project is intensively monitored for evaluating the various tree planting and marsh establishment methods, biological integrity, and the quality and quantity of ground and surface waters within the project. Data from this reclaimed area over the next several years should aid in evaluating the rate and type of habitat development in the reclaimed surface mined lands in Florida.

In order to provide a complete evaluation of the Agrico Swamp reclamation project's changing conditions, a comprehensive water quality monitoring program was designed and implemented in June, 1982. Water quality parameters are monitored quarterly within the Agrico Swamp open water areas, Agrico Swamp groundwater wells, background wells in natural areas in the vicinity of Agrico Swamp, a natural marsh open water near Agrico Swamp and within Payne Creek—both upstream and downstream of the point of discharge of Agrico Swamp.

The water quality monitoring of Agrico Swamp is designed to assess the surface and groundwater quality on-site as well as the receiving waters of Payne Creek. This report covers the period from August, 1983, to July, 1984.

MATERIALS AND METHODS

The project site, referred to as Agrico Swamp, is located adjacent to the flood plain of Payne Creek at Agrico's Fort Green Mine in Sections 27 and 34, Twonship 32S, Range 23E, Polk County, Florida (Fig. 1). The surface mined land was contoured so that all drainage in the project is from west to east. A levee constructed along the eastern boundary of the project impounds drainage from the 148 hectare watershed to form wetlands at the design elevation. Two swale outlets were constructed in the levee to allow overflow discharge of water from Agrico Swamp into the Payne Creek flood plain. The elevation in Agrico Swamp along the base of the levee is 35.97 m MSL. The elevation rises gradually to 36.88 m MSL along the western boundary of the wetland and less gradually on westward across the upland portion of the project to 40.84 m MSL. A water budget for the project was developed to evaluate the disposition of storage, inflow, and outflow of water within the project area during a typical year.

Ponds were constructed within the wetlands with bottom elevations of approximately 32.92 m MSL to maintain open water areas all the year round. Small, shallow depressions were constructed randomly throughout the fluctuating water zone to retain water and harbor fish populations during periods of low water. Two lakes were also constructed in the uplands which overflow via swales eastward into the wetlands.

Water samples were taken from an open water area and four wells within Agrico Swamp and off-site from an open water area, two wells, and three stations along Payne Creek (Fig. 2). Water samples were analyzed for as many as 20 different physical and chemical parameters. These groundwater and surface water test parameters are temperature, fluoride, iron, pH, dissolved oxygen, copper, turbidity, chromium, specific conductance, lead, nitrite nitrogen, alkalinity, zinc, bio oxygen demand, barium, gross alpha, beryllium, total nitrogen, total phosphate, and dissolved solids.

FIGURE 1 - LOCATION MAP

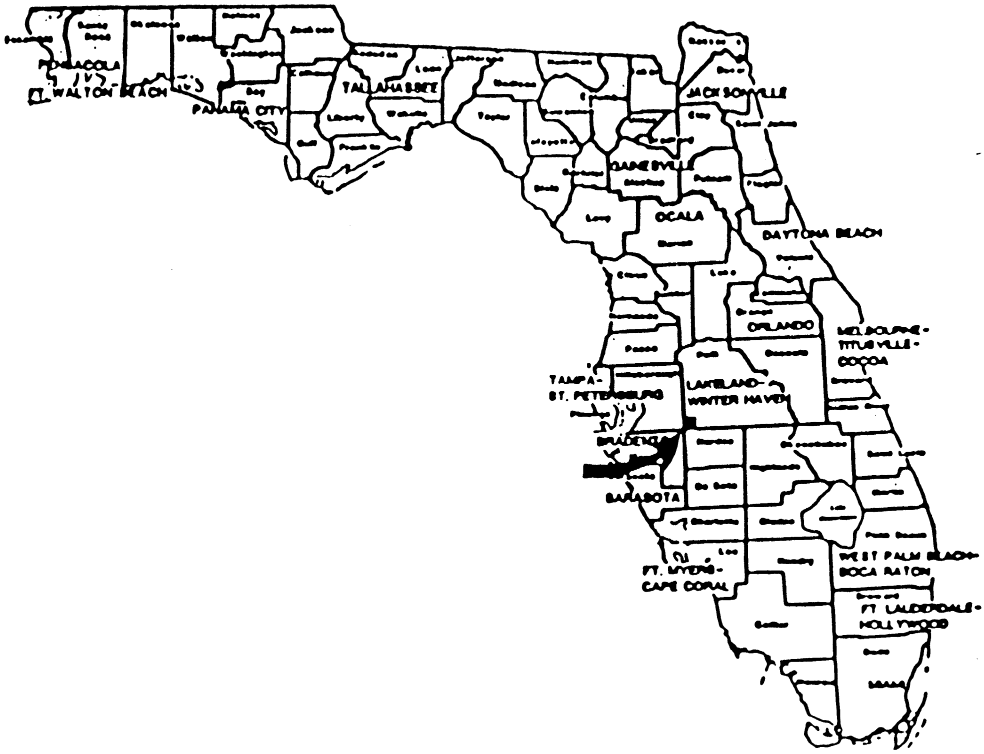
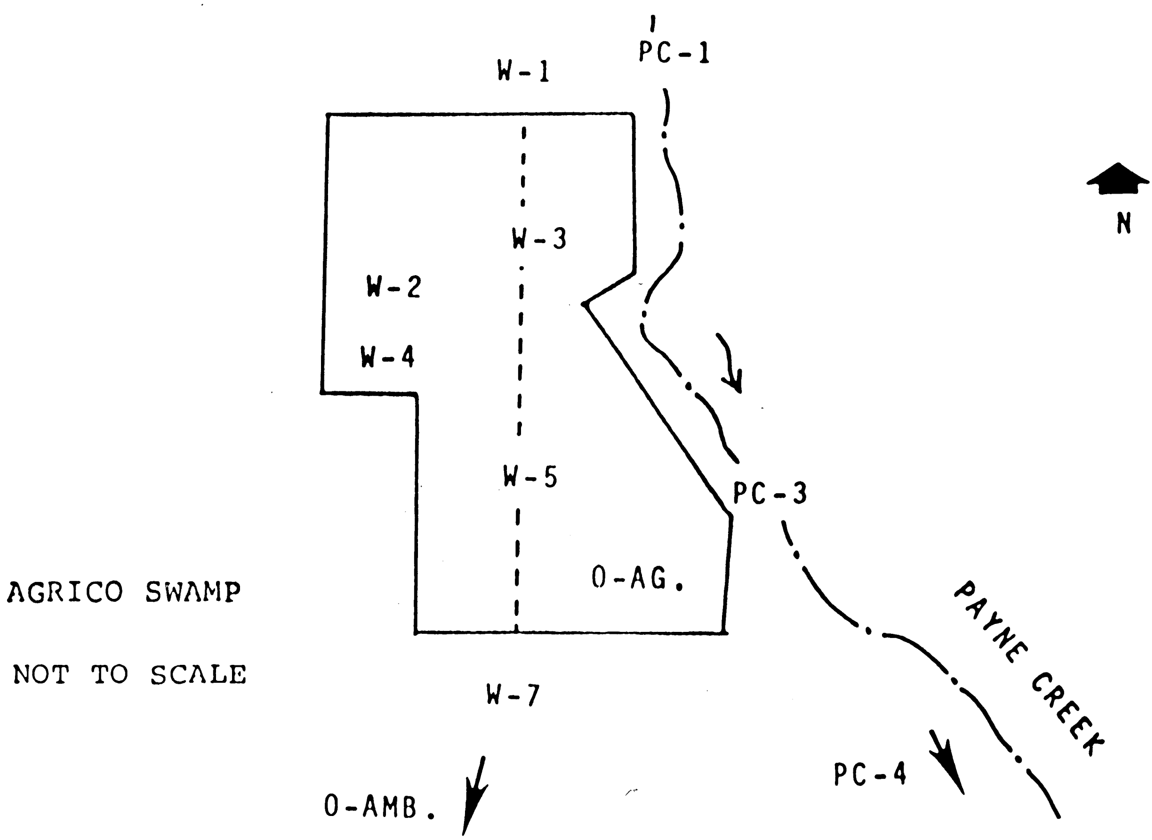


FIGURE 2 - SAMPLING STATIONS



General Description of Sampling Stations

Open water areas. The open water sampling site within Agrico Swamp (O-AG) is located at the southern end of the large marsh and swamp that extends along the eastern side of the project (Fig. 2). At this location, samples were taken from the top and bottom (-3.0 m) of the water column. An ambient station (O-AMB) in a natural open water area located approximately 8 kilometers south of Agrico Swamp in Hardee County was sampled in a similar manner.

Payne Creek. Payne Creek is located north and east of Agrico Swamp and flows in a southerly direction to the Peace River. Water sampling stations are located in the vicinity of State Road 37, where the creek enters Agrico property northwest of Agrico Swamp (PC-1), immediately downstream of the Agrico Swamp overflow discharges (PC-3), and in Hardee County where the creek leaves Agrico property (PC-4).

Wells. Within Agrico Swamp, two wells (W-2 and W-4) are located in uplands on the western side of the project and two wells (W-3 and W-5) are centrally located in transitional wetland areas. The off-site wells are located immediately outside the boundary of the project to the north (W-1) and south (W-7).

Water Sampling Procedure

Groundwater. Six wells were constructed to monitor the groundwater quality of the Agrico Swamp project. Details of the construction of these wells are in Table 1. An ISCO Model 2600 portable, compressed air operated, submersible well pump was used to evacuate the wells a minimum of one well volume before sampling. These wells were usually evacuated one day and sampled the next. The wells were pumped ten minutes before samples were collected in the ISCO sampler.

Surface water. Surface water samples were collected directly in sample containers by dipping.

Water samples were collected in new plastic bottles (nonreusable), except for dissolved oxygen which was collected in glass, and preserved as follows:

1. Metals were preserved with nitric acid to a pH < 2.0 and refrigerated.

2. Nitrogen samples were preserved with sulfuric acid to pH < 2.0 and refrigerated.

3. Dissolved oxygen samples were preserved with a manganous sulfate solution and an alkaline iodide-azide solution (Winkler method).

4. The temperature was measured in the field at the time of sampling.

5. All samples were refrigerated.

The water samples were then hand carried on the day of collection to a laboratory for measurement of selected chemical constituents and physical parameters according to Standard Methods (1980).

Table 1. Well Construction Specifications

Well #	Diameter	Casing Depth	Well Depth
1	15.24cm PVC	0-9m	11m
2	15.24cm PVC	0-9m	12m
		screen 9-12m	
3	15.24cm PVC	0-7m	10m
		screen 7-10m	
4	15.24cm PVC	0-9m	11m
5	15.24cm PVC	0-9m	11m
7	15.24cm PVC	0-6m	8m

Schedule 40 PVC used.

RESULTS AND DISCUSSION

Surface Water

Classification. The two open water areas and Payne Creek are classified by Chapter 17-3, Flroida Administrative Code (FAC) (1983) as Class III Waters—Recreation—Propagation and Maintenance of a Healthy, Well-Balanced Population of Fish and Wildlife. Criteria for compliance with this classification are contained in Chapter 17-3 FAC.

Open water. There is a marked contrast between the water quality of the Agrico Swamp open water association and the natural (ambient) open water association. The surface water pH in Agrico Swamp ranged from 7.9 to 9.8, while the ambient station ranged from 5.7 to 6.0. The low pH readings of the ambient station are similar to those found in other central and north Florida cypress lakes according to Best (1984) and Cowell (1984). State water quality standards (Chapter 17-3, FAC) permit a pH of 6.0 to 8.5. Therefore, the two open water areas are periodically outside of limits of the standard on opposite sides of the scale. Alkalinity measurements, in milligrams per liter (mg/l) substantiate the difference with values ranging from 73 to 115 from the Agrico Swamp station and 10 to 23 for the ambient station. State water quality standards for this parameter are not less than 20 mg/l. Within the two water bodies, fluctuations in pH and alkalinity do not appear to be interrelated. When alkalinity increases, a concomitant rise in pH is generally expected. This is not the case as can be seen in Figure 3. Additional sampling is necessary to verify this relationship and determine the cause of the high pH in the Agrico Swamp open water area.

The natural open water area station was below the standard of 5.0 mg/l dissolved oxygen, with recorded values ranging from 0.8 to 1.8 mg/l. Biological oxygen demand (BOD) ranged from 1.2 to 3.0 mg/l in the natural open water area. The Agrico Swamp open water area had values of 7.8 to 15.8 mg/l dissolved oxygen and 2.4 to 15.0 mg/l BOD. The unusually low dissolved oxygen in the natural open water area was probably caused by the respiration of dense aquatic vegetation and decomposing plant material.

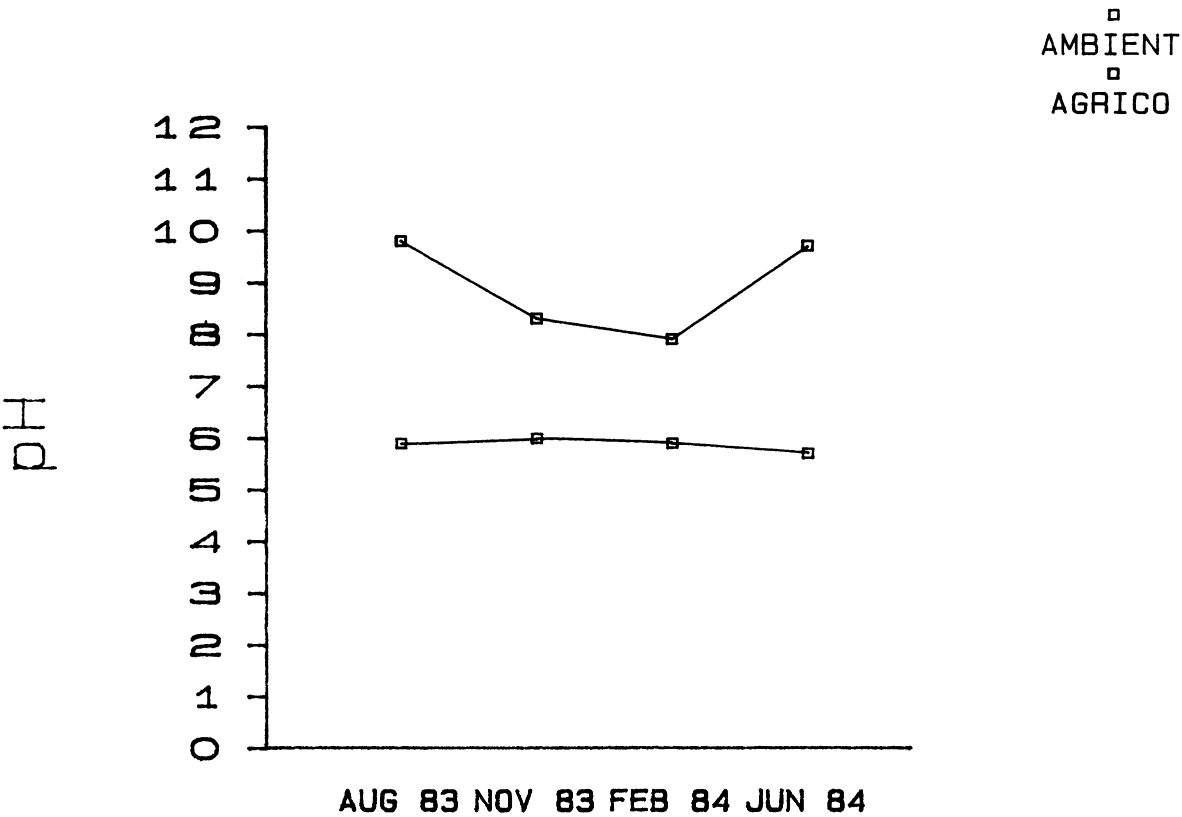
The most unexpected finding the the water quality assessment of these two bodies of water is that the nutrients total nitrogen (N) and total phosphorus (P) were relatively low. Despite the fact that algal blooms have been reported in Agrico Swamp by Ruesch (1984), the Agrico Swamp open water area station ranged from 0.67 to 2.50 mg/l (N) and 0.065 to 0.460 mg/l (P), and for the ambient station 0.62 to 2.67 mg/l (N) and 0.10 to 0.39 mg/l (P). These compare with the values obtained from seven wetland systems in nearby Orange County of 1.0 to 5.0 mg/l (N) and 0.15 to 0.39 mg/l (P) (Best, 1984). The unexpected low values for phosphorus in the Agrico Swamp open water area may be partially due to its insolubility in alkaline water as reported by Upchurch (1979). Future monitoring, as the reclaimed wetland matures and becomes more acidic, should reveal whether or not the high alkalinity of the Agrico Swamp open water area is the cause of the low phosphorus concentration.

Fluoride levels were also expected to be quite high. Upchurch (1979) reported the principle source of fluoride is from the predominate phosphate mineral carbonate fluorapatite. However, the values obtained ranged from 0.73 to 0.96 mg/l in the Agrico Swamp open water area and 0.54 to 0.84 mg/l in the natural open water area. These are all well below the state standard for fluoride of 10.0 mg/l in Class III fresh waters. The low solubility of apatite could explain the low levels of fluoride.

Gross alpha measurements in the Agrico Swamp open water area periodically exceed the state water quality standard of 15 picocuries per liter (pCi/l). The August, 1983, measurement of this parameter was 21.2 pCi/l and was as high as 39.4 pCi/l in December, 1982 (Erwin, 1983). In the three other sampling periods covered by this report, the values ranged from 0.8 to 2.8 pCi/l. The natural open water association values ranged from 1.6 to 6.4 pCi/l. The presence of radioactivity is natural in groundwater. Uranium and its breakdown products, radium 228 and 226 are associated with phosphate deposits in Flroida (Cathcart, 1966). Future investigations will be made to determine the relative contribution of uranium, radium 226, and radium 228 to the observed radioactivity.

The analyses for metals in these two open water associations showed low quantities and little variation, except that in August, iron was dectected in the natural open water association at a level of 0.93 mg/l. This approaches the state standard of 1.0 mg/l.

Dissolved solids, specific conductance, and turbidity are considerably higher in the Agrico Swamp open water area than in the natural open



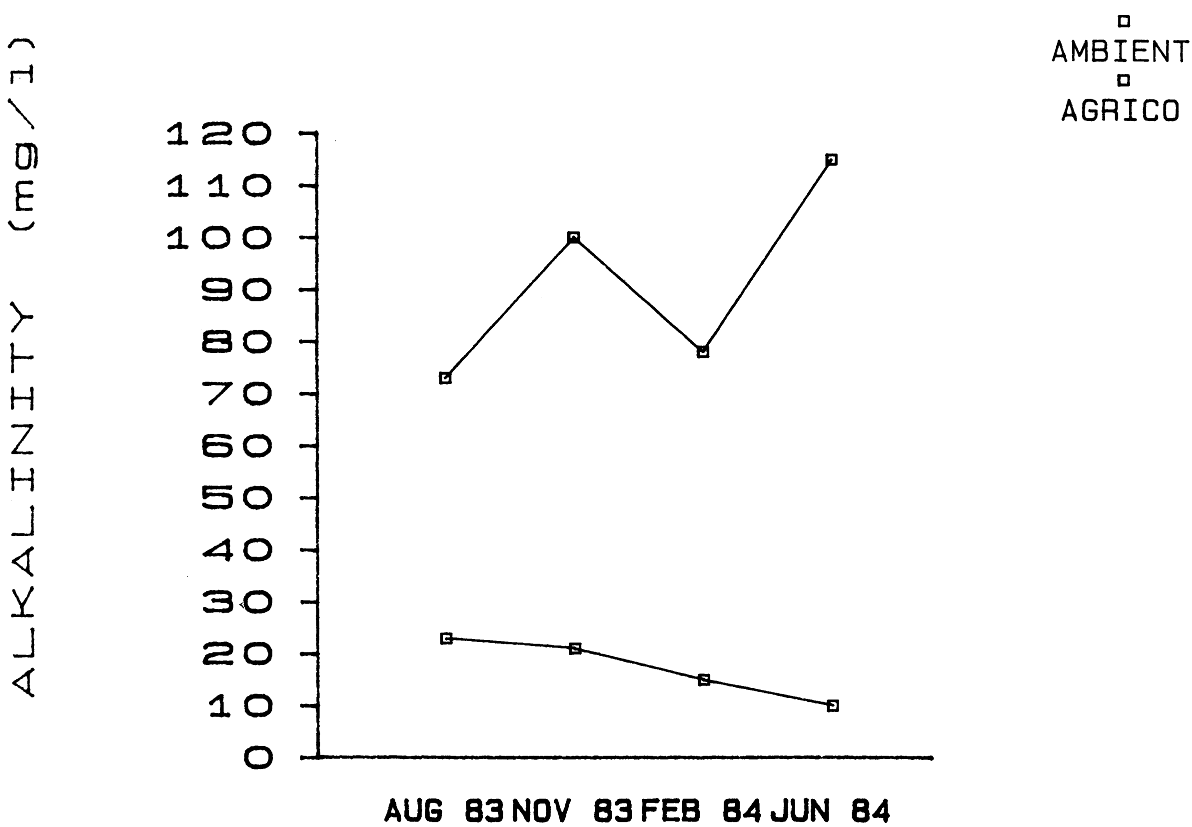


FIGURE 3 pH AND ALKALINITY

water association. This is probably due, at least in part, to the lingering effects of previous massive destabilization. Periodic blue-green algal blooms also affect turbidity and other water quality parameters. Future studies will include the analysis of chlorophyll and its impact on water quality. Other factors, such as rainfall, influence these and other parameters. In fact, rainfall could be responsible for peak values found in gross alpha, fluoride, phosphorus, and other parameters that may be associated with the virtual insolubility of phosphate in alkaline waters reported by Upchurch (1979).

Payne Creek. Water sampling data from Payne Creek during this reporting period are inadequate to fully evaluate the impact of project Agrico Swamp on water quality in Payne Creek. Water quality parameters were measured upstream of the project (PC-1), just downstream of the Agrico Swamp overflow discharge points (PC-3) and approximately 9 kilometers downstream (PC-4), but only for the periods of November, 1983, and February and June, 1984. No data were available for August, 1983, which would be indicative of the peak discharge period from Agrico Swamp. Comparative water quality data from upstream and downstream of the Agrico Swamp discharge were limited to the parameters of pH, turbidity, total nitrogen, nitrate nitrogen, total phosphorus, and fluoride.

The pH of Payne Creek waters generally increased as it flowed downstream with average values during the recording period of 7.08, 7.19, and 7.60 at the three sampling stations. The highest value obtained (7.80) came from PC-4 in June, which is the end of the dry season. Although there are no data from PC-3 for August, it appears that discharges from Agrico Swamp did not significantly impact the pH in the creek. The values of pH obtained from PC-1 and PC-4, at that time, were 7.2 and 7.5, respectively.

Turbidity in the creek decreased downstream. Average values expressed in NTU were 2.75 (PC-1), 1.57 (PC-2), and 1.25 (PC-4).

Nutrient measurements of total nitrogen and nitrate nitrogen (in mg/l) also decreased downstream. Total nitrogen ranged from 0.48 to 12.70 upstream and 0.03 to 0.50 downstream. The highest values were all found in the June sampling period. The values found for total phosphorus in water samples from the creek averaged 0.45 mg/l and do not reflect any apparent trend. Most values were in the range of 0.4 to 0.5 mg/l, with extremes of 0.18 (PC-3) and 0.6 (PC-1) occurring in November.

Fluoride levels detected in the creek ranged from 0.370 mg/l (PC-4) to 1.88 mg/l (PC-1). These extremes both occurred in June. Observed differences in fluoride levels at the sampling stations do not appear to be significant, but will be measured statistically when more data are available.

During the next annual reporting period, additional parameters and another sampling station (PC-2) will be added just upstream of the Agrico Swamp discharge points. This will permit a clearer assessment of the impact of Agrico Swamp on the creek without the perturbations caused by adjacent landowners.

Groundwater

Classification. Groundwater from the four water quality monitoring wells inside Agrico Swamp (W-2, W-3, W-4, and W-5) and the two wells outside the project (W-1 and W-7) all contain far less than 3,000 mg/l dissolved solids and are therefore assigned a state water classification of G-1 (potable). This classification requires compliance with the water quality standards of Chapter 17-22 FAC (Public Drinking Water Standards) as well as Chapter 17-3 FAC (Water Quality Standards).

Water quality. The pH of groundwater varied from 6.3 to 7.6 inside Agrico Swamp and from 6.2 to 7.9 outside the project. Groundwater flow into the Agrico Swamp open water area does not appear to contribute to the high pH values observed in the Agrico Swamp open water area. However, alkalinity in the project ranged from 153 to 385 mg/l with an arithmetic mean of 238 mg/l and outside the project alkalinity ranged from 43 to 168 mg/l, with a mean of 104.5. Specific conductance (in umhos) was considerably lower in off-site well samples which averaged 261 while the on-site samples averaged 540. This was the expected result from the previous earthmoving activities. As the area stabilizes, the specific conductance is anticipated to decline.

Total nitrogen was generally less than 1.0 mg/l in the wells. However, water from wells 2 and 3 consistently exceeded 10 mg/l and in June increased to over 40 mg/l. Nitrate nitrogen reflected a similar trend with an average of less than 0.1 mg/l in the groundwater from all wells except wells 2 and 3. These two wells had a slight increase in nitrate to about 0.5 mg/l in February and then jumped to over 20 mg/l in June. This exceeds the state water quality standard of 10 mg/l for nitrate nitrogen. A possible explanation for this finding is that Wells 2 and 3 were the first wells constructed for the project and unlike the other wells, had screens below the casings that could act as a substrate for biological growth. It appears that the high values for total nitrogen and nitrate nitrogen may be due to well contamination and/or sampling error.

Total phosphorus in the wells was generally less than 1.0 mg/l, but rose to 11.04 (February) and 1.83 (June) in Well 7, located south of the project. Well 7 also had a sizable fluctuation in iron, of 0.4 mg/l (November) to 10.4 mg/l (February). This was not unlike the variations shown in wells within the project. Well 1, located north of Agrico Swamp in an unmined area, was more stable, ranging from 0.12 to 0.68 mg/l. It appears that groundwater from Agrico Swamp does have an effect on Well 7, south of the project.

In August, the values for gross alpha (in pCi/l) exceeded state water quality standards in wells 1, 2, and 4, with readings of 22.1, 24.5, and 19.0 respectively. Otherwise, there were no findings above 4.2 pCi/l. Sutcliffe (1981) found that 60 percent of the wells tested in nearby Sarasota County exceeded radioactive contamination levels recommended by the U.S. Environmental Protection Agency (1976).

The fluoride levels were relatively low in all wells, ranging from 0.12 to 0.78 mg/l.

Copper, chromium, zinc, barium, lead, and beryllium, if detected, were in trace amounts except for one lead sample which may have been contaminated by sampling technique. A summary of the results of the other test parameters are shown in Table 2.

CONCLUSIONS

Open water. It is too early to make conclusions, but Agrico Swamp appears to be maturing and vegetative growth should have a buffering effect on the high alkalinity and moderate the peak values of other parameters. The increasing organic matter will permit retention of phosphorus by binding it in an organic matter-Al-PO_4 bridge as has been reported by Bloom (1981). However, the waters of Agrico Swamp and the present natural open water station will not be directly comparable. The off-site open water association is being overgrown with dense emergent aquatic vegetation and only a small open water space remains. This accounts for many of the observed differences in water quality.

Payne Creek. Payne Creek has been altered considerably from its natural state upstream of Agrico Swamp. The headwaters and watershed area of the creek have been mined and the creek channelized upstream of Agrico Swamp. Recent water management practices by adjacent landowners in that area have further decreased the flow into Payne Creek. These conditions have been aggravated by the fact that the rainfall during this reporting period was only 44.9 inches. The average annual rainfall for the area as reported in Erwin (1983) was 51.77 inches. Improvements noted in water quality as Payne Creek flows through the Agrico property can be primarily attributed to increased quantity and flow rate of the water moving downstream.

Groundwater. With the exception of the high levels of nitrate and total nitrogen found in two of the wells that were probably due to well contamination, the groundwater quality in Agrico Swamp is quite good considering the recent surface mining of the area. There is some evidence that the mining and recontouring of Agrico Swamp may still be having minor effects on the groundwater immediately south of the project. This is expected to continue to subside and will continue to be monitored through the existing wells and four additional monitoring wells located south of the project in Hardee County.

Future reports will expand the scope of water quality monitoring by adding an additional sampling station on Payne Creek and four additional monitoring wells south of Agrico Swamp. Water quality test parameters are also being expanded, as previously mentioned, and will be reviewed in the next annual report.

ACKNOWLEDGEMENTS

This research is being done to provide the phosphate mining industry and environmental regulatory agencies with proven wetland reclamation guidelines. Agrico's continued support for this project makes this research possible. We are indebted to Randy Mathews, Dale Carson, and

TABLE 2 Water Quality Summary
Range and Mean Values

	Agrico Swamp Open Water	Agrico Swamp Wells	Ambient Open Water	Ambient Wells	Payne Creek PC-1	Payne Creek PC-3	Payne Creek PC-4
Temperature (°F)	66-94 (79.8)	72-86 (78.8)	63-84 (72.3)	68-83 (75.0)	-	58-80 (67.7)	-
pH	7.9-9.8 (8.9)	6.3-7.6 (6.9)	5.7-6.0 (5.9)	6.2-7.9 (7.0)	6.8-7.4 (7.1)	7.1-7.2 (7.2)	7.5-7.8 (7.6)
Dissolved Oxygen (mg/l)	7.8-15.8 (12.1)	0.1-3.0 (1.6)	0.8-1.8 (1.0)	1.4-4.5 (2.4)	-	1.7-9.5 (5.6)	-
Turbidity (NTU)	1.5-18.0 (9.2)	0.5-110.0 (38.8)	0.5-1.5 (1.0)	0.5-125.0 (27.1)	0.5-4.0 (2.3)	0.8-2.8 (1.6)	0.5-1.8 (1.3)
Specific Conductance (UMHOS)	116.0-358.0 (249.5)	148.0-385.0 (250.8)	78.4-237.0 (126.6)	115.0-491.0 (261.6)	-	244.0-860.0 (468.7)	-
Alkalinity (mg/l)	73-115 (91.5)	153-783 (255.8)	10-23 (17.3)	43-168 (104.5)	-	46-57 (51.0)	-
Bio Oxygen Demand (mg/l)	2.4-15.0 (7.4)	1.2-31.0 (15.5)	1.2-3.0 (2.3)	0.4-35.0 (7.3)	-	0.4-3.0 (1.5)	-
Gross Alpha (mg/l)	0.8-21.2 (6.6)	0.19-24.5 (4.0)	1.6-6.4 (3.1)	0.1-22.1 (3.7)	-	0.7-4.4 (2.1)	-
Nitrate Nitrogen (mg/l)	<0.02 (<0.02)	<0.02-25.0 (<2.9)	<0.02-0.06 (<0.05)	<0.05-0.11 (<0.06)	0.43-5.90 (2.65)	0.39-1.25 (0.85)	0.03-0.50 (0.20)
Total Nitrogen (mg/l)	0.67-2.50 (1.7)	0.20-41.4 (15.0)	0.62-2.67 (1.3)	0.09-0.91 (0.34)	0.48-12.70 (4.7)	0.45-2.65 (1.3)	0.02-0.92 (0.4)
Total Phosphorus (mg/l)	0.07-0.46 (0.31)	0.02-1.34 (0.40)	0.10-0.39 (0.32)	0.44-11.04 (1.84)	0.42-0.60 (0.51)	0.18-0.50 (0.36)	0.28-0.58 (0.45)
Fluoride (mg/l)	0.73-0.96 (0.82)	0.12-0.78 (0.45)	0.54-0.84 (0.63)	0.32-0.68 (0.49)	1.69-1.88 (1.78)	0.59-1.67 (1.15)	0.37-1.78 (1.36)
Iron (mg/l)	<0.02-0.07 (<0.45)	0-17.80 (4.30)	0.28-0.93 (0.61)	0.12-10.40 (0.83)	-	0.05-1.56	-
Dissolved Solids (mg/l)	172-220 (189.0)	202-618 (343.7)	84-122 (103.3)	74-224 (158.8)	-	202-284 (246)	-

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ESTABLISHMENT OF NATIVE HAMMOCK VEGETATION ON SPOIL ISLANDS DOMINATED BY AUSTRALIAN PINES

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ABSTRACT

Spoil islands constructed during dredging of the Intracoastal Waterway and other navigable channels constitute a significant man-made habitat type in east-central Florida and offer unique opportunities for upland habitat enhancement. To study the feasibility of native vegetation establishment on these islands, 19 native species and one ornamental exotic species were planted on an upland portion of a spoil island in the Indian River near Vero Beach. Seedlings from two-inch liners were planted beneath the Australian pine (*Casaurina equisetifolia*) canopy, with no soil preparation, or subsequent fertilization or watering.

After 19 months, species' survival rates ranged from 100 percent for pigeon plum (*Coccoloba diversifolia*) and Spanish stopper (*Eugenia foetida*) to 27 percent for scaevola (*Scaevola frutescens*) and West Indies mahogany (*Swietenia mahogoni*). Net survival was 74 percent (221 of 300 seedlings planted). Extremely high tides during a severe fall storm inundated portions of the planting area (37 seedlings) and buried 79 seedlings beneath vegetative debris, dramatically increasing seedling mortality in the affected areas. All species exhibited substantial growth ranging from 74 percent (West Indies mahogany) to 933 percent (wild lime, *Zanthoxylum fagara*).

Disturbance of the Australian pine litter and soil surface facilitated invasion by Brazilian pepper (*Schinus terebinthifolius*), which soon dominated an adjunct live oak acorn (*Quercus virginiana*) planting site. Preservation of an intact soil and litter layer, and retention of at least a moderate canopy of Australian pines appeared vital to successful vegetative restoration on these spoil islands.

INTRODUCTION

Land development and human population growth during the last few decades have altered the landscape to the extent that major upland community types are endangered or, in fact, no longer exist in some areas of coastal Florida. Wetland protection laws have indirectly contributed to loss of upland habitats by forcing landowners to concentrate development activities in uplands while preserving wetlands, or even by requiring that wetland disturbances be mitigated by conversion of existing upland habitats into wetlands. Because of these threats to upland communities, opportunities for upland habitat creation or restoration should be emphasized in land use planning and major development permitting processes.

Upland habitat restoration on spoil derived from dredging projects has been studied by numerous agencies, especially the U.S. Army Corps of Engineers (Hunt et al., 1978, Lunz et al., 1978, Ocean Data Systems, 1978). Most of these efforts have concentrated on temperate-zone plant species of proven value for wildlife or soil erosion prevention. Beaman (1973) and Carlson (1972) reviewed vegetative patterns on southwest Florida spoil islands. Other than shorebird or colonial wading bird utilization (Lewis & Lewis, 1978, Schreiber & Schreiber, 1978, Soots & Landin, 1978), there has been little study of upland wildlife habitat development on spoil islands in Florida.

The Indian River is a 195 km-long shallow estuarine lagoon separating the mainland from a series of barrier islands along Florida's east central coast. Spoil islands are numerous, with over 60 major islands in the 45 km lagoon segment between Sebastian and Fort Pierce inlets. Most of these islands were entirely created by deposition of spoil dredged from the Indian River during the excavation of the Intracoastal Waterway, although a few are natural islands which were spoiled upon. Additional description of these islands is found in reports by Indian River County (1980) and Fernald et al. (1982).

The floral and faunal diversity of the area's coastal hammocks is impressive, especially when contrasted with their spoil island counterparts (Fernald et al., 1982). During our surveys of natural islands we were also impressed by the number of native tropical plant species spreading into the understory and groundcover of Australian pine dominated spoil sites adjacent to native coastal hammocks. These observations prompted us to investigate the feasibility of establishing native hammock vegetation on spoil islands in the Indian River. Gilmore (Indian River County, 1980) reported colonization of these islands by native species, and discussed the potential for revegetation to improve wildlife habitat conditions. Several publications have reviewed ways of propagating native plant species normally found in these hammocks (Gann, 1979, Watkins & Sheehan, 1969, Workman, 1980).

STUDY SITE

Our study site is typical of the county's spoil islands. It covers approximately 1.8 hectares and is about 300 meters from the mainland shoreline. It has a wetland fringe and transitional area averaging 15 meters in width which is best developed on the southern and western shorelines. This zone is dominated by red mangrove (Rhizophora mangle), black mangrove (Avicennia germinans), white mangrove (Laguncularia racemosa), and buttonwood (Conocarpus erecta). The northern and eastern shorelines are partially eroded, with rocky oyster reefs occurring in the nearshore shallows, and numerous windfallen Australian pines along the shoreline. The island has little topographical relief, being essentially a low dome with maximum elevation of about 1.5 meters above mean sea level.

Much of the island uplands support a near monoculture of 10 to 15 meter tall Australian pines, with Brazilian pepper as the primary, although widely scattered, understory species. Light penetration to the forest floor is approximately 25 to 75 percent. Australian pine litter covers the forest floor, with a 5 to 10 cm deep sub-layer of decomposing pine debris. Below this, the sediment is sand and shell, with an apparently declining organic content.

Gilmore (Indian River County, 1980) reported a depauperate terrestrial flora for this island, including only 12 species. Gumbo limbo (Bursera simaruba), Florida privet (Forestiera segregata), and Hercules-club (Zanthoxylum clava-herculis) are the most common native woody species, but strangler fig (Ficus aurea), cabbage palm (Sabal palmetto), and seagrape (Coccoloba uvifera) also occur. All six species occur as scattered individuals or groups of seedlings (gumbo limbo), however, and only a few Florida privets and strangler figs are over two meters tall.

MATERIALS AND METHODS

Table 1 presents the species we selected for our test planting. The first 11, including black ironwood (Krugiodendron ferreum), gumbo limbo, inkwood (Exothea paniculata), Jamaica dogwood (Piscidia piscipula), lancewood (Ocotea coriacea), pigeon plum, seagrape, strangler fig, West Indies mahogany, wild mastic (Mastichodentron foetidissimum), and willow bustic (Dipholis salicifolia) are often emergent or canopy species in tropical hammock or transitional scrub communities. All but Jamaica dogwood, West Indies mahogany, and willow bustic occur naturally in Indian River County. Three of these, including gumbo limbo, seagrape, and stangler fig occur naturally on our study island. The next eight species are small trees or shrubs often found in coastal or tropical hammocks. These include Jamaica caper (Capparis cynophallophora), myrsine (Rapanea punctata), necklace pod (Sophora tomentosa), Spanish stopper, white indigo berry (Randia aculeata), white stopper (Eugenia axillaris), wild coffee (Psychotria nervosa), and wild lime. They all occur naturally in Indian River County, but none were previously found on this island. Finally, we selected one herbaceous shrub; that being scaevola, or inkberry. This is an ornamental exotic species often planted in south Florida. The test species were selected primarily for their availability, low cost, and, for the most part, their significance in the region's native flora. Jamaica dogwood, West Indies mahogany, and willow bustic were included because of their importance in Florida Keys hammocks.

Fifteen individuals of each species were purchased from a Florida Keys native plant nursery. All seedlings were in two-inch liners and ranged in height from 2 to 28 cm. Our planting site was arranged in a 15 x 20 rectangular grid, using nylon cord to mark one-meter centers for seedling placement. The seedlings were well-soaked with fresh water, then transferred to the island and hand-planted. The soil and surface litter were disturbed as little as possible.

Table 1. Species planted on spoil island test site in September 1983.

SPECIES	OCCURS NATURALLY IN INDIAN RIVER COUNTY (Fernald et al. 1982)	FOUND ON INDIAN RIVER CO. SPOIL ISLANDS (I.R.Co. 1980)	OCCURS NATURALLY ON STUDY ISLAND
Black Ironwood *Krugiodendron ferreum*	X	X	
Gumbo Limbo *Bursera simaruba*	X		X
Inkwood *Exothea paniculata*	X		
Jamaica Dogwood *Piscidia piscipula*			
Lancewood *Ocotea coriacea*	X		
Pigeon Plum *Coccoloba diversifolia*	X		
Seagrape *Coccoloba uvifera*	X	X	X
Strangler Fig *Ficus aurea*	X	X	X
West Indies Mahogany *Swietenia mahagoni*			
Wild Mastic *Mastichodendron foetidissimum*	X		
Willow Bustic *Dipholis salicifolia*			

Jamaica Caper *Capparis cynophallophora*	X		
Myrsine *Rapanea punctata*	X		
Necklace Pod *Sophora tomentosa*	X	X	
Spanish Stopper *Eugenia foetida*	X	X	
White Indigo Berry *Randia aculeata*	X		
White Stopper *Eugenia axillaris*	X	X	
Wild Coffee *Psychotria nervosa*	X	X	
Wild Lime *Zanthoxylum fagara*	X		

Scaevola *Scaevola frutescens*	(ornamental exotic)		

No water or fertilizer was applied during the 19-month study. The only maintenance performed was to casually remove Brazilian pepper seedlings from the planting grid and removal of Australian pine debris which occasionally covered a few seedlings.

Seedling condition was monitored throughout the study. On each monitoring date every plant was assigned to one of the following condition classes: (1) seedling apparently healthy, with less than 50 percent of leaves showing damage; (2) seedling revealing recent loss or extensive damage to at least 50 percent of leaves; (3) seedling leafless but with green or supple stem and green buds; (4) seedling showing no apparent signs of life; and (5) seedling missing or "dead" for at least two consecutive condition reports. We made no attempt to distinguish between normal, anticipated events such as loss of leaves by deciduous species, and seedling condition declines caused by freeze damage, insects, or other causes.

Each surviving seedling was measured at 4 days, 1 month, 6 months, 12 months, and 19 months post-planting. Height to the uppermost terminal bud was measured to the nearest 0.5 cm during the first 6 months and to the nearest centimeter thereafter.

As an additional experiment, we collected several thousand live oak acorns from a mainland hammock to see if they would germinate and survive on the spoil island. We chose a barren area typical of the forest floor, raked the surface litter aside, and sowed the acorns over the prepared bed. We then raked the surface litter back over the planting site. Adjacent to that planting strip, we scattered acorns on the ground with no soil preparation.

RESULTS AND DISCUSSION

Seedling Survival and Condition

After 12 months we had a net survival rate of 91 percent (273 of 300 seedlings), which declined to 74 percent (221 of 300 seedlings) after 19 months. Seedling condition after 19 months, in April, 1985, is presented in Figure 1. Of the first six species including pigeon plum, Spanish stopper, Jamaica caper, white stopper, mysrine, and lancewood, we lost one or fewer of the 15 planted seedlings. All six of these species occur naturally in Indian River County. At the other extreme scaevola and mahogany suffered 73 percent mortality; that is, 11 of 15 seedlings per species were lost. Neither of these species occurs naturally in Indian River County.

Condition-time profiles were constructed for each species, and we found that the 20 species could be grouped into 6 general categories according to these curves. The mahogany seedlings (Fig. 2) died at a rather steady rate throughout the study. There were two peak leafless periods, closely following the severe freezes of December, 1983, and January, 1985. Since mahogany is a deciduous species, this was anticipated. The condition decline in 1984 began in November, in response to a severe northeaster which passed through Indian River County during the 21st through the 23rd of November. This sharp condition decline and

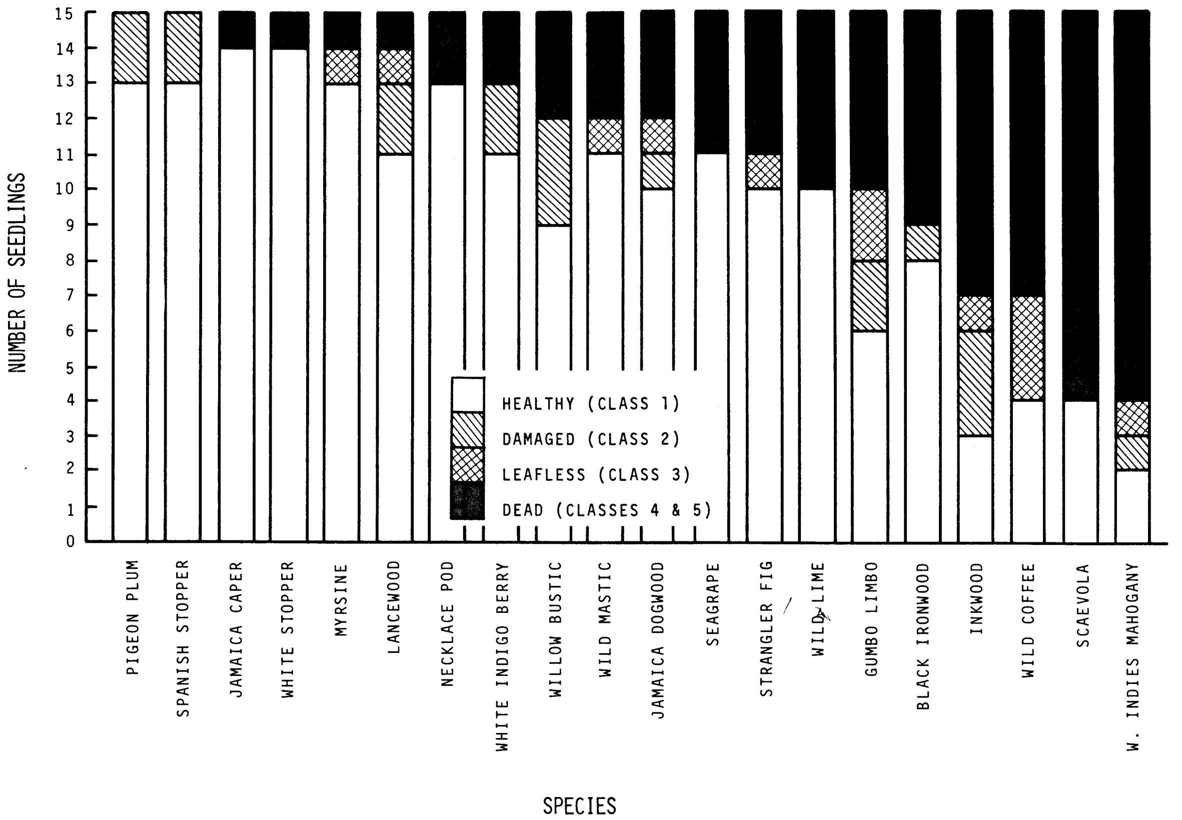

Figure 1. Seedling condition after 19 months - May 1985.

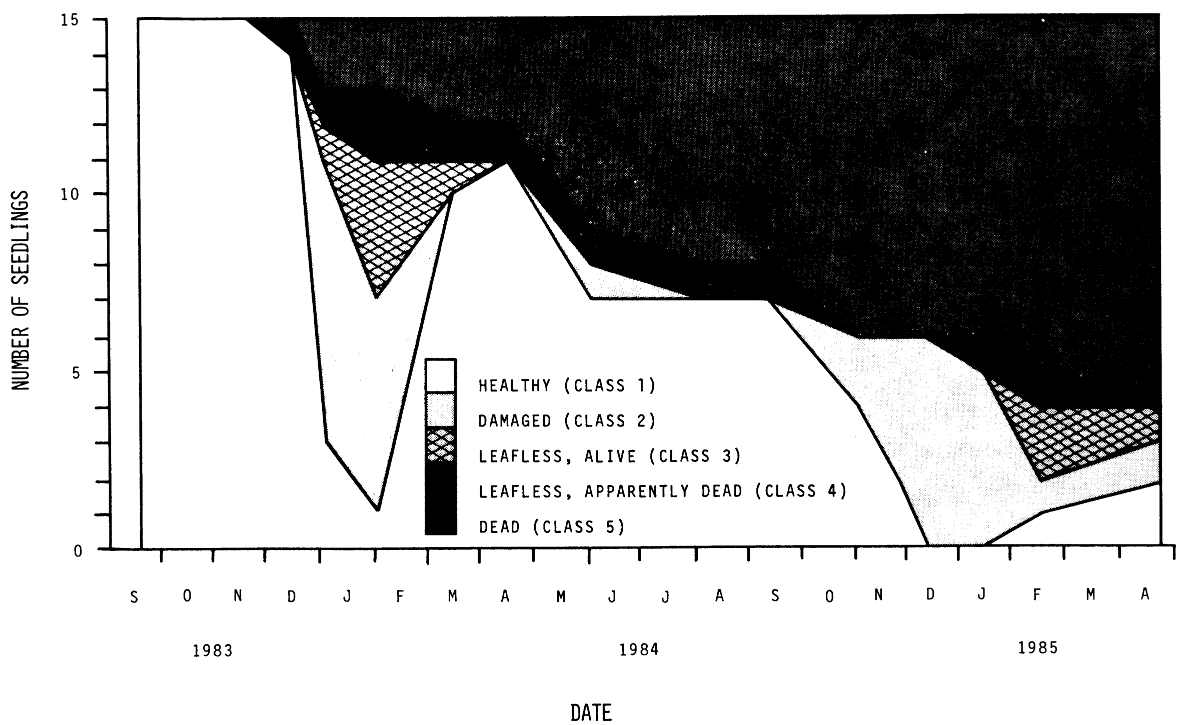

Figure 2. West Indies mahogany condition-time profile.

partial recovery after severe weather, and rather steady seedling mortality throughout the study were also characteristic of scaevola. The survival rate of 19 months was 27 percent for both species.

Gumbo limbo (Fig. 3) represents the next group of species, which also includes Jamaica dogwood and inkwood. These species all showed an immediate response to the December, 1983, freeze, but largely recovered during the following growing season. Their condition also declined during the 1984-1985 winter and several seedlings of each of these species died during the study. Gumbo limbo and Jamaica dogwood are deciduous, and lose their leaves in response to winter drought. These two species are also well-known for their regenerative capacity, so it is likely that several of the seedlings now presumed dead will yet recover with the onset of regular summer rainfall. Gumbo limbo clearly reveals the effect of the November, 1984, storm as distinct from effects of the January, 1985, freeze.

The condition-time curve for wild lime (Fig. 4) is characteristic of the third group of seedlings which also includes black ironwood and willow bustic. These three species showed a moderate response to the first winter's freeze, but all seedlings recovered fully. They also responded negatively to the November, 1984, storm and second winter freeze, with survival ranging from 60 to 90 percent.

Wild coffe had a unique condition-time profile (Fig. 5). It was not affected by the first winter, but was severely damaged by both the November, 1984, storm and January, 1985, freeze. Wild coffee is abundant in Indian River County, but it is typically an understory species in mature hammocks. It may be unable to survive in such an unprotected setting.

The fifth group of four species is represented by wild mastic (Fig. 6), and also includes seagrape, strangler fig, and white indigo berry. No major, abrupt condition changes were noted for any of these species, and survival after 19 months ranged from 73 to 87 percent.

The most successful group of seven species included pigeon plum, Jamaica caper, lancewood, myrsine, necklace pod, Spanish stopper, and white stopper. None of these species suffered any major condition declines during the study, and survival over 19 months ranged from 87 to 100 percent.

As previously mentioned, a severe "northeaster" storm passed through Indian River County on 21, 22, and 23 November, 1984. These dates coincided with the full moon, and also with the fall period when sea level is about 30 cm higher than normal on Florida's east central coast. The resulting storm tides and waves overwashed portions of our planting area (Fig. 7). In the northeastern corner of our grid, 37 seedlings were inundated by tide and storm waves, and virtually all of the surface litter was washed away. Another 79 seedlings were covered with debris washed up by the storm. The debris ranged up to 35 cm in depth and consisted primarily of Australian pine litter, mangrove leaves and propagules, seagrasses, and drift algae. Despite this disturbance, by carefully locating seedling positions we were able to uncover nearly all of the buried seedlings, although a few had been washed away. Except for resulting "holes" in the debris, the wrack line was left intact.

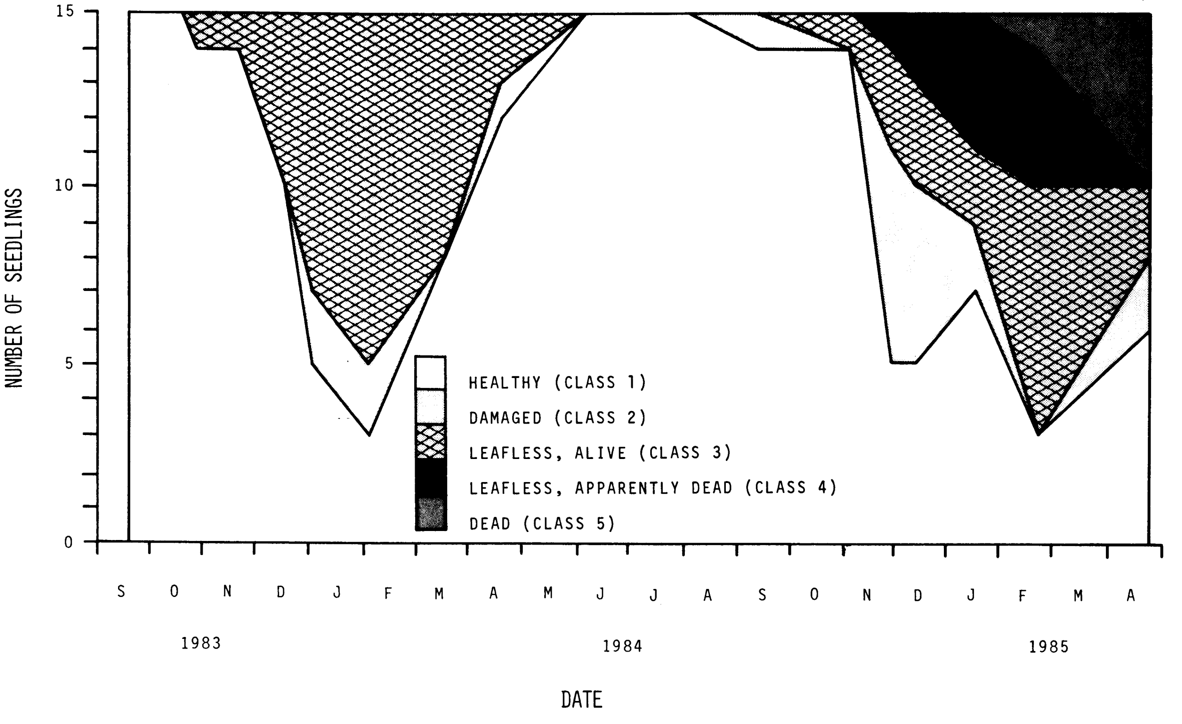

Figure 3. Gumbo limbo condition-time profile.

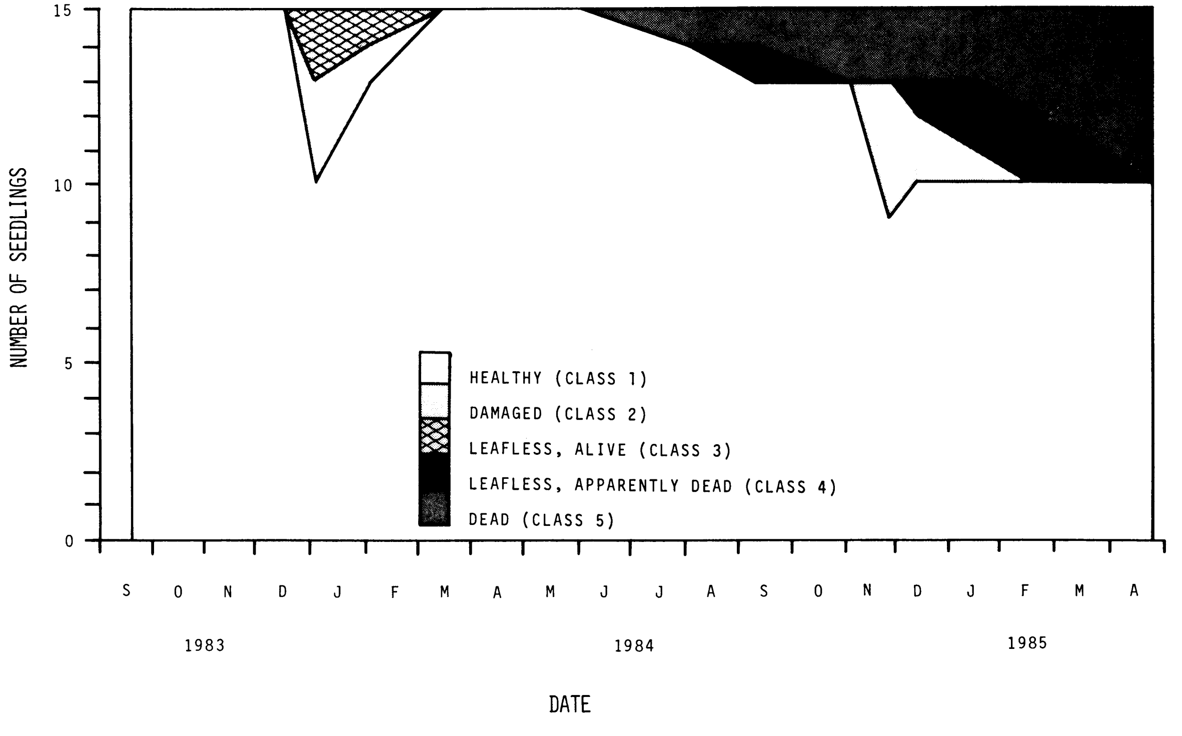

Figure 4. Wild lime condition-time profile.

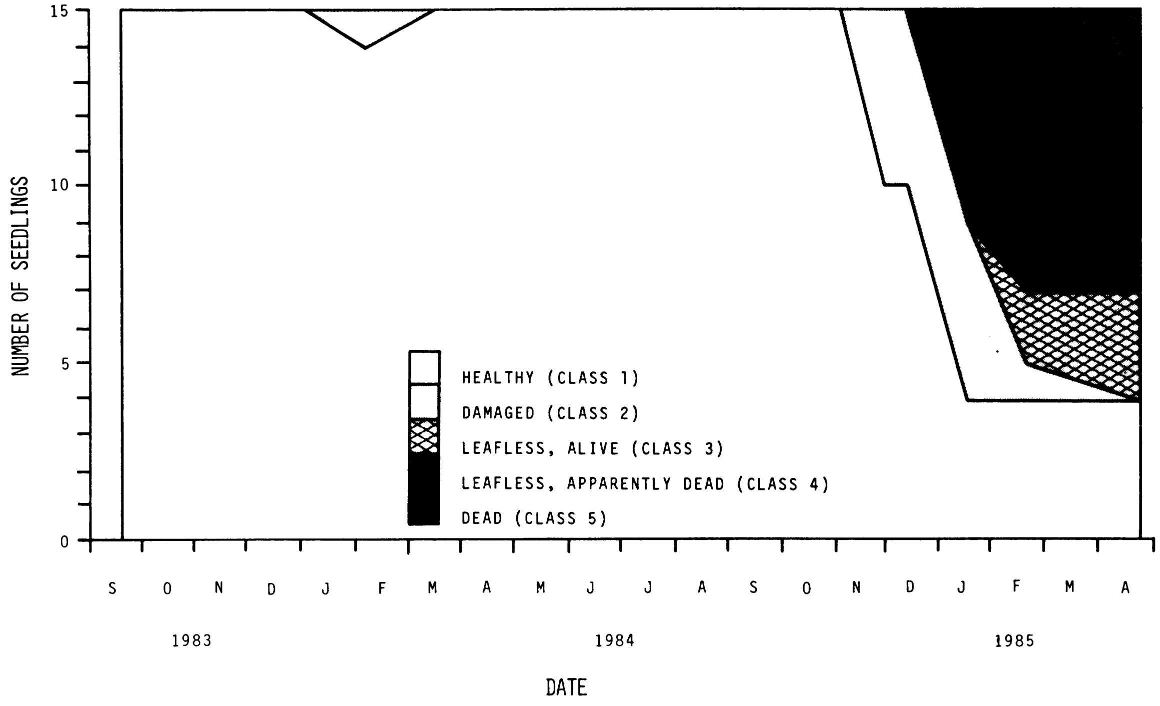

Figure 5. Wild coffee condition-time profile.

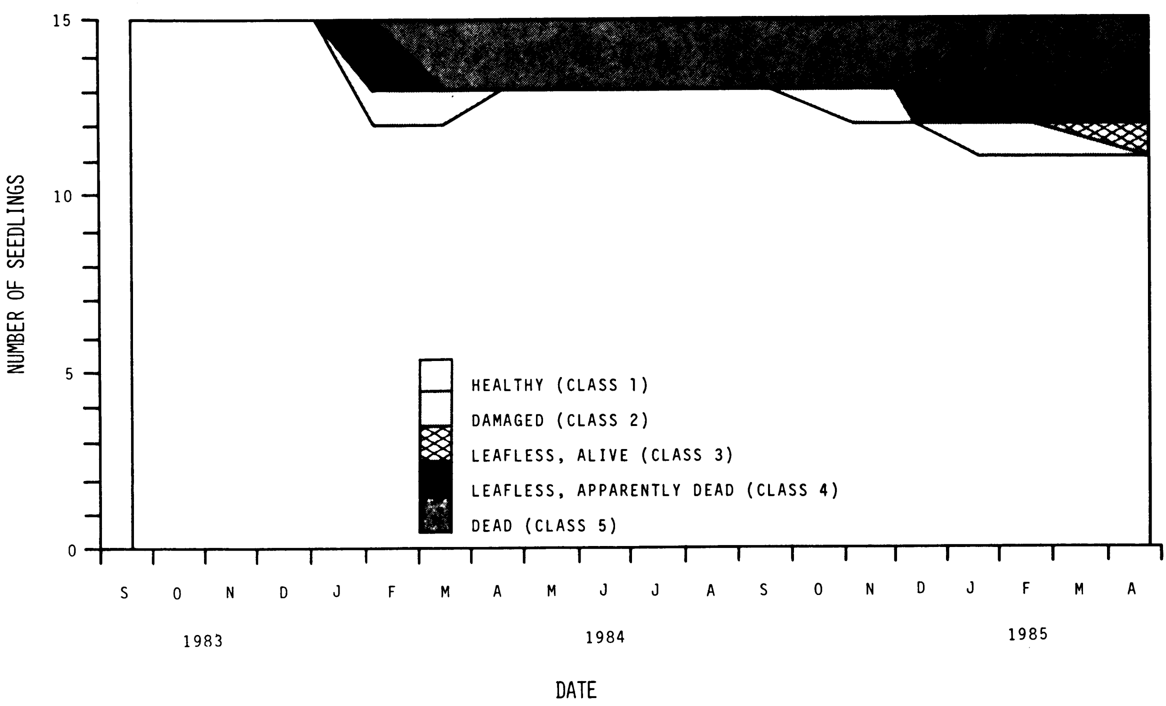

Figure 6. Wild mastic condition-time profile.

To determine the effect of the storm on survival rates, we plotted the dead or missing seedling locations on the grid. During the 19-month study, seedling survival in the undamaged zone was 84 percent. The survival rate was 66 percent in the debris zone, and only 38 percent in the inundated zone. Although those figures reveal the effects of the storm, we felt that species with low overall survival (less than 50%) may have been masking some of the storm's impact. Therefore, we deleted mahogany, scaevola, inkwood, and wild coffee from consideration. Of the remaining 240 seedlings, the survival rate in the undamaged zone was 94 percent, compared to 71 percent in the debris zone, and 52 percent in the inundated zone. This clearly illustrates the importance of locating such planting efforts above the anticipated storm zone.

Seedling Growth

Mean planting heights ranged from 5 to 21 cm, and mean seedling height after 19 months ranged from 10 to 70 cm (Fig. 8). All 20 species exhibited significant growth. Wild lime showed the greatest growth, averaging 933 percent growth from a planting height mean of 5 cm to an April, 1985, mean height of 61 cm. Other standouts included Jamaica dogwood (55 cm growth, 367%), stangler fig, (38 cm, 181%), necklace pod (36 cm, 600%), and black ironwood (32 cm, 267%). Most species exhibited growth of 100 to 300 percent over 19 months. West Indies mahogany was the only species with a growth rate lower than 100 percent, but even that was impressive (15 cm, 76%).

Live Oak Germination

Thirty-two oaks sprouted from the acorns planted on the island, but only nine survived the entire 19 months. No acorns germinated where they were simply scattered on the surface litter. The surviving live oaks averaged 10 cm in height after 19 months, and were being overgrown by Brazilian pepper seedlings which volunteered in the disturbed planting strip.

Project Design Implications

We chose not to disturb the Australian pine canopy or soil surface. The pines appear to be instrumental in soil development, and their continuous deposition of litter creates an undisturbed layer which prevents widespread germination of Brazilian peppers or Australian pines in the understory. The canopy also provided protection of our seedlings from cold winds, excessive salt spray, and extreme desiccation. We were encouraged by the lack of any apparent allelopathic effect of Australian pines on our seedlings.

Even minor soil disturbance often led to invasion by Brazilian pepper. For example, Brazilian peppers often sprouted in the holes where test seedlings had been uprooted. Also, rapid invasion of the live oak acorn planting site by Brazilian pepper testifies to the importance of disturbing the soil surface as little as possible during the planting process.

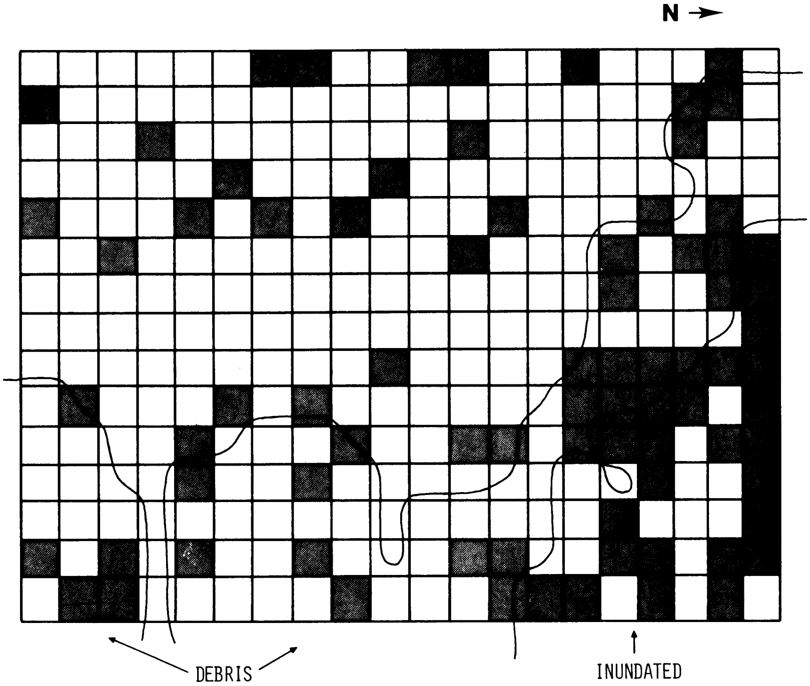

Figure 7. Detail of planting grid depicting storm damage zones and seedlings which died during study (shaded).

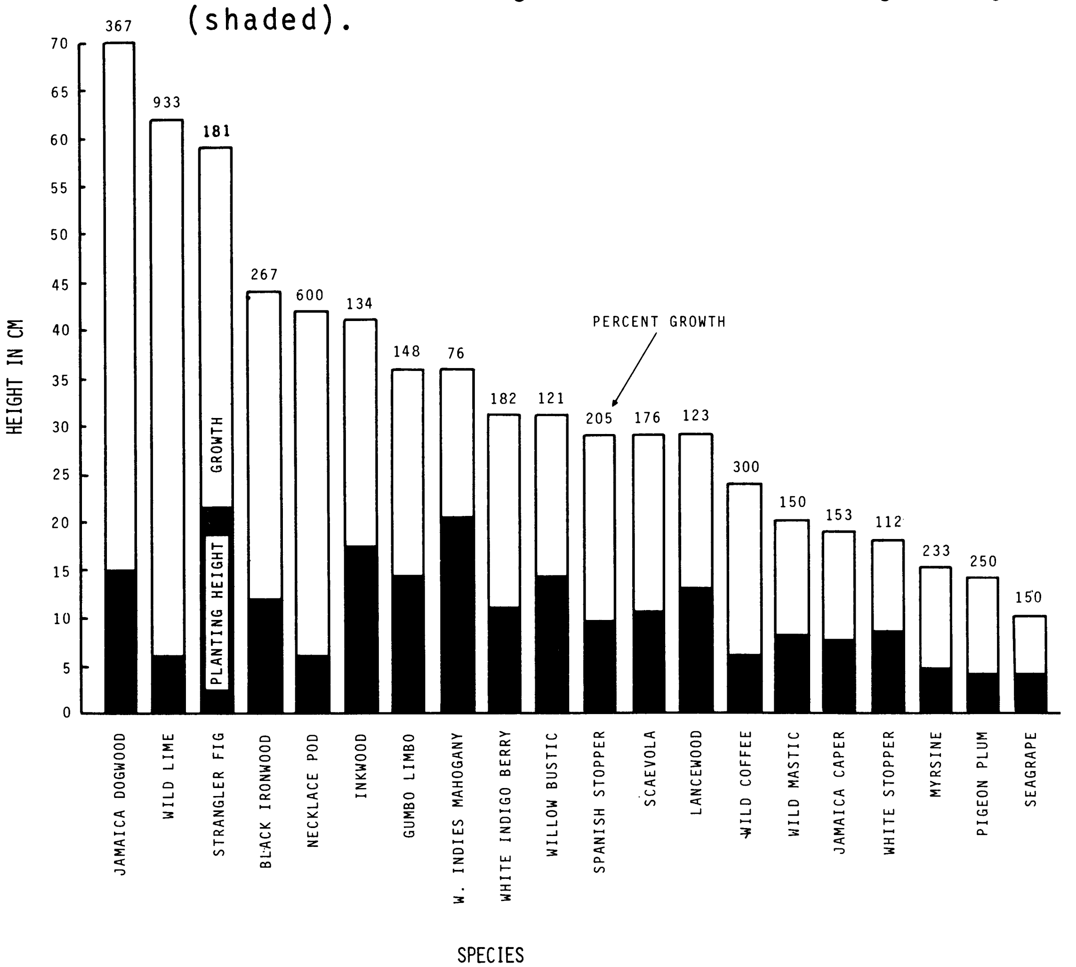

Figure 8. Average seedling height by species at planting, and growth over 19 months.

If viable seeds are present, they are likely to germinate and outcompete the planted species upon sufficient contact with soil and moisture. Even if viable seed is not present during the planting process, disturbance and mixing of the surface litter may create a suitable substrate for future seed deposition and seedling survival. Thinning out or clearing the uplands of Australian pines or other pre-existing vegetation, or even to prepare a planting bed by disturbing the surface soil, would probably have resulted in complete site takeover by Brazilian pepper.

Cost is a major factor in large scale restoration efforts, thus the desire for widely-available, low-maintenance species. Raw materials for our study cost approximately $100, equivalent to about $3,400 per hectare. Discounts for large volume sales, or germination of seeds from local sources could significantly reduce this figure, while selection of less common species such as red stopper (Eugenia confusa) could greatly increase the cost. Private revegetation or mitigation efforts may also involve considerable labor costs, while public projects could make effective use of volunteer labor by scout troops, schools, churches, and other civic organizations.

CONCLUSIONS

Our observations of native hammock species thriving under an Australian pine canopy on natural islands which were spoiled upon (research in progress), and the results of this study to date suggest that it is feasible to establish native hammock vegetation on spoil islands with relatively little expenditure of financial or manpower resources. Certain native species such as pigeon plum, Spanish stopper, Jamaica caper, white stopper, myrsine, and lancewood, all with over 90 percent survival, are recommended for additional planting trials in east-central Florida. The species with 60 to 90 percent survival over 19 months also deserve further study, since most of them occur naturally in Indian River County, and several exhibited outstanding growth (wild lime, necklace pod, Jamaica dogwood, strangler fig, and black ironwood). Of the remaining species, low survival rates for West Indies mahogany, inkwood, and scaevola reduce their suitability for revegetation efforts, although public appreciation for such species as mahogany may override other criteria. Wild coffee may best be reserved for more protected sites, perhaps as second-phase plantings in long-term projects.

Because of the small number of seedlings planted during our study, detailed statistical analysis of our data is inappropriate, and our results are best considered a trial run for further research of much greater breadth. Soil analyses, topography, light availability, rainfall, impacts of resident herbivores, and numerous planting technique variables warrant additional research.

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VEGETATIVE GROWTH PATTERNS IN PLANTED MARSHES OH THE VEGETATIVE EROSION CONTROL PROJECT

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ABSTRACT

The Vegetative Erosion Control Project consisted of establishing and monitoring a number of Spartina fringes within the Chesapeake Bay system in Virginia during 1981 to 1984. The 24 marsh sites of the project were selected on the basis of maximizing the diversity of environmental conditions under which salt marshes might be established. Because many of the marshes were exposed to damaging forces of the environment each year, maintenance planting was regularly practiced to maintain satisfactory fringes. Several of our sites were planted with Spartina alterniflora and S. patens; however, most contained S. alterniflora only. Various parameters of vegetative production were measured each year providing information on the early development patterns of our planted marshes. Comparisons of the data (base maps and vegetative samples) resulted in the following observations: (1) marshes of lower energy shorelines are more productive than those of higher energy shores; (2) Spartina alterniflora is much more productive in the higher regions of the intertidal zone, and best extends itself into lower elevations through secondary shoot production; (3) stem densities appear to be relatively greater in southerly-facing marshes; (4) culms are thickest in diameter in lower energy environments.

INTRODUCTION

Waterfront property owners rarely admire the effects of erosion on their land. Countless portions of shoreline along most coastal rivers attest to this fact, in that they are reinforced with revetments, bulkheads, and seawalls in attempts to halt bank erosion. Where the rivers are wider than several kilometers, large waves can be generated. These can quickly erode considerable widths of the bank. Under such fetch conditions, well-designed shoreline protection structures are essential to prevent loss of land. These structures, however, are expensive both to replace and (when in becomes necessary) to repair.

In the smaller tidal tributaries, wind-generated waves are greatly diminished. Erosion on these shores occurs mostly as a result of longshore currents and tidal action. Boat wakes may also contribute substantially to bank erosion in some cases. Many who own property bordering these waters choose to protect their land with the same costly fortifications as those placed on the more exposed shorelines. Quite often, though, a planted salt marsh fringe may be an effective, inexpensive, and attractive alternative to a bulkhead or revetment.

The use of vegetation for erosion control is not a new concept in the Chesapeake Bay area. Indeed, some of the first and longest-standing planted marshes in the country occur in this region (Knutson et al., 1981). Most of the older marshes, however, have not been monitored closely and records of year-to-year growth patterns are largely unavailable. There are exceptions for more recently planted marshes. Benner et al. (1982) and Woodhouse et al. (1974) have examined patterns of development in salt marshes planted within the last decade or so. These and similar studies (Knutson and Woodhouse, 1983; Knutson et al., 1981; Woodhouse, 1979) indicate certain factors which may be important in the establishment of a marsh and its success in abating erosion. A very critical determinant is the site's wave climate or fetch. Other conspicuous factors include: shore geometry, shore orientation, bank type and elevation, nearshore bathymetry, sediment type, and tide range.

The purpose of our research was to substantiate the existing data base, particularly in regard to the lower Chesapeake Bay system. We wished to evaluate the effects of planted marshes on previously unvegetated beaches, and were especially interested in their impact on the beach's historic erosion rate. The information produced by this research would be used to more effectively select shorelines which may be conducive to marsh development. Also, the results could assist landowners in deciding whether to attempt this less expensive means of shoreline stabilization or choose fixed physical structures.

STUDY SITE

Twenty-four marshes were planted during the course of this project on shores of the lower Chesapeake Bay and several of its tributaries (Fig. 1). Since exposure to waves was obviously a prime determinant of marsh success, we chose sites on the basis of evaluating an upper limit of fetch under which marshes could thrive. Seven of the sites were classified as low energy (average fetch exposure < 1.8 km), ten were medium energy 1.8 to 9.2 km average fetch), and seven were high wave energy environments (> 9.2 km avereage fetch). Also important were the compass orientations of the shorelines. Strong northeast winds are relatively common in the Chesapeake Bay area during certain times of the year, whereas winds from other directions are rarely as persistent or powerful. Consequently, to the extent possible, we attempted to maximize diversity in regard to site orientation.

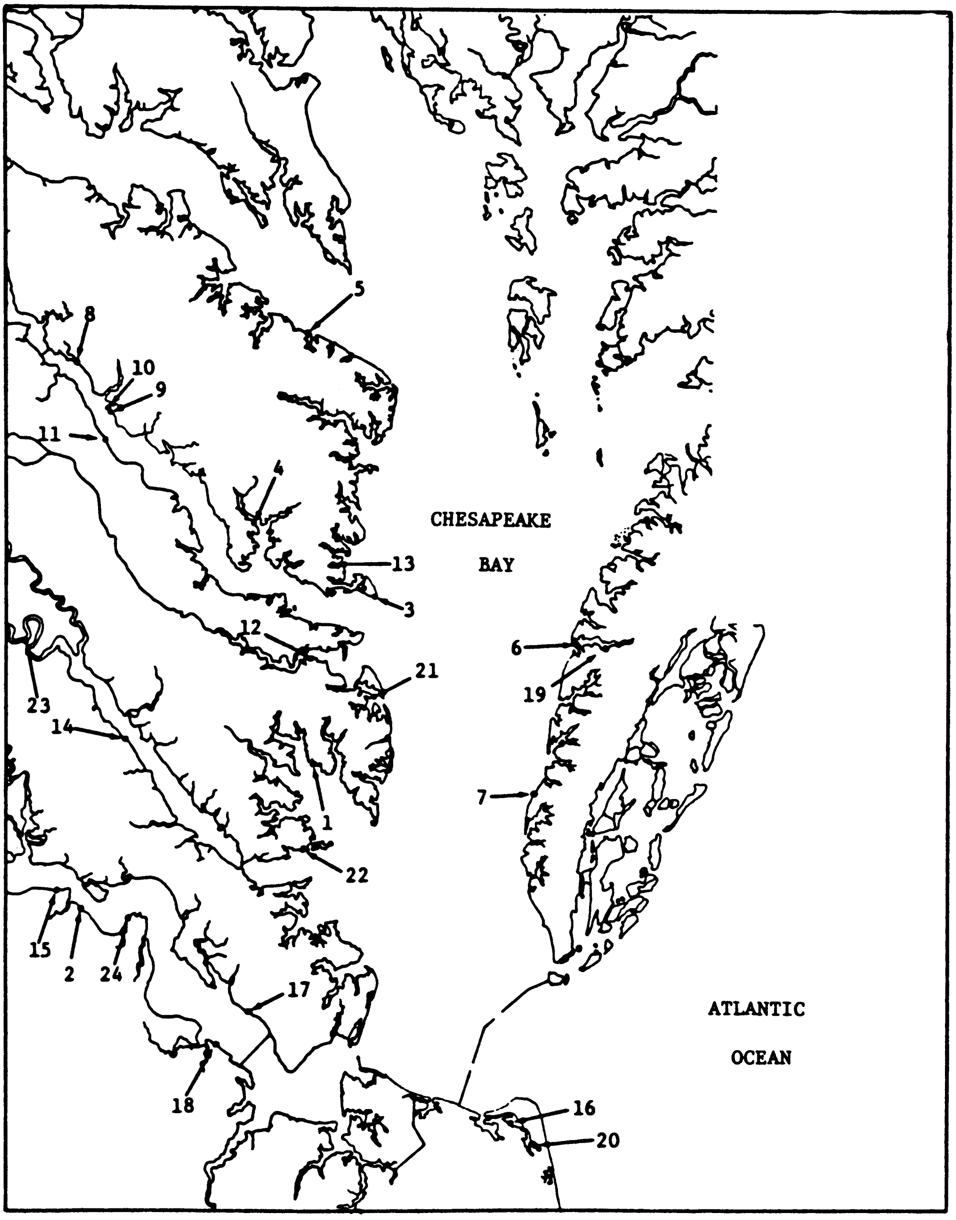

Figure 1. Locations of planted marsh sites.

MATERIALS AND METHODS

The Vegetative Erosion Control Project was initiated in the late spring of 1981 with the planting of 12 marshes. Young shoots of Spartina alterniflora growing in peat pots were planted into the beaches at each site. The plants were spaced at 0.5 m intervals in rows which extended from the elevation of mean high water down to an elevation between mean sea level and spring low water. Several of the sites were also planted with Spartina patens above the upper limit of the S. alterniflora. Approximately 30 ml of Oxmocote fertilizer (a 14-5.2-11.6, slow release mixture; see Broome et al., 1983) were poured into each hole dug on the beach before inserting the sprigs. In the ensuing two years, 12 additional sites were planted in similar fashion. Also, many sites which sustained damage were maintenance planted as necessary. Osmocote fertilizer was scattered over the marsh surfaces once or twice each summer to encourage vigorous growth of vegetation.

The marsh sites were monitored throughout each year. Accurately scaled maps were drawn for each site when it was initially planted using a plane table and alidade. Maps were revised or new maps created periodically thereafter whenever significant changes were observed in the marsh boundaries. Several times each year, elevations were taken on permanent profile lines at points within and outside the marshes using a level and a fixed point of reference. This data indicated areas where sediment had accreted or eroded. Each fall, below-ground and above-ground vegetation were sampled in order to deduce patterns of growth in the marshes. Below-ground samples were taken along transects across the marshes. At fixed intervals, cores were taken with a 6.5 cm diameter auger. Depths of root penetration were observed, then the roots and rhizomes were oven-dried and weighed. Above-ground vegetation was sampled by clipping stems at ground level within 0.25 m^2 areas. The height and basal diameter of each stem were recorded. The stem samples were then oven-dried and weighed. Two other measurements, "rhizome spread" and "plant spread," were also estimated at each site. Rhizome spread refers to the distance between the original sprigs and their most distant secondary shoots. Plant spread is the diameter of the clump of stems which emanate from the original plant. These were measured for each plant occurring along lines transecting the marshes.

RESULTS AND DISCUSSION

The establishment of a continuous fringe is essential if a marsh is to be effective in deterring backshore erosion. We found this difficult to accomplish where the average fetch was longer than 6 km. Even moderately strong winds, when sustained for an hour or more, are able to generate waves large enough to significantly scour a beach with this fetch. All of our marshes flourished as long as conditions were calm. When winds began to blow, however, those sites which were more exposed usually suffered to varying degrees. An average fetch greater than 2 km was often sufficient to impede growth of the marshes unless the shoreline geometry or nearshore bathymetry created a relatively protected environment. On these medium to high energy sites, we noticed that the most successful marshes were those which were widest and planted high in the intertidal zone. Several of our first year sites were planted where the fetch was greater than 20 km, and only one of these survived through the first few months. That particular site was very protected by the shallowness of the nearshore waters, and by a spit of land projecting from the northwest shore less than one km away. Nearly all of our sites with a fetch exposure less than two km were highly effective. These marshes were relatively undisturbed by wind waves. Excessive shading and extremely coarse substrate slightly inhibited marsh development at certain low energy sites.

The periodic mapping of our sites chronicled the changes in the marsh boundaries. We noticed that many of the marshes began expanding laterally and toward the backshore during their first summer. By the end of the project, many sites had Spartina alterniflora growing well above the mean high water level. The lower edges of most of the marshes, however, regressed soon after planting to an elevation nearly corresponding to mean tide level (see also, Broome et al., 1982). This was followed by either negligible change or expansion of the marshes through rhizome spread at low energy sites. Marshes at the medium energy sites, depending on their degree of protection, continued to show loss or progressed slightly toward the water. Elevation profiles revealed that as rhizome spread extended the lower marsh border, sediment would accumulate in front of the marshes. This forced the elevation of mean tide level away from the marshes, enabling them to expand without new shoots being subjected to limiting conditions.

The analysis of growth within the marshes was complicated by the maintenance procedures usually required at the sites. New sprigs were planted each year throughout most of the existing marshes in order to reestablish a continuous fringe. Newly planted areas often merged with the older marsh during the first summer. This made it difficult, if not impossible, to determine expansion in the originally planted vegetation. By the end of 1983, only three of the sites had recognizable areas of three year old marsh. Many of the sites, however, had supported two years of marsh growth. From information collected at these sites, we were able to recognize some trends in the early development of our planted marshes.

One important factor in marsh establishment is the time necessary for secondary growth to fill in the open areas of beach between the planted sprigs of grass. This growth in vegetation increases the stem density in the marsh, an important variable in evaluating a marsh's potential for wave reduction (Knutson et al., 1982) and sediment retention (Gleason et al., 1979). At many of our sites, stem densities increased rapidly throughout the first summer. By the end of the second growing season, plant spread and rhizome spread usually caused the originally planted sprigs to be indistinguishable in the marsh. There was a weak tendency for stem densities to be slightly greater in the southerly facing marshes of our project (Table 1). This may have been due to the longer daily exposure to sunlight that these sites have received.

Culm diameter is also an important consideration in wave reduction through a marsh (Knutson et al., 1982). In general, the diameters of the grass stems at our sites increased with decreasing fetch (Table 2).

Table 1. Stem density in relation to the site orientation of planted <u>Spartina</u> marshes in Virginia (data is only for those sites with stems remaining at the end of the 1983 growing season).

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                      Average Stem Density (stems/m^2)

Site  Orien-     100       200       300       400       500       600
No.   tation   ...|....:....|....:....|....:....|....:....|....:....|....:....|...

 19    N                                  *
 12    N                      *
 15    N                                      *
  4    NNE                        *
 20    NE                                  *
 16    NE                  *
  2    ENE                                 *
 11    ENE                                  *
 18    ENE                                     *
 21    E         *
 22    S                                                            *
 13    SSW                                           *
  9    SSW                                                            *
  8    SW                                   *
 17    SW                                          *
 10    W                                       *
 24    W                                  *
  6    NW                          *
 23    NNW                        *
```

Table 2. Culm diameter in relation to the average fetch at 24 planted *Spartina* marsh sites in Virginia.

```
                              Average Culm Diameter (cm)

Site    Average   0.0        0.2       0.3       0.4       0.5       0.6
No.    Fetch(km)   |..   //  ..|....:....|....:....|....:....|....:....|

 20        0.02                                      *
 13        0.07                                              *
 23        0.56                                                *
  4        1.11                                      *
 19        1.30                                              *
 16        1.48                                                     *
 12        1.55                                              *
 21        1.85                                      *
  9        2.70                            *
 22        2.78                                         *
 10        2.96                                         *
 14        3.70     --
  8        4.26                                 *
 15        5.55                            *
  2        5.74                       *
 24        5.92                                    *
 11        6.48                                 *
 18        9.44                            *
 17       10.55                            *
  6       20.35                               *
  3       24.79     --
  5       27.75     --
  1       28.86     --
  7       40.70     --
```

The correlation between culm diameter and fetch was much more pronounced than for stem density versus shore orientation.

Measurements of peak standing biomass were taken in our marshes to compare vegetation growth between sites and from year to year within a site. In general, high energy sites had little vegetation remaining at the end of the summer, whereas marshes of the low energy sites were the most prolific. Our data indicated that production was lower in the first year than in following years on low and medium energy sites. Throughout the fall, material which has been generated through photosynthesis is slowly translocated to the roots and rhizomes before the above-ground vegetation dies. New shoots produced after the plant's first year, then, have the advantage of stored nutrients and energy to spur their growth.

Information on below-ground production was collected for several reasons. The fibrous mat of roots and rhizomes helps stabilize the sediments, and is a particularly valuable asset when the protection afforded by the above-ground vegetation is minimal. Also, a healthy supply of below-ground vegetation is probably useful in initiating the production of numerous viable shoots during the following spring. Finally, vigorous rhizome growth is responsible for extending the marsh borders laterally as well as maintaining high stem densities within the marsh. In analyzing the below-ground biomass samples, we observed that vegetation of the low energy sites synthesized the greatest densities of roots and rhizomes. This was undoubtedly a consequence of alleviated stress. We also noticed that below-ground production was generally much greater in the upper portions of the marshes than at lower elevations. Plants higher in the marsh are more protected from wave action and less affected by inundation than plants lower in the marsh. This probably results in less stressed conditions, allowing the higher vegetation to be more productive. It is irrelevant that this pattern is contrary to that observed in studies of natural marshes (e.g., Valiela & Teal, 1974) because our sites receive balanced fertilization, are fringing marshes as opposed to extensive, and are newly developing as opposed to mature.

Spartina patens was generally a profitable addition above mean high water. It grew more densely than S. alterniflora, its recumbent nature giving the appearance of a wind-blown meadow. This form of growth caused S. patens to be very effective in accumulating sediment. Like the S. alterniflora, it was very productive and had the potential to quickly develop a gibrous network of roots. In contrast, though, the majority of S. patens stems persisted through the winters, impeding loss of the backshore at a time when wave action was most prevalent.

CONCLUSIONS AND RECOMMENDATIONS

A planted salt marsh fringe can be an excellent alternative to structural wave barriers in protected environments. The cost of establishing a marsh is relatively low and the effort required in maintaining the fringe is usually minimal. After the first year, noticeable reduction in backshore erosion is often observed. Marshes have the potential for not only preventing erosion of the backshore, but for encouraging sediment accretion. On the basis of our experience and the research done by others, we suggest that:

1. Marsh plantings should not be attempted, in most cases, where the average fetch exposure is greater than 6 km without the addition of offshore breakwaters of some sort (see Hardaway et al., 1985).

2. The width of the *Spartina alterniflora* fringe should be as great as possible, extending from no lower than mean sea level to an elevation between mean high water and spring high water. Often, the lower border of the marsh will gradually extend toward the water.

3. On beaches which are unvegetated above spring high water, *Spartina patens* may be planted from the upper limit of the *S. alterniflora* to the lower limit of the backshore vegetation or the toe of the bank, whichever the case may be. *Spartina patens* is effective in accumulating wind-blown sediment, thereby elevating the backshore.

4. Maintenance planting should be performed early in the growing season, as soon as large bare or sparsely vegetated areas are observed in the fringe. Gaps in the marsh smaller than one meter in diameter need not be replanted as they will usually fill in naturally before the end of the summer. Replanting larger gaps ensures maximum protection against backshore erosion.

5. Marshes should be fertilized twice each summer by scattering a suitable fertilizer (such as Osmocote) over the entire marsh surface during a spring low tide.

Marsh width, stem density, and culm diameter are the predominant variables in a marsh fringe's wave reduction potential. Unfortunately, there is little a landowner can do to increase the stem density or diameters in a marsh which receives adequate fertilization. Marsh width can often be maximized by planting both *Spartina alterniflora* and *S. patens*. A certain amount of time is occasionally necessary after a marsh is planted for a formerly eroding bank to attain a stable slope. Although some loss of bank may continue during this time, it is not reflection on the performance of the marsh, but rather on the instability of a steep bank.

For a detailed review of the changes which occurred in the beach profiles and in the marshes at each of our 24 sites, refer to our final report and its appendices (Hardaway et al., 1984).

ACKNOWLEDGEMENTS

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SEAGRASS TRANSPLANTING: 10 YEARS OF US ARMY CORPS OF ENGINEERS RESEARCH

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ABSTRACT

The U.S. Army Engineer Waterways Experiment Station (WES) and previously the Coastal Engineering Research Center (CERC) of the U.S. Army Corps of Engineers have been involved with seagrass transplanting research for over a decade. Corps interest in transplanting research centers around dredged material stabilization, habitat enhancement, development, and mitigation. Primary emphasis has been on development of low-cost transplanting techniques and site selection and design criteria. The recommended bare root technique has a number of advantages over other techniques including relatively low donor bed impact and ease of transporting large amounts of transplant material in relatively small volumes.

Studies of shoot addition and areal coverage rates have enabled development of transplant spacing guidelines which can vary in accordance with the management objectives of a site. Guidelines have been developed for Zostera marina, Syringodium tilitorme, and Halodule wrightii. As a result of relatively slow asexual reproductive rates of Thalassia testudinum, transplantation of pure stands of this species is discouraged.

Present research is directed toward improving plant survival through the critical first 60-90 days of a transplant and increasing subsequent shoot addition rates by the application of slow-release fertilizers.

INTRODUCTION

Involvement of the U.S. Army Corps of Engineers in seagrass transplanting began in the early 1970's and has continued to the present, including work sponsored by the Waterways Experiment Station (WES), the Coastal Engineering Research Center (CERC) and the U.S. Army Engineer District (USAED), Mobile. Research has involved approximately a dozen principal investigators working on all three coasts of the United States.

Corps interest in transplanting research has centered around sediment stabilization and habitat enhancement, development, and mitigation. Sediment stabilization applications include erosion protection for low-energy shorelines and stabilization of dredged material deposits and subtidal bottoms near channel cuts. Research has emphasized development

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of a reliable, cost-effective transplanting method and criteria to choose transplant locations and determine success.

HISTORY

The earliest Corps-sponsored work was that of Eleuterius (1974, 1975) and Eleuterius and McClellan (1976) for the USAED, Mobile, in which the primary interest was sediment stabilization, especially of shallow-water dredged material disposal mounds. Eleuterius attempted transplanting Halodule wrightii, Thalassia testudinum, and Syringodium filiforme, reporting good to poor establishment success in respective order. Both sod and bare root techniques were attempted, with bare root planting being preferred due to lower labor cost and greater success. Eleuterius also tried several different anchoring methods using both hand planting and broadcast techniques. The importance of selecting sites with relatively stable sediment profiles was shown to be important since many transplants died as a result of erosion or deposition.

The WES became involved in seagrass transplanting research with the advent of the Dredged Material Research Program (DMRP). The only field research conducted under this program was a transplant at Port St. Joe, Florida (Phillips et al., 1978). This study showed good transplant growth and survival during the early phases, but failure was attributed to extremely abnormal weather conditions (freezing and hurricanes). In addition to this work, Boone and Hoeppel (1976) conducted a transplanting feasibility study for San Diego Bay, California.

More recently, seagrass transplanting research has been sponsored by both the Coastal Ecology Branch, CERC, Fort Belvoir, Virginia, and the Coastal Ecology Group, WES, through the Environmental Impact Research Program (EIRP). Initial results of this effort were an annotated literature review (Knight et al., 1980) and publication of planting guidelines (Phillips, 1980). Beginning in 1981, the Corps and the National Marine Fisheries Service, Southeast Fisheries Center, Beaufort Laboratory, have cooperated in an interagency agreement to develop a more unified Federal approach to seagrass restoration and system management. Research has focused on improvements to the transplant methodology and development of population growth models for the various species (Fonseca et al., 1982a, 1982b, 1984, 1985). Primary objectives have been to develop performance standards for transplant sites and objective methods to evaluate success or failure. This information would allow Corps District offices to determine compliance in seagrass mitigation projects.

Knowledge of seagrass population growth also permits computation of spacing guidlines for planning purposes. With spacing guidelines one can determine how much habitat can be developed depending on the amount of time that can be allowed for meadow development. For example, if one can prepare enough planting units to cover one acre, allowing a year growth period, then the same number could be used to plant greater acreage if the meadow development time were allowed to be longer (Fonseca et al., 1982a, 1984, 1985).

CURRENT RESEARCH

Presently our research is directed toward determining the effects and efficacy of slow-release fertilizer application on seagrass transplants. Resarch by Orth (1977), Orth and Moore (1982), and Roberts et al. (1984) suggests that such fertilizer application may significantly increase asexual reproduction rates of *Zostera marina*. Our interest in fertilizer application is to reduce or eliminate an apparent critical period that occurs during the first 60-90 days of a transplant when the majority of transplant loss and growth lag occurs (Fig. 1). The advantages of faster growth are a need for less donor material, reduced labor, and faster meadow establishment.

Experimental design has employed control plots versus treatment plots containing either 10, 90, or 170 g of 14-14-14 or 18-0-0 (nitrogen-phosphorus-potassium) slow-release fertilizer. Experiments using *Zostera marina* or *Halodule wrightii* are in progress in Beaufort, North Carolina, Perdido Bay, Florida, and San Francisco, California. Preliminary work is under way at Key West, Florida, with *Thalassia testudinum*.

SUITABILITY OF A PLANTING SITE

One of the first needs for a successful seagrass transplant is selection of a suitable site. To the nonspecialist, it may not be intuitively obvious that not all shallow-water sites barren of seagrass will suffice. On the other hand, people familiar with a need to have some type of proper environmental conditions might ask, "If seagrass does not currently exist at the site, what makes you believe it can be successfully established?" This valid question demands an answer based on thorough investigations of the physical attributes of a site.

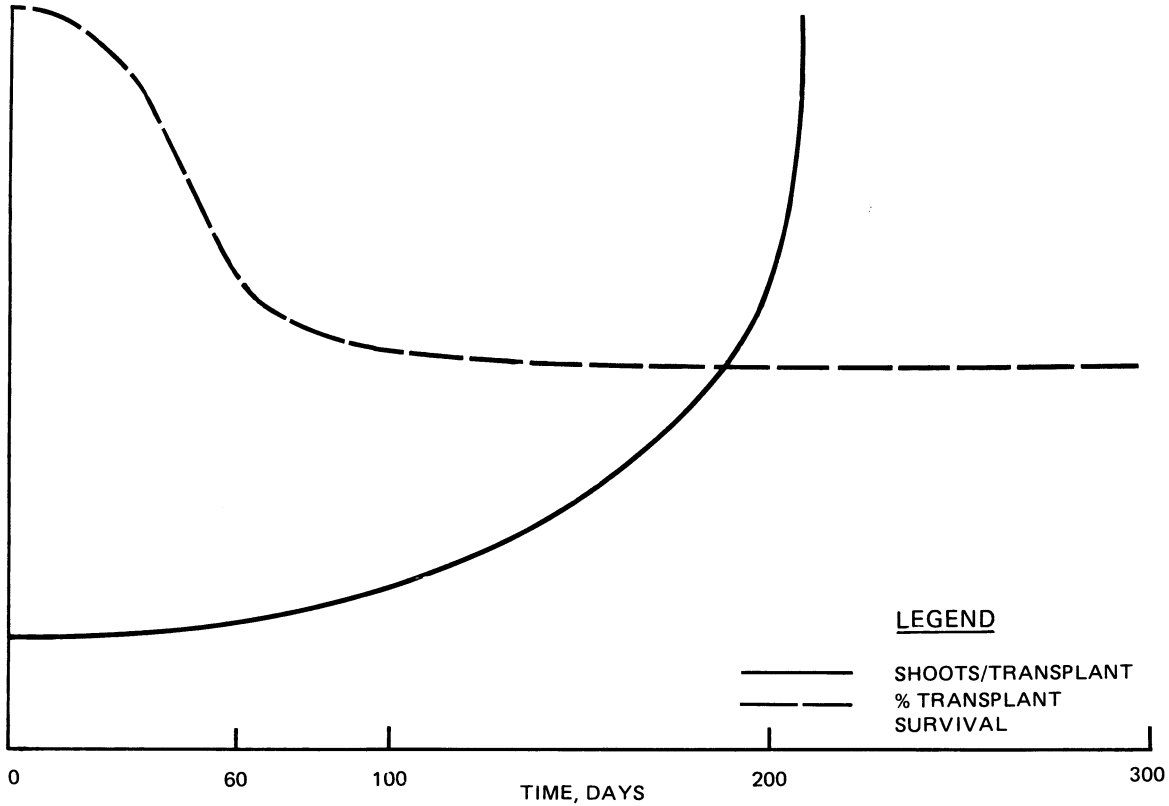
If, after examination, the site appears suitable, the question that still remains is "Why does it not support seagrass?" One answer might be that the site historically had supported seagrass, but an environmental (i.e., hurricane, extreme temperatures) or human-induced (i.e., pollution, hydraulic changes) disturbance resulted in loss (Fonseca et al., 1979).

At sites with no historical record of seagrass, absence could possibly be due to poor recruitment. Recruitment abilities of seagrasses are not well studied. For many seagrass species, seed production is unpredictable, and seed germination is low (Phillips, 1967, McMillan, 1976, Thorhaug & Austin, 1976, Kenworthy et al., 1980, Thayer et al., 1984).

RECOMMENDED ENVIRONMENTAL MEASUREMENTS

To evaluate potential planting sites, we recommend collection of several data sets. We feel the most important variables to measure are (1) light/depth penetration, (2) salinity, (3) temperature, (4) sediment flux rate, (5) currents, (6) fetch/waves, and (7) sediment depth (Fonseca et al., 1985). Data at potential sites should be collected as continuously as possible for several months. A weekly or biweekly

Figure 1. Generalized progress of seagrass transplants, depicting a growth lag and critical planting unit loss period during the first 60-90 days. Based on experiences with experimental plots of _Zostera_, _Halodule_, _Syringodium_, and _Thalassia_.



sampling schedule may be sufficient to determine seasonal cycles and variability. Similar measurements in nearby extant seagrass beds will provide a basis for comparison with the potential site. In addition, literature values from local surveys may be used, but cautiously, since regional variation could be misleading.

Depth and light penetration covary but both should be measured. Seagrasses are limited at deep sites, due to insufficient light to maintain the plant's carbon balance (Dennison & Alberte in press), and at shallow sites due to exposure and dessication. Similarly, salinity and temperature extremes of a site are critical variables that will affect plant growth, survival, and reproduction (Phillips, 1980).

Sediment flux rate is a variable that describes the dynamics of a site in terms of both erosion and deposition events. If these events are of sufficient magnitude or freuqency there is a low likelihood of transplant success (Fonseca et al., 1985).

Currents and waves are the causative factors of sediment flux though precise correlation of sediment movement to these factors is difficult. High-current sites should be considered cautiously since successful transplants are possible, but the costs and risks will be higher (Fonseca et al., 1985). Currents will need to be measured directly, while wave conditions can be estimated based on fetch except in areas of heavy boat traffic.

Sediment depth applies in situations where the sediment may be underlain by bedrock. This is especially important in subtropical or tropical habitats where species such as Thalassia testudinum require relatively thick sediment deposits for successful growth (Scoffin, 1970, Zieman, 1972).

PLANTING METHOD

The planting method we suggest is a bare root technique using 5 to 15 shoots attached to anchors (Fig. 2) (Fonseca et al., 1982a, 1984). Admittedly there are other methods that can be used with success (i.e., sod or plug transplants), but we believe the advantages of the bare root technique outweigh those of others in most circumstances (Table 1). Two major advantages of the bare root technique are relatively low donor bed impact and much smaller volumes and weight of transplant material to transport. These advantages are especially important in large-scale projects (as opposed to experimental research) where a large number of plants are needed. In donor beds where adventitious runners are present (Fonseca et al., 1984) harvest impact is negligible because no sediment disturbance occurs. People who have experienced hauling and handling buckets of sediment should appreciate the relative ease of transplanting hundreds of bare root transplants.

The two major disadvantages of this technique involve the fabrication of planting units. In addition to root damage sustained during harvest, plants may be injured while sorting, handling, and attaching plants to anchors. However, severity of such damage can be minimized

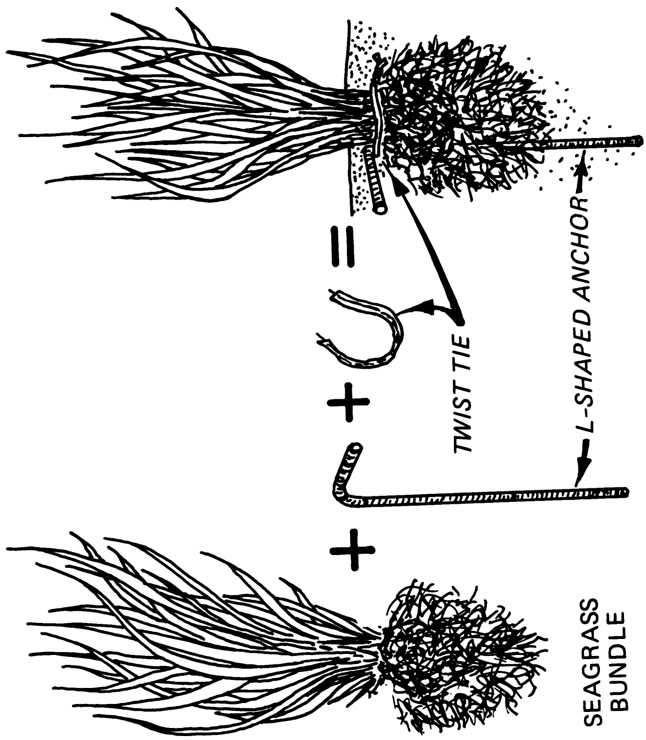


Figure 2. Bare root planting unit and anchor (redrawn from Fonseca et al. 1985).

Table 1. Comparison of planting methods. H = high, M = moderate, L = low, 0 = none. Ratings are subjective.

	ROOT DISTURB-ANCE	PLANT HANDLING	DAMAGE TO DONOR BED	VOLUME TO TRANSPORT	LABOR	STOCK AVAILA-BILITY
SOD/PLUG W/SEDIMENT	L	L	H	H	H	H
NURSERY SEED/SEEDLING	0-L	L	0	L	L-M	VARIABLE
BARE ROOT	H	M	0-L	L	L	H
FIBER MESH	H	H	0-L	L	H	H

through gentle handling. The labor needed to prepare planting units is a disadvantage we have chosen to accept since it allows less donor bed impact and movement of heavy, bulky sods.

The fiber mesh technique is a modified bare root method in which plants are woven into biodegradable mesh. We do not recommend use of this method since it involves much greater plant handling and labor costs (Fonseca et al., 1979). Seed and seedling techniques have some promise (Thorhaug, 1985), but this varies tremendously among species. Currently, seedling techniques are not very feasible due to underdeveloped culture techniques and uncertainty of stock supplies (Durako & Moffler, 1985).

SPECIES SELECTION

In considering a seagrass transplant, a choice needs to be made regarding species to transplant. Frequently, this is a moot point due to biogeographic limitations or salinity, though, in areas where more than one species is available, the relative advantages of each must be considered.

Thalassia testudinum, *Halodule wrightii*, and *Syringodium filiforme* are all present along the Gulf of Mexico and south Florida coasts and Caribbean. Each has characteristic environmental tolerances, though all three are frequently found in mixed beds or in close proximity (Zieman, 1982) and are all likely to be options in species selection. A major difference in these species, as shown by our research, is population growth rates of transplanted stock. *Halodule* is capable of very rapid vegetative reproduction (Fig. 3), *Thalassia* grows very slowly, while *Syringodium* is intermediate.

When giving consideration to such results in planning a mitigation project where a *Thalassia* meadow is being impacted, we must keep in mind the management objectives. Based on our projection, *Thalassia* meadows can be replaced only at very great expense in terms of either time or money. Since *Thalassia* slowly reproduces vegetatively, transplanting efforts must employ high intitial shoot densitites to attain natural densitites within a few (two to four) years. Without effective culture and nursery techniques, this translates to a large donor bed impact. However, one could transplant a site with *Thalassia* using low initial shoot densities and wait several (eight to ten) years until density reaches that of natural beds, though the primary objectives of habitat function restoration and sediment stabilization also would be delayed. In addition, the risks of transplant failure over this protracted period of time before coalescence would be greater.

In light of this, we recommend that a multispecies transplant is the most pragmatic and effective approach in the Gulf area (Mangrove Systems, Inc., 1985). Such an approach would involve planting either two or all three species in combination or *Halodule* and *Syringodium* followed at a later date by *Thalassia*. This program would best meet our objectives as the relatively faster reproducing *Halodule* and *Syringodium* would stabilize sediments and restore ecosystem function

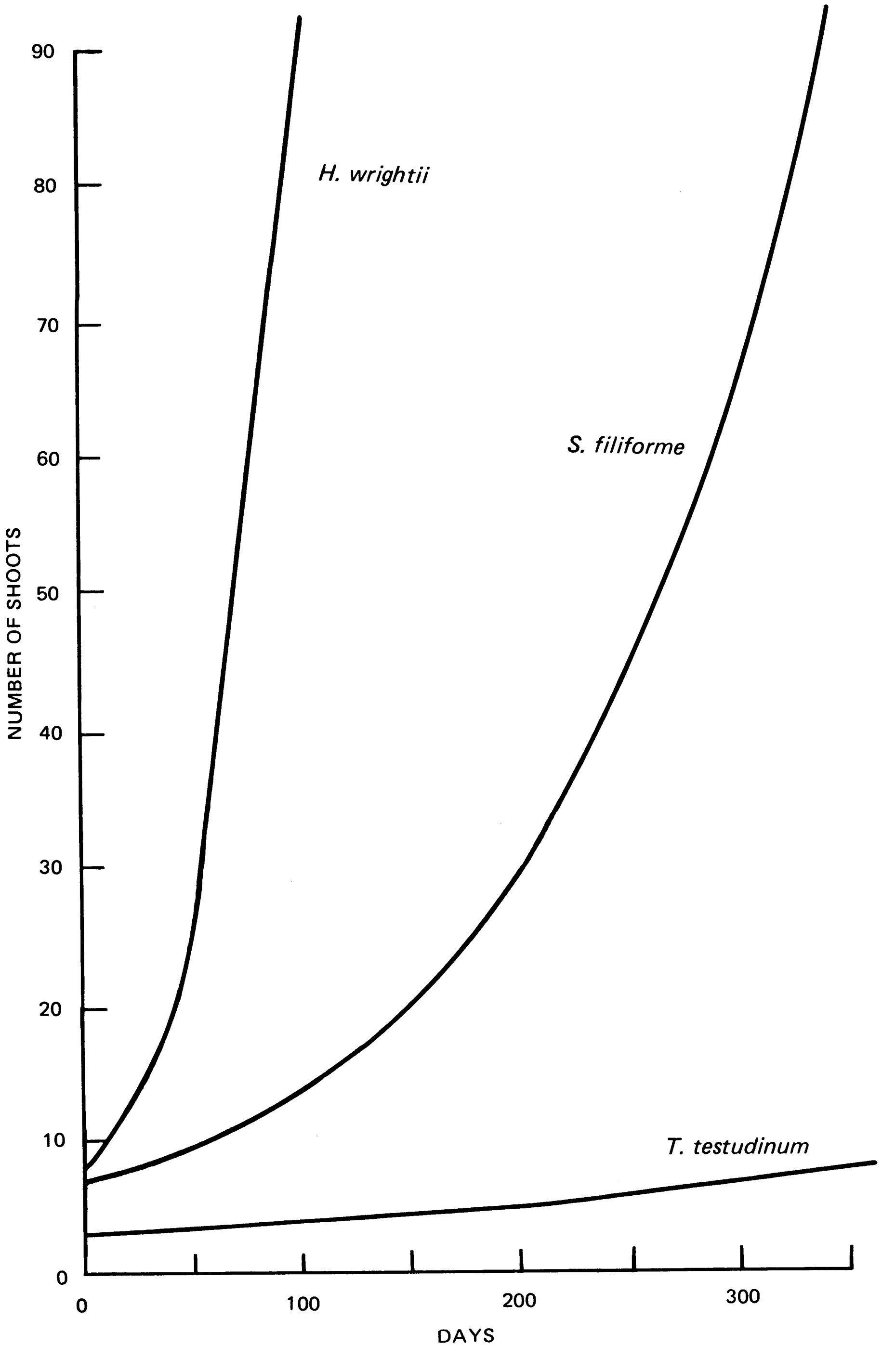


Figure 3. Average growth rate of transplants in south Florida.

and in subsequent years Thalassia, in situations where it has competitive dominance, would increase in relative importance.

SUMMARY

During the last decade there has been a marked increase of research into all aspects of seagrass biology (Fig. 4) with transplanting research paralleling the trend. Compared with ten years ago, we now know more about site selection, plant growth, and the pitfalls of transplant techniques, but there are still many uncertainties.

We currently have very little long-term data on transplanted seagrass beds, as most of the major projects (this cooperative work, Biscayne Bay, Florida Keys bridge replacement, Chesapeake Bay) only began in the last few years. There is a great need to monitor these sites and to document reasons for failure or success (and to state clearly how failure or success is measured) so that we may modify future decisions. Seagrass meadows are unique, highly productive environments that are not easily replaced and deserve protection (Thayer et al., 1984). However, as a result of constant pressure to develop coastal areas, seagrass beds are likely to receive impacts. It is in the best interest of all concerned that we continue to obtain better information on seagrass population growth and coverage so that we are equipped to provide objective management guidance for the preservation and, where necessary, restoration of seagrass beds.

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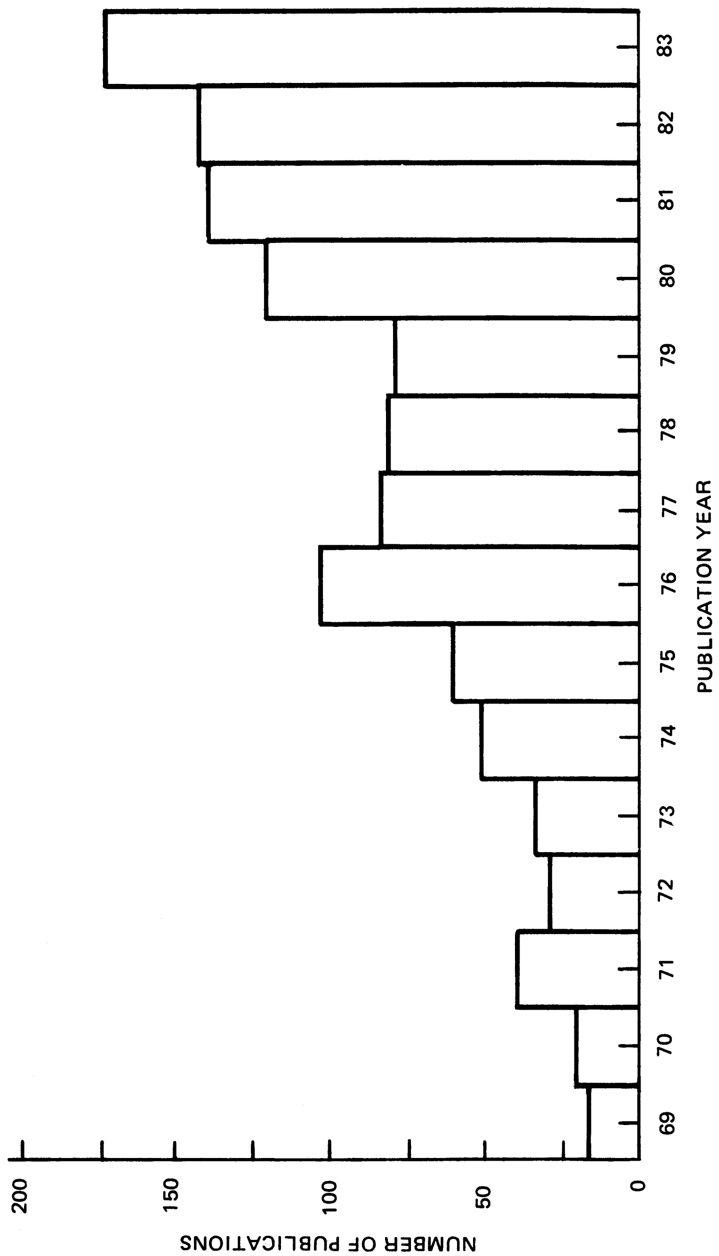

Figure 4. Seagrass publication trend. (Based on computer search of Biosis. Previews using several seagrass-specific keywords)

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ENVIRONMENTAL DESIGN GUIDELINES FOR MAN-MADE LAKES IN FLORIDA

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ABSTRACT

There are numerous opportunities for enhancing the environmental, aesthetic, and water quality values of man-made lakes within the state of Florida. Man-made lakes are created regularly for the purposes of stormwater management, providing fill material, or mitigating the effects of altered natural habitats.

Man-made lakes can provide improved water quality and habitat for fish and wildlife if the following guidelines are utilized in the design and construction of the lakes: (1) maximum depths between 10 and 20 feet, (2) side slopes of 6:1 horizontal to vertical out at least two feet below the control elevation, (3) shoreline development index greater than 1.2, (4) placement of two to three inches of hydric soils around the perimeter of the lake, and (5) shoreline planting consisting of native trees, shrubs, and aquatic species.

INTRODUCTION

There are growing opportunties for improving the environmental values associated with Florida's man-made impoundments. These opportunities exist within the urban and suburban developments where numerous man-made lakes have been or will be built for stormwater management, recreation, excavation, or mitigating altered natural habitat. Past and present mining activities have left extensive areas pocketed with small to medium size lakes and ponds in Florida. Man-made lakes represent a potential natural resource if they are designed from the point of view of habitat and water quality enhancement. The purpose of this paper is to provide guidance in the design of man-made lakes. This paper outlines guidelines for shoreline depths, shoreline vegetation establishment, and water quality enhancement. Brown, et al. (1984 & 1983), Gogly and Callahan (1983), and Posey et al. (1984) present excellent papers on the success of various shoreline and revegetation techniques. Many of these methods are refined here.

PHYSICAL PARAMETERS

Several physical factors are important in the study and design of man-made lakes. These factors include maximum depth, mean depth, surface area, volume, shoreline development index, and mean side slopes, are all important in determining the biological productivity and ultimate water quality of the lake. Maximum depth is important in determining the productive volume of the lake basin or that portion of water in which biological production occurs (Welch, 1952). Since the potential for statification creating an oxygen-poor hypolimnion increases with increasing depth, attempts should be made to bring depth to a point where optimum circulation and biological production can occur. Too much biological production, an end result of rapid eutrophication, can be minimized by depth regulation of the hypolimnetic zone, which is a factor in the regulation of dissolved oxygen and organic matter decomposition from the epilimnetic photic zone (Welch, 1952).

Shoreline development (SD) is the ratio of the area of the lake to a lake of equal area, but in the shape of a circle. The index, SD, is calculated from the formula:

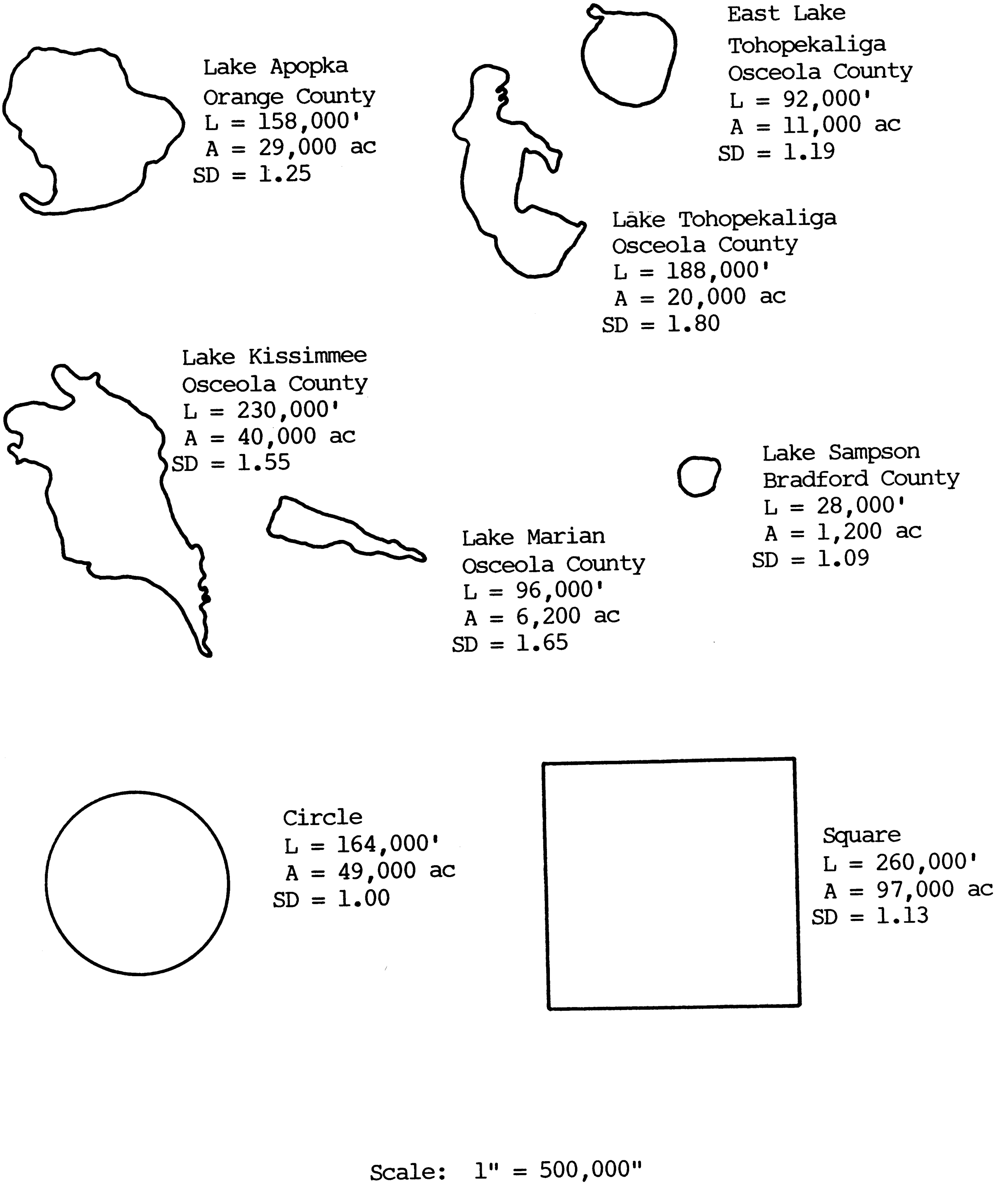
$$SD = \frac{SL}{2\ (\pi A)^{\frac{1}{2}}}$$

in which SL is the length of the shoreline and A is the area of the lake (Reid & Wood, 1976). With increasing index values for SD (greater than 1.0) there is a greater ratio of length to survace area. SD values for various lakes in Florida compared to SD values for a circle and square at a scale of 1 to 500,000 are provided on page 137.

The shoreline gradient determines the extent of littoral productivity and is quite important to overall lake water quality and ecosystem stability. Gravity has a pronounced effect on steep slopes by influencing the amount of sedimentation from decomposing organic matter to lower elevations. The movement of allochthonous matter (those sediments introduced from outside the lake) to the deep portions of the basin can result in decomposition products becoming inaccessible to plants and animals in the shallower littoral zone. One the other hand, a very gentle slope decreases erosion, thus keeping in place littoral produced organic matter, to finally establish the shoreline as a habitat for beneficial plants and animals. The littoral zone is by far the most productive and diverse habitat in an aquatic ecosystem.

DESIGN AND CONSTRUCTION RECOMMENDATIONS

Most man-made lakes are constructed for a single purpose, such as stormwater retention, excavation activities, recreational or aesthetic value. Rarely are man-made lakes initially designed for multipurpose featers, i.e., water storage capabilities, aesthetic and habitat value, and water quality benefits. Water quality and habitat functions are usually secondary to permit approval. Because of the potential for significant environmental benefits from man-made lakes, the initial design should be based on obtaining the maximum mulipupose value at the least cost to the development. The following criteria are suggested for designing a lake system. These specific guidelines should be interpreted in the context that environmental conditions are site specific and require a great deal of flexibility in design depending on specific development constraints.


Lake Apopka
Orange County
L = 158,000'
A = 29,000 ac
SD = 1.25
East Lake
Tohopekaliga
Osceola County
L = 92,000'
A = 11,000 ac
SD = 1.19
Lake Tohopekaliga
Osceola County
L = 188,000'
A = 20,000 ac
SD = 1.80
Lake Kissimmee
Osceola County
L = 230,000'
A = 40,000 ac
SD = 1.55
Lake Sampson
Bradford County
L = 28,000'
A = 1,200 ac
SD = 1.09
Lake Marian
Osceola County
L = 96,000'
A = 6,200 ac
SD = 1.65
Circle
L = 164,000'
A = 49,000 ac
SD = 1.00
Square
L = 260,000'
A = 97,000 ac
SD = 1.13
Scale: 1" = 500,000"

Side Slopes

Various state and local regulations specify what side slopes gradients are appropriate for man-made lakes. These regulations have been established to reduce safety hazards associated with shoreline activity and to ensure that adequate vegetation zones are established around the perimeter of the lake. Florida Administrative Code, Chapter 17-25, requires at least a 4:1 horizontal to vertical slope, out to at least two feet below the control elevation before fencing is required. Similarly, Chapter 16C-16, Florida Administrative Code, requires at least a 4:1 horizontal to vertical slop out 25 feet or more from the lowest anticipated water level before fencing is required. If a 4:1 slope is impractical, then a 10 foot bench, one foot below the water line, is required.

I recommend at least a 6:1 horizontal to vertical slope out two feet or more below the control elevation. This will increase the establishment of native shoreline vegetation while providing a more than adequate margin of safety. The littoral zone created by 6:1 slopes will provide increased nutrient assimilation capabilities and shoreline protection. It is also recommended that the 6:1 slope extend from the water surface up to approximately one foot above the control elevation (eulittoral) to provide a six foot wide zone above the water line for aquatic emergents and native shrubs and trees. A shallow, grassy swale and berm is also recommended to intercept surface water runoff prior to discharge into the lake. The swale should be free-draining within four days to prevent mosquito production. A typical section of lakeshore is depicted in Figure 1.

Maximum Depth

The maximum depth should strike a balance between excessive vegetation production within shallow areas of artificial lakes and the regulation of thermal and chemical stratifciation in the lower depths. Because shallow areas receive more light, the rate of photsynthesis is enhanced. Likewise, the construction of extremely deep lakes can result in low dissolved oxygen concentrations and reduced plant production. Generally, the balance in vegetative production between a shallow and a deep system in Florida is reached at maximum depths ranging from 10 to 20 feet. The optimum depth will depend on site specific factors such as location constraints, groundwater discharge, and the soils and geologic nature underlying strata.

Shoreline Development

In order to derive the full benefits of habitat and water quality enhancement, efforts should be made to design a lake with a shoreline development index greater than 1.2. This will increase the littoral shelf area in relation to the total surface of the lake, providing a high degree of nutrient assimilation capacity, shoreline stabilization, and habitat value.

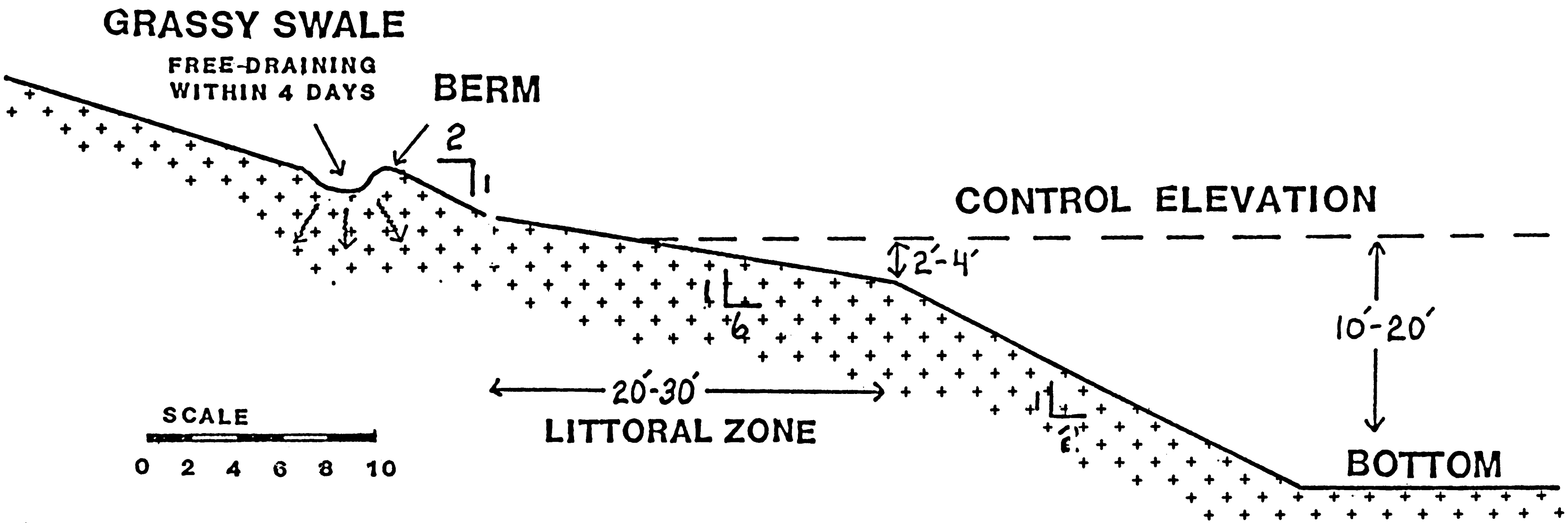


FIGURE 1. TYPICAL CROSS SECTION OF LAKESHORE.

Revegetation Guidelines

The first step, after contouring, grading, and sloping, is to stabilize the highly erodible littoral slope and provide a bedding for native plants. One method which has proven successful in phosphate surface mining environments is by mulching with peat or organic soils. Brown et al., 1984, found that germination and growth of wetland and transitional species on sites where hydric soil was applied was far greater than on sites without hydric soils. We have found this to be the case in our phosphate reclamation work with CF Industries in Hardee County. These soils can usually be located on site, stockpiled for short periods of time, then spread 2-3' deep and +/-15' wide, to provide an organic soil base, seeds and propagules for the growth of aquatic plants, trees, and shrubs. Stockpiled soils should be used as soon as possible to assure the viability of existing plant propagules. If possible, direct application of the bedding soils from donor sites to lakeshores is preferred.

The types of native flora used to vegetate the shoreline of man-made lakes will depend on the geographic location of the site, landscape objectives, and plant availability. Many native plants rival non-native nursery stock in beauty, growth, and overall aesthetic qualities. Native plants are cold tolerant and can readily adapt to ecological conditions of a man-made lakeshore. Exotic species will need management to reduce their competition against native species. With proper planning and design, native trees, shrubs, and aquatic plants can provide a lakeshore environment which is aesthetically pleasing and at the same time provide water quality improvements and a diversity of habitat. Some of the native aquatic plants that have proven to be successful in the aquatic landscape are the fragrant water lily, southern blue flag, golden canna, bur marigold, climbing aster, pickerel weed, duck potato, coastal arrowhead, golden club, and others, Table 1. The appropriate vegetation planting locations of the littoral zone are depicted in Figure 2.

SUMMARY AND CONCLUSIONS

There are many opportunities in Florida where a multiuse approach can be utilized in the development of land. Man-made lakes and retention areas can be designed and built to accommodate not only stormwater volume requirements but also fish and wildlife, aesthetic, and water quality needs.

The following guidelines whould provide the maximum multiuse values of man-made lakes:

1. Maximum depths between 10 and 20 feet;

2. Side slopes of 6:1 horizontal to vertical out at least two feet below the control elevation;

3. Shoreline development index greater than 1.2;

Table 1. A selection of recommended plants for man-made lake shorelines.

Suitable Planting Zone	Common Name	Scientific Name
Trees:		
A/B	Cypress	Taxodium spp.
A/B	Gums	Nyssa spp.
B/C	Sweet bay	Magnolia virginicana
B/C	Popash	Fraxinus caroliana
B/C	Loblolly bay	Gordonia lasianthus
B/C/D	Cabbage palm	Sabal palmetto
B/C/D	Dahoon holly	Ilex cassine
B/C/D	Water oak	Quercus nigra
C/D	Sweet gum	Liquidambar styraciflua
C/D	Red maple	Acer rubrum
C/D	Pignut hickory	Carya glabra
C/D	Sugarberry	Celtis laevigata
D	Southern magnolia	Magnolia grandiflora
D	Laurel oak	Quercus laurifolia
Shrubs:		
C/D	Wax myrtle	Myrica cerifera
C/D	Shiny lyonia	Lyonia lucida
Herbaceous Cover:		
A	Giant bull rush	Scirpus californicus
A	Arrowheads	Sagittaria spp.
A	Maidencane	Panicum hemitomom
A	Fragrant water lily	Nymphaea odorata
A	Pickerelweed	Pontederia cordata
A	Spatterdock	Nurphar lutem
A	Golden club	Orontium aquatium
A/B	Cord grass	Spartina spp.
A/B	Soft rush	Juncus effusus
A/B/C	Climbing aster	Aster carolinianus
B	White-top sedge	Dichromena spp.
B	Canna lilies	Canna indica
B	Blue flags	Iris spp.
B	Spider-lily	Hymenocallis sp.
B/C/D	Ferns	

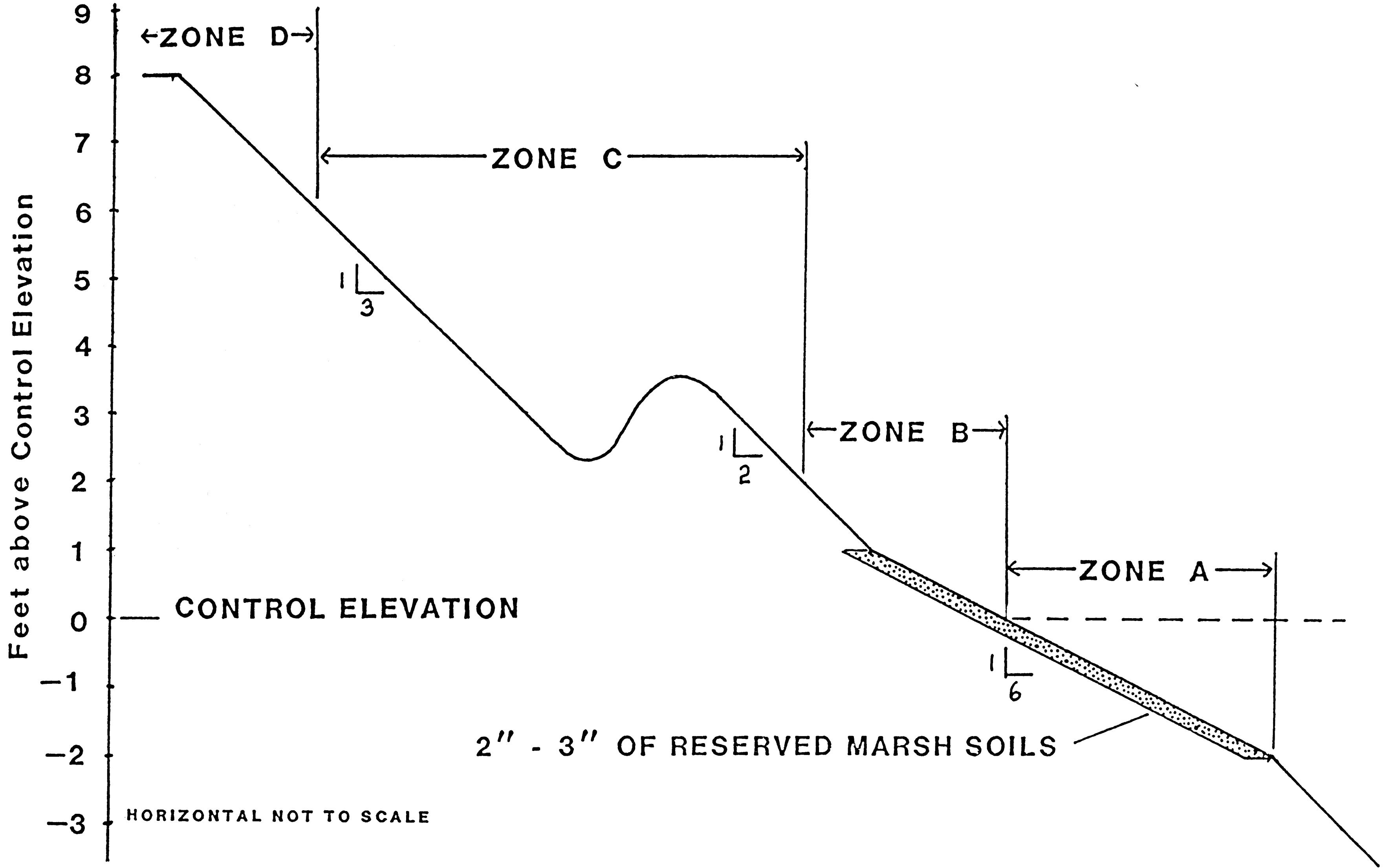


FIGURE 2. REVEGETATION ZONES OF TYPICAL LAKESHORE.

4. Placement of two to three inches of hydric soils around the perimeter of the lake; and

5. Shoreline planting consisting of native trees, shrubs, and aquatic species.

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RESULTS OF THE VEGETATIVE EROSION CONTROL PROJECT IN THE VIRGINIA CHESAPEAKE BAY SYSTEM

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ABSTRACT

Reestablishing a marsh grass fringe for estuarine shoreline erosion control is an accepted alternative to construction of bulkheads, revetments and groins. The physical limit of creating a marsh grass fringe is mainly the severity of wave climate acting on a given shore. The main variables used to determine the relative intensity of wave climate are (1) average fetch exposure, (2) shore geometry, and (3) shore orientation.

In the Virginia Chesapeake Bay system 24 planted marsh grass fringes were monitored from 1981 to 1983. These sites were selected to include a variety of average fetch exposures. Results of the planting project showed that (1) establishing a marsh grass fringe can be accomplished with little or no maintenance planting on relatively low wave energy shores (average fetch exposure less than 1.0 nautical mile). (2) Along medium wave energy shorelines exposed to 1.0 to 3.5 nautical miles average fetch, the establishment of a combination fringe of smooth cordgrass and saltmeadow hay is necessary. (3) On straight shorelines with average fetch exposures of 3.5 to 5.5 nautical miles it will be impractical to try and establish a marsh fringe without some type of permanent offshore wave stilling device (i.e., a breakwater). (4) Shorelines exposed to an average fetch greater than 5.5 nautical miles should not be considered for marsh grass implantation unless well protected by a headland, island, or spit. The use of offshore breakwaters in combination with marsh implantation is a consideration but further research is needed.

INTRODUCTION

Research on the planting of salt marsh grasses for shoreline erosion control has been conducted for the past 30 years with varying degrees of success. Some early plantings in the United States were done in the estuaries of Virginia (Phillips & Eastman, 1959; Sharp & Vaden, 1970). A number of these plantings remain established today. In the 1970's, research at North Carolina State University sponsored by the U.S. Army Corps of Engineers, Coastal Engineering Research Center produced data on planting methods for shoreline protection and dredge spoil stabilization

(Woodhouse, Seneca & Broome, 1972, 1974, 1976; Knutson & Woodhouse, 1983). Additional studies were done in the Chesapeake Bay by Garbich et al. (1973) and Garbich, Woller and McCallum (1975). In 1981 Knutson performed a national survey of 86 planted salt marshes. Most research was focused on the intertidal zone and smooth cordgrass (Spartina alterniflora).

It was recognized that the principal factor affecting the initial establishment and the long term durability of planted salt marshes is wave stress or wave climate. The only definitive work on wave damping in Spartina alterniflora narshes was done by Knutson in 1982. According to Knutson, on the average, more than 50 percent of the energy associated with waves passing through a fringe marsh were dissipated within the first eight feet (2.5 m) of the marsh. Thus, the wider the marsh, the more wave reduction. Marshes are most effective in damping waves when water depth is less than plant height. As wave energy impacting a shoreline is reduced, there is increased potential for sediment deposition and decreased potential for erosion (Knutson, 1982).

The average fetch is a fairly accurate indicator of potential wave stress that a given shore is exposed to. Previous research suggests that attempting to establish a marsh fringe on a shore exposed to greater than five nautical miles average fetch is extremely difficult. Successful marsh planting on shores exposed to less than one nautical mile average fetch is highly probable as long as there is sufficient sunlight and limited boat wake activity. The one to five nautical mile average fetch exposure is an area where marsh planting has a chance of partial success. According to Knutson (1981) other factors such as shoreline geometry (straight, headland or cove), longest fetch and beach sand grain size will play important roles in determining potential areas for marsh grass establishment.

The Vegetative Erosion Control (VEC) Project was a four-year study with coordinated efforts by the Virginia Soil and Water Conservation Commission (VS&WCC) through their Shoreline Erosion Advisory Service (SEAS), the Soil Conservation Service (SCS) and the Virginia Institute of Marine Science (VIMS). The purpose of the VEC Project was to supplement existing data and research with detailed site analysis of the early stages of marsh development and to define more precisely the physical (wave stress) limitations on marsh implantation for shoreline erosion control without the aid of beach fill and/or sand retaining structures, especially on sites exposed to average fetches of one to five miles.

METHODOLOGY AND PROCEDURE

Previous research by Knutson and others indicates that marsh establishment along low wave energy shores such as creeks is much easier than on more exposed shores like those on the Chesapeake Bay. Categories of average fetch exposure for the Virginia Chesapeake Bay system were delineated for the VEC Project. Low energy shores are exposed to average fetch of less than 1.0 nautical mile, high energy shores greater than 5.0 nautical miles, and medium energy shores 1.0 to 5.0 nautical miles.

The 24 sites chosen for the VEC Project are located throughout the

Virginia portion of the Chesapeake Bay system (Fig. 1). They were selected not only to represent different average fetch exposures but also shoreline orientations and shoreline geometries (Fig. 2).

Shoreline orientation was originally felt to be an important factor because the north facing southern shores of the major rivers (James, York, Rappahannock & Potomac) in Virginia are historically eroding at a rate two to three times faster than the south facing northern shores (Byrne & Anderson, 1977). The southern shores are more exposed to northeast storm waves and the seasonal wind and wave climate.

Shoreline geometry will affect wave climate also. A straight shoreline may receive wind and wave action from several directions. A headland is generally very exposed and can be affected by winds blowing from many directions. Some sites are semi-protected by being situated in a cove, embayment, or behind and island, spit, or shoal. Semi-protected shores may receive wave action from only one or two directions.

In general, the seasonal winds are responsible for the wave climate throughout the Chesapeake Bay region. During the fall and winter northerly winds dominate and in the late spring and summer, southerly winds occur most frequently. Wind direction and speed measured over the past 30 to 40 years support these wind patterns (U.S. Department of Commerce).

The VEC sites were planted over a three year period during the spring of 1981, 1982, and 1983 (Table 1). Annual and/or bi-annual maintenance planting was done on several sites due to winter storm effects. Two species of marsh grass were planted all on 1.5-foot centers. They are smooth cordgrass (Spartina alterniflora) and saltmeadow hay (Spartina patens). Smooth cordgrass grows best between mean sea level (MSL) and mean high water (MHW). It receives the initial effects of impinging wave action. Naturally occurring smooth cordgrass grows between three to six feet high. Saltmeadow hay grows well above MHW and helps trap sand and elevate the backshore. Saltmeadow hay was planted along the backshore when there was six feet or more between MHW and the base of an eroding bank. Smooth cordgrass was planted between MHW and several feet below MSL.

Monitoring of the VEC sites was accomplished by semi-annual profiling with a Zeiss Ni2 level in the fall and spring of each project year. The profiles depicted the response of the intertidal beach, backshore and nearshore to the implanted marsh grass fringe through time. Also, annual mapping (plane table and alidade) of the marsh fringes was done at the end of each growing season. The changes in aerial extent were measured to reflect where the grasses had expanded or were eroded out.

In most cases, there is a stepwise progradation or sequence a marsh goes through from the initial planting through time. Knutson and Woodhouse (1983) recognize a "functional life" of a planted marsh in which the marshes proceed through a cycle over which the marsh has served to reduce erosion. The cycle begins with marsh establishment. Stability and, finally, erosion of the marsh constitute the second and third stages of the cycle. The life of a planting is influenced by the

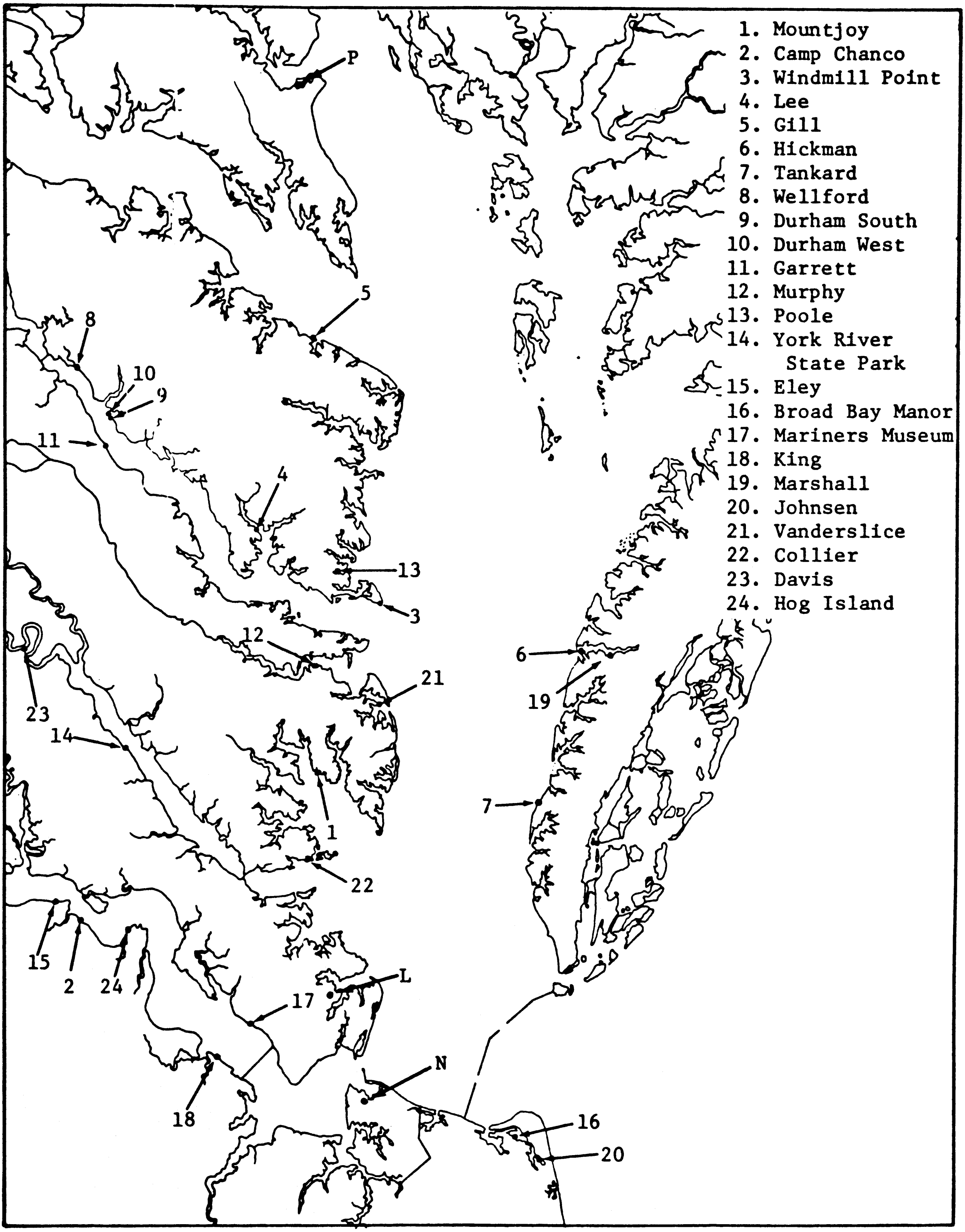

Figure 1. Planting site locations with Patuxent(P), Langley(L), and Norfolk(N) wind recording stations. Richmond is not shown.

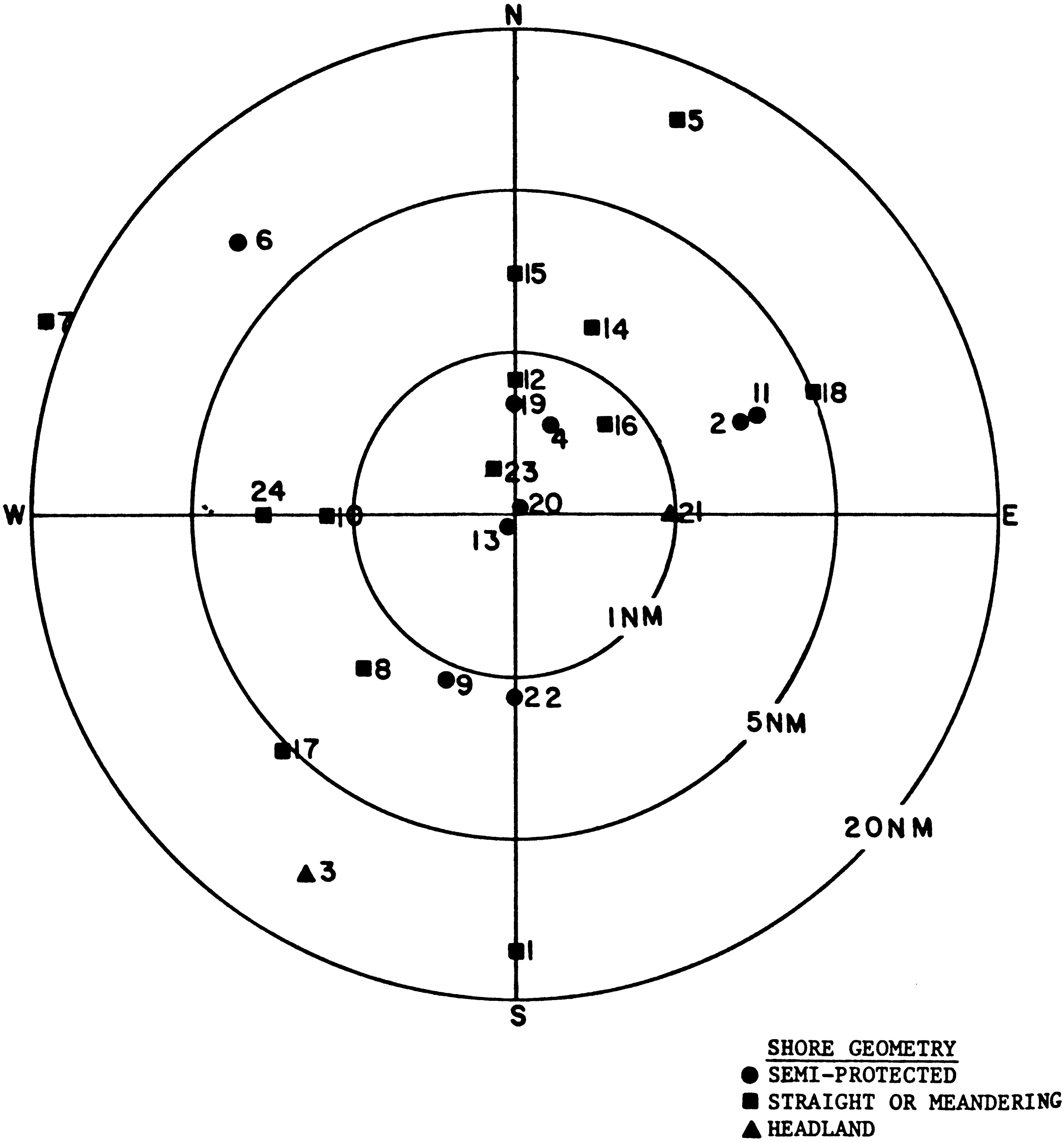

Figure 2. Planting Sites - Their average fetch, shore geometry, and orientation (the direction that the shore faces).

TABLE 1

PLANTING SCHEDULE OF VEC SITES

		Spring 1981		Spring 1982		Spring 1983	
		Date	Species	Date	Species	Date	Species
1.	Mountjoy	27 May	Sa	19 May	Sa	-	-
2.	Camp Chanco	26 May	Sa, Sp	-	-	-	-
3.	Windmill Point	27 May	Sa, Sp	18 May	Sa, Sp	-	-
4.	Lee	6 May	Sa, Sp	-	-	3 Jun	Sa
5.	Gill	6 May	Sa, Sp	-	-	-	-
6.	Hickman	29 May	Sa, Sp	-	-	-	-
7.	Tankard	29 May	Sa, Sp	-	-	-	-
8.	Wellford	5 May	Sa	-	-	2 Jun	Sa
9.	Durham South	5 May	Sa, Sp	21 May	Sa, Sp	-	-
10.	Durham West	5 May	Sa, Sp	21 May	Sa	-	-
11.	Garrett	6 May	Sa, Sp	18 May	Sa, Sp	2 Jun	Sa, Sp
12.	Murphy	27 May	Sa, Sp	19 May	Sa	2 Jun	Sa, Sp
13.	Poole	-	-	18 May	Sa	3 Jun	Sa
14.	York River State Park	-	-	2 Jun	Sa	7 Jun	Sa

TABLE 1
Cont'd.

		Spring 1981		Spring 1982		Spring 1983	
		Date	Species	Date	Species	Date	Species
15.	Eley	-	-	3 Jun	Sa	15 Jun	Sa
16.	Broad Bay Manor	-	-	15 Jun	Sa, Sp	8 Jun	Sa
17.	Mariners Museum	-	-	3 Jun	Sa, Sp	8 Jun	Sa
18.	King	-	-	3 Jun	Sa, Sp	7 Jun 8 Jul	Sa, Sp
19.	Marshall	-	-	15 Jun	Sa, Sp	-	-
20.	Johnsen	-	-	15 Jun	Sa	8 Jun	Sa
21.	Vanderslice	-	-	19 May	Sa	22 Jun	Sa
22.	Collier	-	-	3 Jun	Sa	7 Jun	Sa
23.	Davis	-	-	2 Jun	Sa	-	-
24.	Hog Island	-	-	-	-	15 Jun	Sa

Sa = *Spartina alterniflora*

Sp = *Spartina patens*

severity of wave conditions which impinge upon the shore. Marshes subject to more severe wave conditions require more time to become established and have a shorter functional life (Knutson & Woodhouse, 1983).

Of the 86 sites Knutson used in his evaluation of planted marshes, 67 were over three years old. In contrast, only 4 of the 24 marshes established during this study were three years old. These are Camp Chanco (2), Lee (4), Hickman (6), and Wellford (8). There has also been a great deal of annual maintenance planting done to establish a fringe on the VEC sites. Hence, most of the sites are two years old with maintenance.

SITE ANALYSIS

Two factors were considered for a planted marsh fringe to be successful:

1. The marsh fringe should be continuous in length and width in order to offer some degree of stabilization along the shore.

2. The elevation of the backshore should be stable or increasing due to the trapping of sediment. Thus, the frequency of direct wave attack on the fastland bank will be reduced and less bank erosion should follow.

The most critical parameter in establishing a successful marsh fringe is the wave climate at the site. This is best expressed by determining the average fetch. Table 2 lists the VEC sites in order of increasing average fetch (Column No. 1). The shoreline geometry (Column No. 2) and orientation (Column No. 3) are listed for each site as is the cumulative yearly erosion rate (Column No. 4) for the top and base of the bank over the monitoring period. The cumulative yearly erosion rate is the rate per year of bank erosion estimated from the net change since the beginning of the project at each site.

A numerical indexing system was devised in order to provide a basis for comparison of planting success among sites. The condition of the marsh fringe at each site as of spring, 1984, is given a number value. Each grass species is treated separately where appropriate. A continuous fringe, defined as having about 75 percent of the original planted length remaining, is assigned a value of 2. A patchy, discontinuous fringe with 25 to 75 percent of its original planted length remaining is assigned a value of 1. A fringe with less than 25 percent of the original planting length is given a value of 0. These values are listed in Column No. 5.

The backshore elevation in terms of a net gain, loss, or no change is also given a numerical value. These are listed in Column No. 6. A value of 2 was given a site if the backshore elevation (that area between spring high water and base of bank) had measurably increased since the initial planting. If there was a stabilized backshore, even with slight gains and losses over time, the site was given a value of 1. Cases with a significant loss in backshore elevation were given a value of 0.

TABLE 2

RESULTS OF THE VEGETATIVE EROSION CONTROL PROJECT AS OF JUNE 1984

		1	2	3	4		5		6	7	8
					Cumulative Erosion Rate (Ft/Yr)		Fringe Condition[2]				Score*
		Average Fetch (Nautical Miles)	Geometry[1]	Orientation	Top of Bank	Base of Bank	Sa	Sp	Backshore Elevation[3]	Columns 5 + 6	(% of Total Possible)
Low Energy	20. Johnsen	0.01	●	NE	0.0	0.0	2	-	1	3	75
	13. Poole	0.04	●	SSW	0.0	-0.1	2	-	1	3	75
	23. Davis	0.30	■	NNW	-0.1	-0.4	2	-	1	3	75
	4. Lee	0.60	●	NNE	-0.1	-0.4	2	2	2	6	100
	19. Marshall	0.70	●	N	-0.6	-1.9	2	2	2	6	100
	16. Broad Bay Manor	0.80	■	NE	0.0	+1.5	2	2	2	6	100
	12. Murphy	0.84	■	N	0.0	-1.5	2	1	2	5	83
Medium Energy	21. Vanderslice**	1.00	▲	E	-3.2	-3.6	1	-	1	2	50
	9. Durham South	1.46	●	SSW	+0.3	-	2	2	2	6	100
	22. Collier	1.50	●	S	-2.4	-2.4	2	-	1	3	75
	10. Durham West	1.60	■	W	-1.7	-2.4	2	-	0	2	50

[1] ● - Semi-Protected Shore
■ - Straight or Meandering Shore
▲ - Headland

[2] 0 - Little or No
1 - Discontinuous
2 - Continuous

[3] 0 - Loss
1 - Stable
2 - Gain

* Sites with Sa will have a score total of 4 possible.
Sites with Sp will have a score total of 6 possible.
** On a headland but semi-protected from the Bay by barrier islands.

Sa = *Spartina alterniflora*

Sp = *Spartina patens*

TABLE 2
Cont'd.

Energy		Site	1 Average Fetch (Nautical Miles)	2 Geometry[1]	3 Orientation	4 Cumulative Erosion Rate (Ft/Yr) Top of Bank	Base of Bank	5 Fringe Condition[2] Sa	Sp	6 Backshore Elevation[3]	7 Columns 5 + 6	8 Score* (% of Total Possible)
Medium Energy	14.	York River State Park	2.00	■	NNE	-1.5	-1.6	1	-	0	1	25
Medium Energy	8.	Wellford	2.30	■	SW	-0.2	-0.3	1	-	1	2	50
Medium Energy	15.	Eley	3.00	■	N	-1.1	-1.4	2	-	1	3	75
Medium Energy	2.	Camp Chanco	3.10	●	ENE	-0.4	-1.4	1	1	1	3	50
Medium Energy	24.	Hog Island	3.20	■	W	-3.3	-1.9	2	-	0	2	50
Medium Energy	11.	Garrett	3.50	●	ENE	-0.1	-0.8	2	1	1	4	66
High Energy	18.	King	5.10	■	ENE	-7.0	-7.3	1	1	2	4	66
High Energy	17.	Mariners Museum	5.70	■	SW	-4.2	-4.6	0	1	0	1	17
High Energy	6.	Hickman	11.00	●	NW	-5.7	-	2	-	0	2	50
High Energy	3.	Windmill Point	13.40	▲	SSW	-4.5	-	0	0	0	0	0
High Energy	5.	Gill	15.00	■	NNE	-3.6	-4.6	0	0	0	0	0
High Energy	1.	Mountjoy	15.60	■	S	-3.9	-3.9	0	-	0	0	0
High Energy	7.	Tankard	22.0	■	WNW	-16.5	-17.7	0	0	0	0	0

[1] ● - Semi-Protected Shore
■ - Straight or Meandering Shore
▲ - Headland

[2] 0 - Little or No
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2 - Continuous

[3] 0 - Loss
1 - Stable
2 - Gain

* Sites with Sa will have a score total of 4 possible.
Sites with Sp will have a score total of 6 possible.

** On a headland but semi-protected from the Bay by barrier islands.

Sa = *Spartina alterniflora*

Sp = *Spartina patens*

The numerical values given to the condition of the fringe marsh and backshore elevation at each site were then added to see the trend toward a successful or failing result (Column No. 7 in Table 2). Since experience has shown that it may take several years for complete marsh establishment, it may still be too early to see a dramatic effect on erosion rates for all sites. Therefore, erosion rate was not used in the numerical ranking system.

Sites with only smooth cordgrass remaining will have a total possible score of 4. A site with smooth cordgrass and saltmeadow hay will have a total possible score of 6. Column No. 8 in Table 2 is the percent of each site's value score out of the total possible.

For a site to be trending toward success, it must have a continuous marsh fringe in order to adeuqately protect an eroding bank. Thus, a marsh fringe condition score of 2 is necessary. The sites most trending toward success will have a continuous fringe and a stable or elevated backshore (if there is an adequate supply of sediment). Thus, a score of 75 percent and a fringe condition of 2 is needed to achieve these criteria. A site with a score of 25 percent or less and a fringe condition of 1 or less will be a failure. Partially successful sites will have a score between 25 and 75 percent with a fringe condition of 2. A score between 25 and 75 percent and a fringe condition of 1 or 0 indicates a trend toward failure.

RESULTS AND DISCUSSION

The success or failure of the marshes planted under this program have been evaluated in terms of their continuity and ability to hold and/or elevate the backshore. Our results show that the low energy sites are trending toward success (i.e., index scores of 75% or greater). The high energy sites generally failed (i.e., index scores of 25% or less). Some of the intermediate sites have marsh fringes trending toward success. They are the sites which are exposed to under 3.0 nautical miles average fetch. The "functional life" of VEC sites trending toward success has yet to be realized. The two and three year old marsh fringes which are successful are just becoming established and partially stabilized. Without maintenance planting, the marshes in the low energy regime would most likely have the longest "functional life." Of course, the frequency of strong wind and storm events will greatly affect how quickly a marsh can be established. The first season is the critical period.

SHORE CLASSIFICATION

One of the main objectives of the VEC project was to better determine the physical limits in which a planted marsh could be established. As we have stressed, the physical limit of a site is indicated by the severity of the impinging wave climate which is largely a function of fetch and shore geometry. In the lower low energy regime (fetch less than 0.5 nautical miles) these factors are negligible. Exposure to sunlight and boat wakes appear to be more critical. Fetch and shore geometry become more important in the upper low energy regime and the medium energy regime.

Referring to Figure 3, approximately 60 percent of the stippled shoreline is low energy. Most of this has natural marsh fringes at present. The need to establish a marsh fringe will occur where the natural fringe has been eroded away or seriously reduced, so as to be an ineffective wave buffer.

Approximately 30 percent of the stippled shore in Figure 3 is high energy shore. Plant establishment on these shores is almost impossible by itself. However, shoreline geometry may provide protection at specific sites.

The remaining 10 percent of shoreline is within the medium energy regime. These shores mostly occur along the trunks of the rivers. It is conjectured that southward facing shores will have a better likelihood of establishment (i.e., less maintenance planting) than the north facing shores due to less exposure to northerly winds. Also, semi-protected shores and those in the lower medium energy regime (fetch-limited) will have a better survival rate. Annual maintenance planting will be the rule even after firm establishment.

A one time planting can be expected to suffice on a lower low energy shore with no boat wakes and good sunlight exposure. The degree of maintenance planting will increase as fetch increases. Annual planting may be necessary when the fetch is on the order of 1 nautical mile; it will be imperative when the fetch exceeds 3.0 to 3.5 nautical miles.

On this note, it would seem that research on the use of offshore wave stilling devices should be implemented. Temporary breakwaters could be installed "cheaply" to allow a planting to become firmly established. The structure could then be removed. Permanent, but more expensive structures might be appropriate for long stretches of eroding bank (e.g., farm land and timber land).

CONCLUSIONS

1. Establishing a marsh grass fringe can be accomplished with little or no maintenance planting on the low wave energy regime shores (average fetch less than 1.0 nautical mile). The majority of this type of shoreline will be along small, protected creeks. A combination marsh fringe of saltmeadow hay and smooth cordgrass should be implanted for best results.

2. Along medium wave energy shorelines exposed to 1.0 to 3.5 nautical miles average fetch, the establishment of a combination marsh fringe of saltmeadow hay and smooth cordgrass in necessary. Much of this shore exposure is along the major tributaries of the Chesapeake Bay. The saltmeadow hay will trap sand, dissipate wave action, and help elevate the backshore. The smooth cordgrass will also trap sand, dissipate wave action, and help protect the saltmeadow hay fringe. Semi-protected shorelines, especially coves and embayments, will have a better chance for marsh establishment than will straight or headland shores. Maintenance planting will be needed at varying intervals.

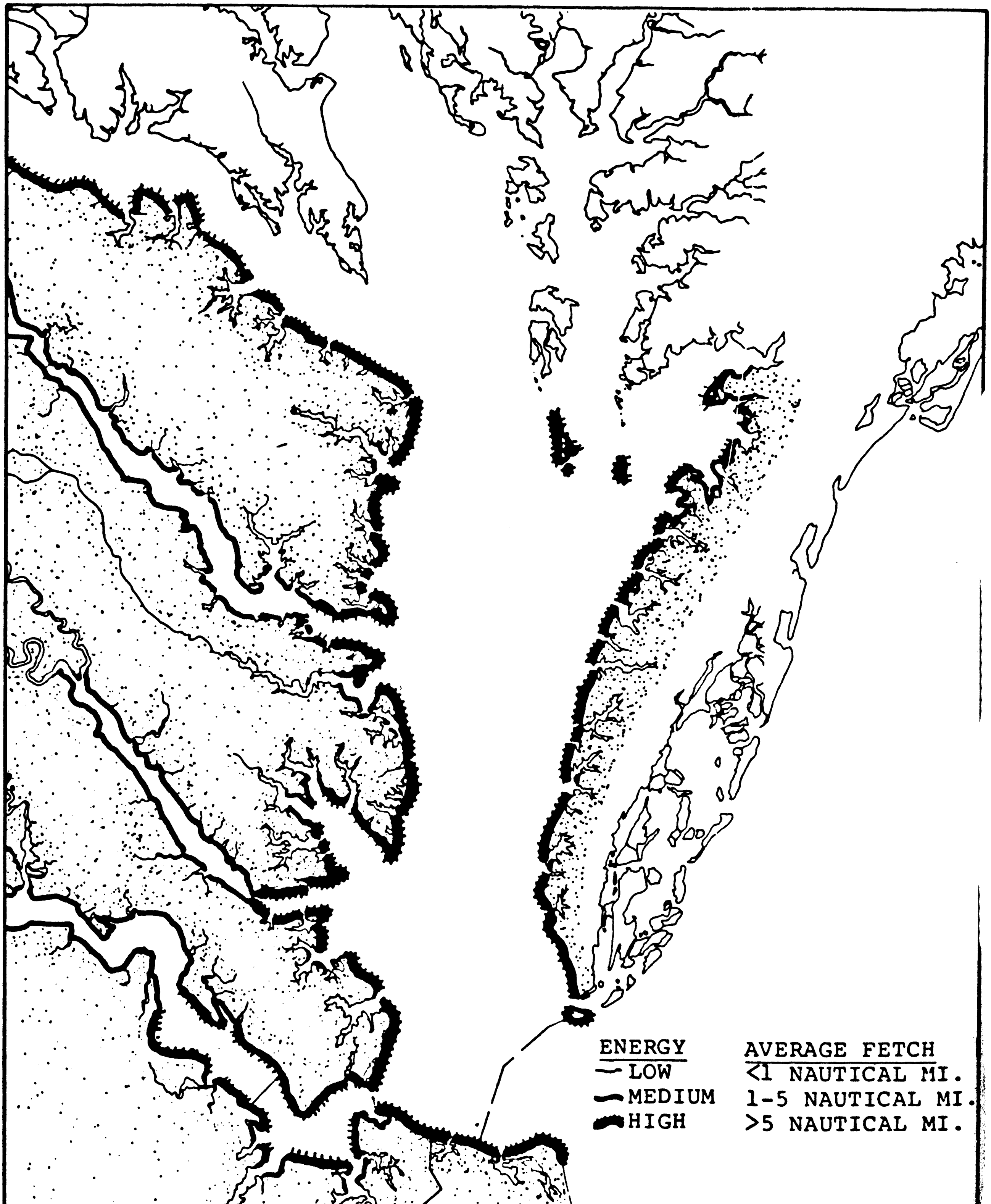

Figure 3. Relative wave energy by average fetch on low, medium, and high energy shores in Virginia's Chesapeake Bay. Shore length measured on stippled portion.

3. On straight shorelines with average fetch exposures of 3.0 to 5.5 nautical miles (mostly found along the lower portions of the major tributaries of the Chesapeake Bay) it will be impractical to attempt to establish a marsh fringe without some type of permanent offshore wave stilling device (i.e., a breakwater). Semi-protected shores will have a better chance of establishment but continual maintenenace planting will be necessary. Design and research on offshore breakwater systems for marsh implantation seems to be an appropriate next step in this line of research.

4. Shorelines with an average fetch of greater than 5.5 nautical miles (mostly along and near the Chesapeake Bay) should not be considered for marsh grass implantation unless well protected by a headland, island, or spit. The use of offshore breakwaters in combination with marsh establishment is a consideration, but further research will be needed.

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WETLANDS FOR EFFLUENT DISPOSAL

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ABSTRACT

This paper addresses the use of existing wetland aquatic plant communities in a freshwater wetland in Florida that are used as a means of effluent disposal of treated wastewater. Evaluation strategy is addressed from several points of view relevant to the effectiveness of wetlands in handling water and nutrient loads. Background projects that have been in operation in Florida over the past 20 years will be discussed. The body of the report will discuss actual projects that Sheffield Engineering & Associates, Inc. have designed and obtained Department of Environmental Regulation approval and are into the monitoring phase. A background data monitoring program is explained and depicted in graphics. System design and construction is discussed and emphasis is placed on water quality as effluent passes through the sampling stations throughout the system. Illustrations within the report graphically depict four years of background data and monthly operation data.

INTRODUCTION

Problems associated with disposal of treated municipal wastewater effluent have plagued man throughout history. History relates that the Romans had a well designed and constructed sewer systems, but these sewers discharged into a log bog area that created a septic nuisance and an epidemic of Yellow Fever from the mosquitoes it produced. Today we read that city, county, and subdivision wastewater treatment facilities throughout the country are causing problems for our surface and groundwaters.

What can a county, city, utility company or private developer do with effluent from their "required" sewage treament plant? There are many alternatives that could be considered for effluent disposal when designing a sewage facility. If the expected flow is less than 0.5 MGD, an evaporation/percolation pond may work successfully if the soil's data indicates a good percolation rate. Other alternatives to meet effluent allocations might include land spreading, advance waste treatment (AWT), deep well injection, wetland disposal, or artifical marsh disposal.

Ponding, spraying, land spreading, injection, and AWT have proven to work under certain conditions, but for the most part are expensive to construct and operate.

STUDY SITES

There are only a few active projects using wetlands as means of assimilating wastewater treatment plant effluent. This section will review actual projects that Sheffield Engineering & Associates, Inc. has either designed and obtained Department of Environmental Regulation approval and are constructed or are in the construction phase.

Lake County Wetland

The owner of a 85,000 GPD extended aeration plant was directed by the Department of Environmental Regulation to eliminate wastewater discharge into Lake Gibson, which is near Lakeland, Florida. Alternative means of effluent disposal were reviewed and it was ascertained that pumping to a 15 acre adjacent wetland would be the most cost effective method of disposal. The 15 acre wetland is basically a cypress dome with a few sweet bay and swamp tupelo. The transitional zone consists of rooted plants including primrose, willows, ferns, and button bush.

A monitoring program was submitted to the Department of Environmental Regulation on October 1, 1976, approved, and consequently enacted in 1976. Sampling stations are shown in Figure 1. Water quality data at four stations were collected on a monthly basis from October, 1976, through December, 1981, and quarterly thereafter. Results of this sampling program indicated a background of 0.19 mg/l Total P, 16 mg/l Chlorides and BOD of 66 mg/l.

Using the Department of Environmental Regulation criteria, a wetland effluent disposal system was designed consisting of a three day detention time pond, pumps, and a 15 acre swamp. At a flow of 85,000 GPD the 15 acre swamp would be loaded at a rate of 5,700 GPD/acre (1.5 in/wk). The plans, application, and other material for this means of effluent disposal were submitted to the Department of Environmental Regulation in February, 1980, and were approved.

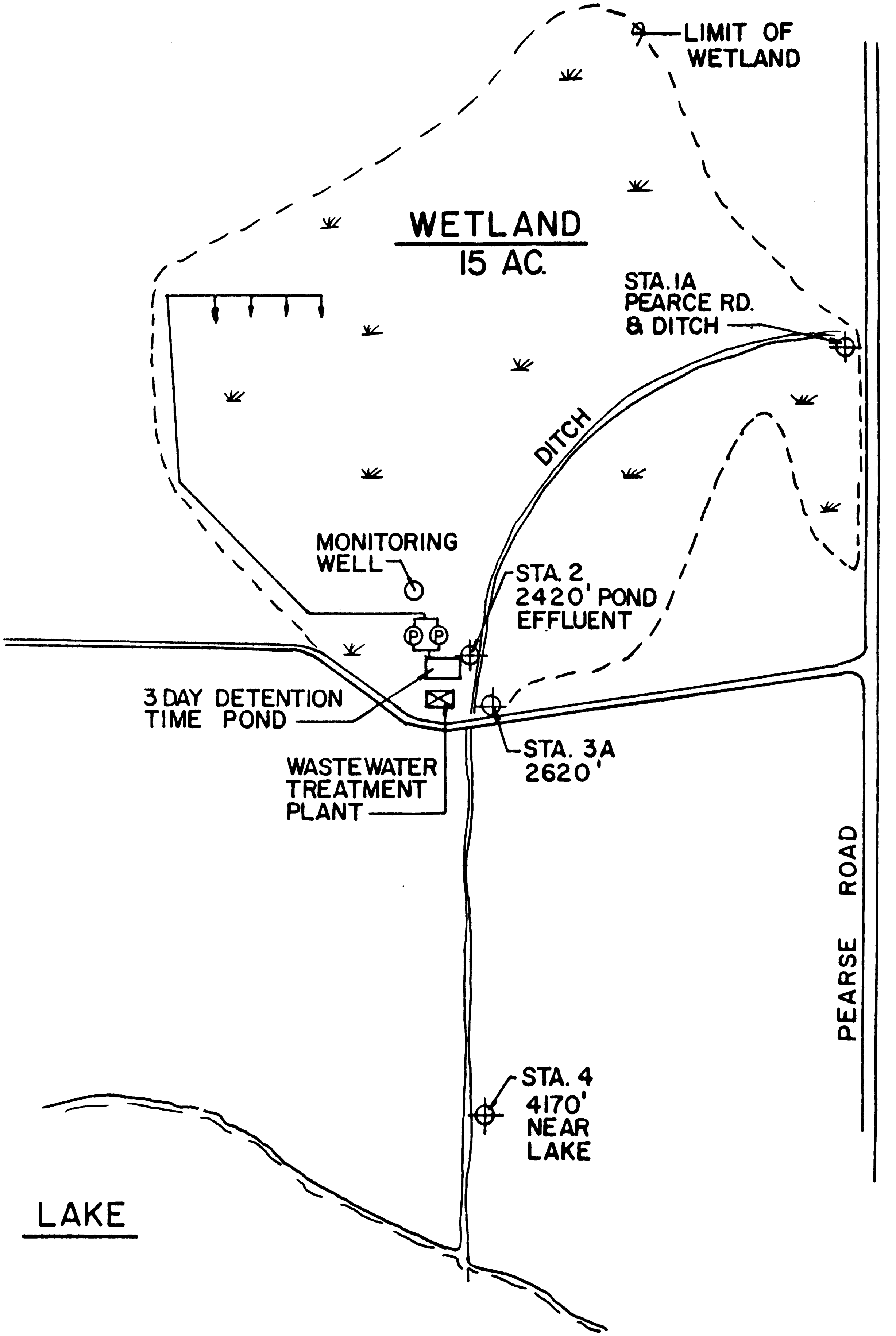
Work on the pond was completed in June, 1980, and the effluent pumping and piping system was completed in January, 1981. The project was placed in operation in February, 1981, and the subsequent monthly water quality sampling has indicated the assimilative capacity of this wetland effluent disposal system.

Total Phosphorus entering Lake Gibson has dropped from an average of 4.38 mg/l to 1.73 mg/l since the direct discharge has been eliminated.

It is anticipated when the ditch that empties into Lake Gibson gets flushed out, the total P will be reduced to less than 0.1 mg/l. The total Nitrogen levels within this ditch has been reduced from 4.5 mg/l to 0.88 mg/l. The monitoring well has a total Nitrogen level of 1.1 mg/l (0.27 mg/l NO_3-N) in the May, 1982, sampling.

Osceola County Wetland

Another project is located in Osceola County just east of Kissimmee,



SAMPLING STATION LOCATION

FIGURE 1

Florida. The existing facilities consist of a 0.75 MGD contact stabilization wastewater treatment plant and a 0.25 MGD extended aeration wastewater treatment plant. These facilities are currently treating 0.45 MGD and 0.069 MGD of sewage respectively. This highly treated and chlorinated effluent is discharged into 6.4 million gallon holding ponds and then pumped into a 39 acre wetland (Fig. 2) and a 130 acre wetland (Fig. 3).

Flora identification within this 206 acre wetland area was determined to be positive for discharge of treated effluent into all or portions of the subject area. The conclusion was this 206 acre wetland would be excellent for wastewater effluent disposal because it has a heavy undergrowth beneath the cypress tress and the transitional zone was distinctly identified which would provide an excellent buffer zone.

Realizing that this wetland disposal is very dependent on keeping the vegetation, especially the trees, in a healthy condition, the project botanist is recommending that each five to seven years a portion of this wetland receiving effluent disposal should be allowed to dry out for propagation of the flora and fauna. Therefore, this wetland totalling 39 acres (lower) plus 130 acres (upper) = 169 acres (after berm construction and isolation) was divided into three separate wetlands as follows: 39 acres of lower wetlands, 60 acres upper portion of the upper wetlands, and 70 acres lower portion of the upper wetlands.

The engineering design reflects that each of these acreages can be isolated and controlled so the areas can be dried out for up to three to six months on a five to seven year cycle.

The preconstruction monitoring program on this swamp commenced in September, 1979, on a weekly basis for four weeks and since has been monthly. Three stations were set up, as shown in Figure 3, in the upper lobe, middle and southern portion of this wetland. Flow was measured at the middle and southern stations. The six months sampling results indicate natural background data of 23-36 mg/l Chlorides, 2.18-2.54 mg/l total N; 0.05-0.06 mg/l total P and BOD of 2.0-2.6 mg/l. These values are very typical for a fresh water cypress swamp in central Florida.

The premise for operating this wetland effluent disposal is: inflow will equal storage plus outfall with the inflow being rainfall plus highly treated effluent and the outflow will be from percolation, evapotranspiration, and emergency outfall in the event of a one in ten year storm or a one in ten year "wet" month.

The design of the 39 acre wetland disposal includes a 48 inch high berm around the entire wetland which will allow for 30 inches of water storage. An outfall structure was constructed in case the water level increased over 30 inches. The berm was 20 to 30 inches wide on the top with 5:1 side slopes on both sides which resulted in a total width of the berm being 60 to 70 inches. When the original design criteria was set for this wetland, it was based on normal rainfall in a month within central Floroida which is 4.25 inches. The peak month in 1984 was July which generated 11.81 inches. It is assumed that evaporation will be 1/8 inch on dry days and 0 inches on rainy days. The transpiration will be approximately 1/4 inch on dry and 1/8 inch on wet days. Soils

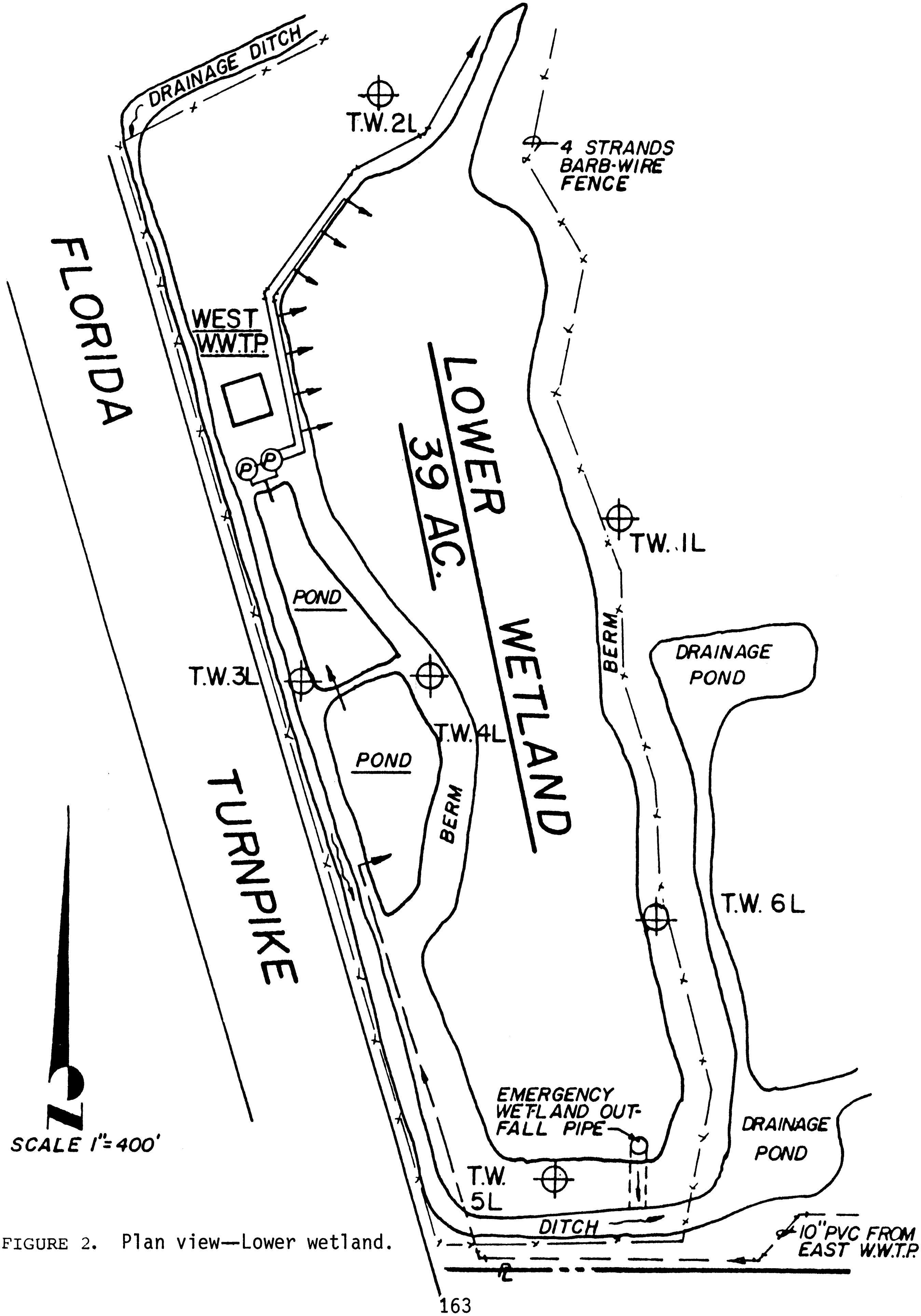
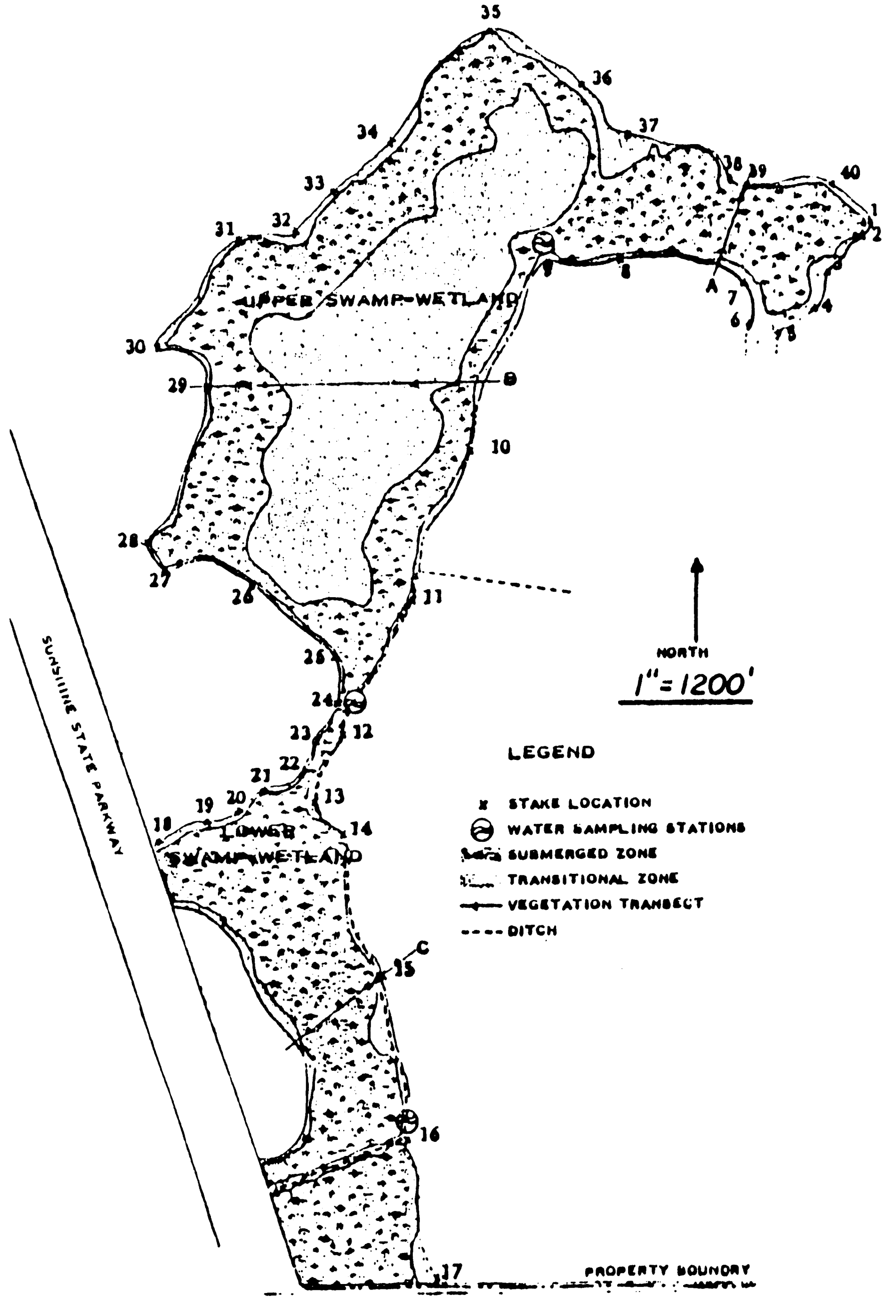


FIGURE 2. Plan view—Lower wetland.



PLAN VIEW
UPPER & LOWER WETLAND

FIGURE 3

studies done by Jammal & Associates show that the wetland can percolate a portion of the flow and this will occur downward due to the berm around the swamp that was compacted to 98 percent Proctor which has virtually eliminated any horizontal seepage in the wide berm.

In order to dry out these wetlands, the lower 39 acres will be kept in their existing state, the upper wetlands, which were recently provided with a construction permit, will be divided by a low level berm to provide a separate 60 and 70 acres wetland. This low level berm will be provided with pipes through it allowing continuous flow when a particular wetland isn't being dried out each five to seven years. The approved plans indicate details on this isolation system.

Both the wetland influent and the lower section of the 39 acre wetland plus six monitoring well stations have been sampled since the highly treated effluent was discharged into the wetland in April, 1982. As stated earlier in the article, the natural background information indicated values typical for a central Florida wetland.

RESULTS AT OSCEOLA COUNTY WETLAND

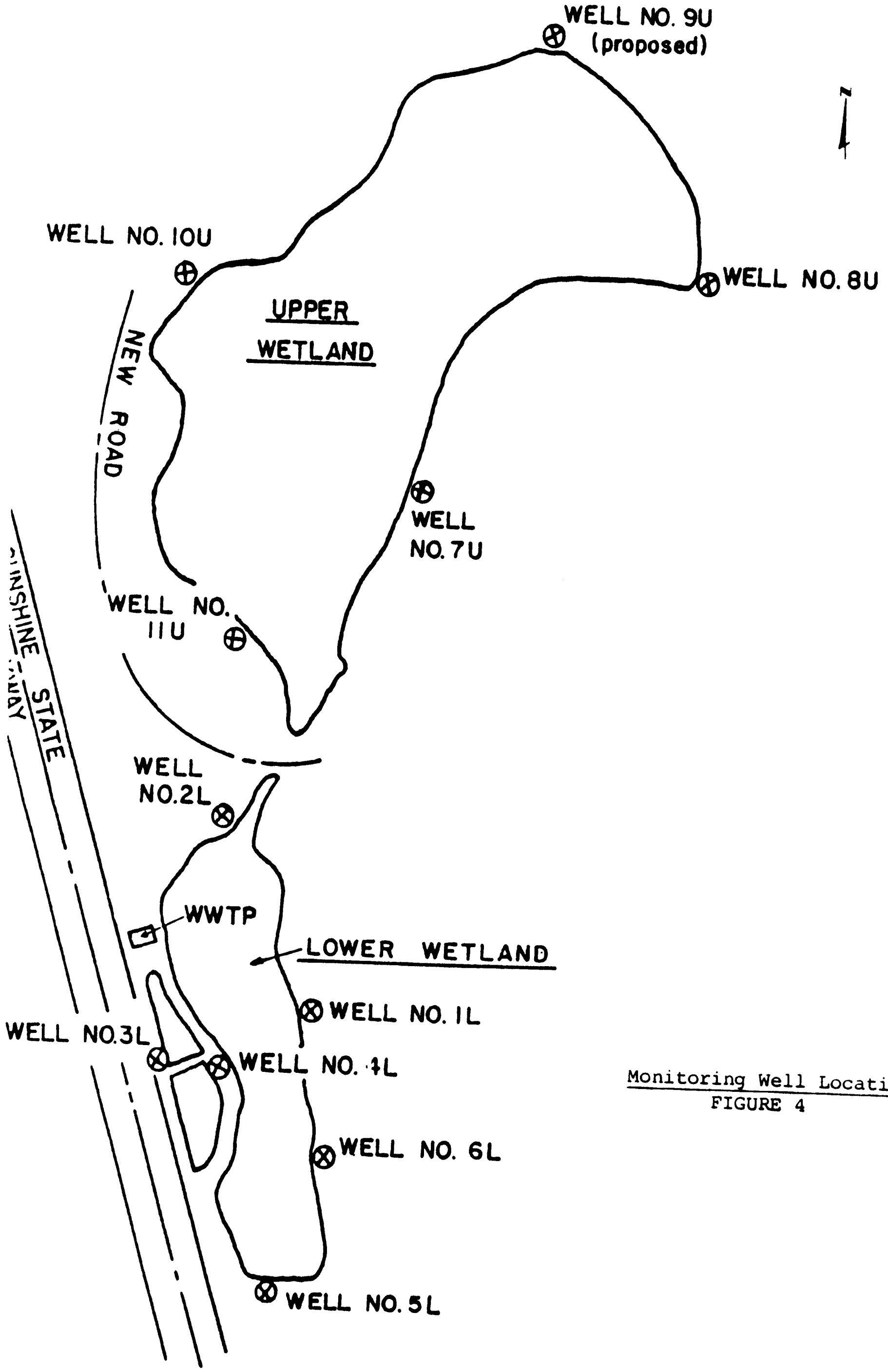
The lower wetland has a total of six active monitoring wells and the upper wetland has five wells as shown in Figure 4. These wells are approximately 30 to 50 feet from the water line in the wetland with well number 1L being 100 feet from the water in the wetland.

Temporary well number 1 (now 5L, south of the wetland), number 2 (now 1L, east of the wetland), and number 5 (now 3L, west of the wetland) were installed prior to construction of the berm around the lower wetland then taken out of service and permanent wells installed in their place. The arithmetic average results of the background sampling of these wells indicated the following, as shown on Table 1.

Table 1.

Well Location	Cl	Total-N	Total-P	Cond.	BOD
(5L) #1 South	34	2.97 mg/l	0.11 mg/l	188	6 mg/l
(1L) #2 East	48	1.98 mg/l	0.06 mg/l	269	8 mg/l
(3L) #5 West	29	1.53 mg/l	0.05 mg/l	163	18 mg/l

Monthly results of the monitoring wells around the lower or upper wetland were tabulated and the average results at these stations from 4/27/82 up to 1/11/84 are as shown in Table 2.



Monitoring Well Location
FIGURE 4

Table 2.

Well Location	Cl	Total-N	Total-P	Cond.	BOD
#1L	23	1.75 mg/l	0.11 mg/l	129	4 mg/l
#2L	49	1.46 mg/l	0.25 mg/l	233	4 mg/l
#3L	35	1.31 mg/l	0.17 mg/l	130	4 mg/l
#4L	45	1.62 mg/l	0.10 mg/l	186	5 mg/l
#5L	41	1.67 mg/l	0.07 mg/l	188	6 mg/l
#7U	44	2.06 mg/l	0.29 mg/l	203	6 mg/l
#8U	55	0.51 mg/l	0.52 mg/l	260	6 mg/l
#10U	55	0.11 mg/l	1.21 mg/l	145	6.5 mg/l
#11U	53	0.08 mg/l	0.42 mg/l	148	1.1 mg/l

Well #9U is a proposed well and sampling will begin in the near future.

Since April, 1982, the influent into the wetland has been averaging total Nitrogen of 5/7 mg/l; total-P of 2-3 mg/l; BOD of 10± mg/l; and flow of 450,000 to 550,000 GPD. The lower portion of this wetland has had sporadic flows, when the rainfall exceeded 3 inches in a 24 hour period, of 100,000 GPD. The water quality averaged at pH of 5.2, total N of 2.25 mg/l, total P of 0.16 mg/l and BOD of 5.5 mg/l. It is interesting to compare the wetland water quality values before and after the highly treated effluent was discharged into this 39 acre wetland, where the total N was 2.54 mg/l before.

One measure of success in any project of this nature is the sample results of a shallow monitoring well sampling program. From the data presented on both background and after discharge into the wetlands, it can be concluded none of the parameters being tested are increasing. It is interesting to note that the total N and Chlorides within these wells seem to have decreased and stabilized at current values.

CONCLUSIONS

The use of freshwater swamps as a means of effluent disposal is comparatively new in Florida. The literature on this subject is mainly on proposed projects or ones that are uncontrolled without good effluent disposal, or outside central Florida's climatic regime.

The Florida Department of Environmental Regulation seems to have an excellent set of guidelines on use of swamps for effluent disposal with their Rule 17-4, Permits, 17-4.243 which provides for the experimental use of wetlands for wastewater assimilations.

From the information reviewed in the literature and our preliminary field and design work, it seems that the use of an isolated freshwater forested wetland could provide an excellent means of effluent disposal. This is true if sufficient background data and continuing water quality and biological monitoring programs are provided. The wetland must be isolated from surrounding areas and the suggested loading rates are dependent upon soil types, the vegetative species that are present in the wetland community, and the amount of rainfall for a given locale.

It is recommended that every five to seven years wetlands used for disposal should be allowed to dry out for propagation purposes.

ADVANCES IN THE MANAGEMENT OF WILD RICE PRODUCTION IN NORTHWESTERN ONTARIO LAKES

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ABSTRACT

Wild rice has been successfully introduced as a cash crop in Canadian lakes. Early seeding experiments indicated that production in these lakes is influenced by water depth, sediment chemistry, plant competition, and the genetic variation of the seed source. Wild rice depth tolerance is primarily affected by photosynthetic light penetration. Wild rice can tolerate increases in water depth of up to 50 cm during the submerged, floating-leaf, and emergent-leaf stages, provided there are no other limiting factors. Water level management is directed at maintaining a stable, optimum depth. The nutrient regime of the sediment in which wild rice grows varied both temporally and spatially causing difficulties in diagnosing exact nutrient requirements. Best production increases results when slow-release fertilizers are used. Plant competition occurs primarily from other emergent species with the most important competitor being Eleocharis spp. Control of these emergent aquatics is achieved with a modified rope-wick herbicide applicator. Genetic variation exists among wild rice populations in terms of seed production, time to maturity, and depth tolerance. The utilization of the appropriate seed source for particular environmental conditions is an essential management strategy. Future expansion of the lake wild rice industry in Canada will depend on its ability to mechanize and to increase yields to levels currently achieved by paddy production in the United States.

INTRODUCTION

Wild rice grows extensively on the many thousands of hectares of freshwater wetlands found in northwestern Ontario. The major stands are in the shallow bays of Lake of the Woods, Rainy Lake, and the Winnipeg River. Other important production areas are found west of Red Lake and near Lac Seul and Pickle Lake (Lee, 1979). The harvest from these rice areas forms an important part of the local economy with an economic value in good crop years of several million dollars. Unfortunately, the annual production of wild rice fluctuates drastically. Harvests of over 700 mt are often followed by complete failures with production of less than 13 mt. The major reason for these fluctuations is the periodic high water levels on the larger water bodies (Lake of the Woods, Rainy Lake, Winnipeg River) which are not controlled for wild rice production.

In order to solve this problem, a program was initiated in 1974 to determine if there existed any shallow lakes of suitable type and size to support commercial production of wild rice and which were not subject to severe water level fluctuations (Lee, 1974). Experimental seeding of 32 of these lakes was encouraging (Lee & Steward, 1984). As a result, in the next three years, several hundred licences were issued to interested private developers for commercial production of wild rice in similar lakes.

With increased activity, production problems arose. For wild rice to be developed as a commercial crop in northwestern Ontario, research was required to develop the required management techniques. In 1981, a program sponsored by the Ontario Ministry of Northern Affairs was initiated to examine the major wild rice production problems: water depth, nutrient requirements, weak control, and stand establishment (Lee, 1982, 1983, 1984, 1985).

This paper summarized some of the key techniques and findings of this research program.

METHODS

The general procedure for all our studies was to sample the wild rice population under natural field conditions, analyze the results to develop hypotheses, test these hypotheses under controlled environmental conditions, develop management strategies, and finally, test these management strategies under field conditions.

Field sampling designs varied with the type of research problem. The strategy for water depth studies was to minimize the variation in plant density and soil nutrients by establishing small (10 m x 10 m) sampling sites within the wild rice stands at specified depths. Random samples were taken throughout the growing season (May to September). For nutrient and genetic variation studies, the variation of plant and soil nutrient characteristics were maximized. Random samples were taken after the wild rice plants achieved maximum biomass. The stand was gridded into 100 m x 100 m (1 ha) cells and randomly sampled. Using a small boat with an air-prop motor, traveling at a constant speed, I estimated how long it took to reach each required grid cell according to the distance per unit time the boat traveled. The boat then proceeded along transects stopping at known time intervals to sample the selected cells.

Quadrats, 0.5 m x 0.5 m, were sampled. At each quadrat the wild rice plants and other macrophytes were removed, a sediment sample taken with a plastic corer, and the depth measured in the center of each quadrat. All material was placed in plastic bags and stored in portable coolers for transport to the laboratory.

In the laboratory, all plant material was oven dried at 80° C until no change occurred in weight.

The sediment samples were analyzed in their wet condition since drying of sediments may affect the levels of many nutrients (Richardson et al., 1978). A modified 50 ml syringe was used to remove 20 cc subsamples from each sediment sample (while still water saturated) for use with all extracting solutions. Phosphorus was extracted with Bray-P1 solution (Jackson, 1958) and analyzed with a Bausch and Lomb spectrophotometer, Model 21, using ammonium molybdate as an indicator. Nitrogen, present as ammonia, was extracted with 2N KCL (Jackson, 1958) and analyzed with an Orion ammonia-specific gas electrode connected to a Fisher Accumet selective ion analyzer, Model 750. Iron, manganese, zinc, and copper were extracted with 0.1N HCL and measured with a Perkin-Elmer atomic absorption spectrophotometer, Model 2380. Calcium, magnesium, and potassium were extracted with ammonium acetate solution (Chapman & Pratt, 1961) and measured with the same atomic absorption spectrophotometer. The percent loss on ignition was calculated fromthe change in dry weight of a 20 cc sample placed in a crucible and ashed at 600° C for eight hours in a muffle furnace (Jackson, 1958). Concentrations of all elements were calculated on a volume basis (1 m x 1 m x 20 cm soil depth) but expressed on a square meter basis.

Environmental conditions for experiments were controlled either by using large floating rafts with suspended plastic buckets containing wild rice (Fig. 1) or by using cultivation tanks under greenhouse conditions (Fig. 2). A larger number of plants can be grown using rafts, however, they can only be employed in the summer months.

Field trials were conducted in experimental study areas near Ignace, Ontario, and Nestor Falls, Ontario, described by Lee (1982, 1983, 1984).

RESULTS AND DISCUSSION

Water Depth

Water depth is the most important factor influencing wild rice production. The ideal depth is approximately 0.3 to 0.6 m. Increases in depths from preceeding years of 0.3 m in Minnesota and 0.6 m in Ontario have been shown to nearly eliminate the crop (Moyle, 1944; Lee, 1975). Deep water does not allow sufficient light penetration for adequate photosynthesis.

Sudden fluctuations in water depth are also detrimental to wild rice growth and have been reported to cause severe damage to the crops (Chambliss, 1940). Such fluctuations are common in the larger water bodies of northwestern Ontario.

The objective of our water depth research has been to determine which phenological stages of wild rice development are most affected by water level fluctuations, and how much fluctuation wild rice can tolerate. Such information could be used by water level control agencies to time water release from control dams to lessen damage to wild rice crops.

Under natural conditions on Lake of the Woods, we found a rapid decline in wild rice population density (Fig. 3) during the floating-leaf stage when water depths increased 12 cm during a two week interval (Atkins, 1983). At the submerged, first floating-leaf, second floating-leaf, and first aerial-leaf stages, the plastic buckets containing wild rice were lowered 1, 15, 30 and 50 cm. Sampling of plants was done at weekly intervals to assess the effect of these fluctuations.

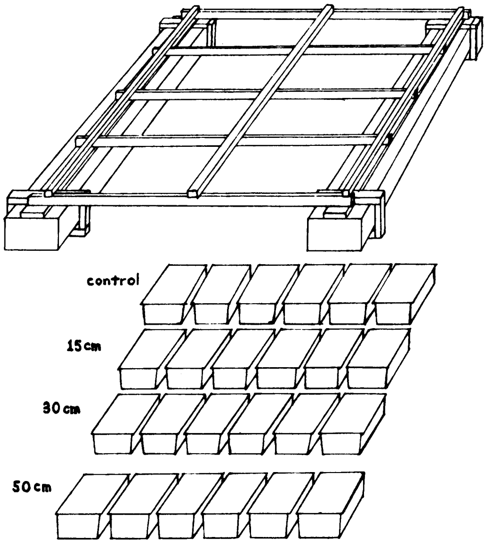


Figure 1. Design of rafts used to cultivate wild rice under controlled environmental conditions. Tubs are suspended by polypropylene ropes from the wooden latticework attached to polystyrene floats.

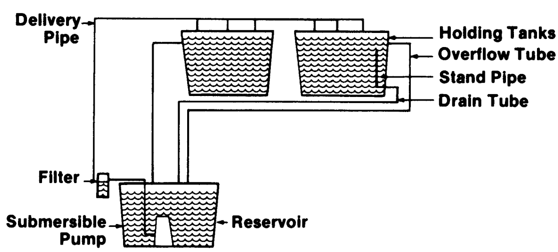


Figure 2. Apparatus used to cultivate wild rice in an algae-free environment.

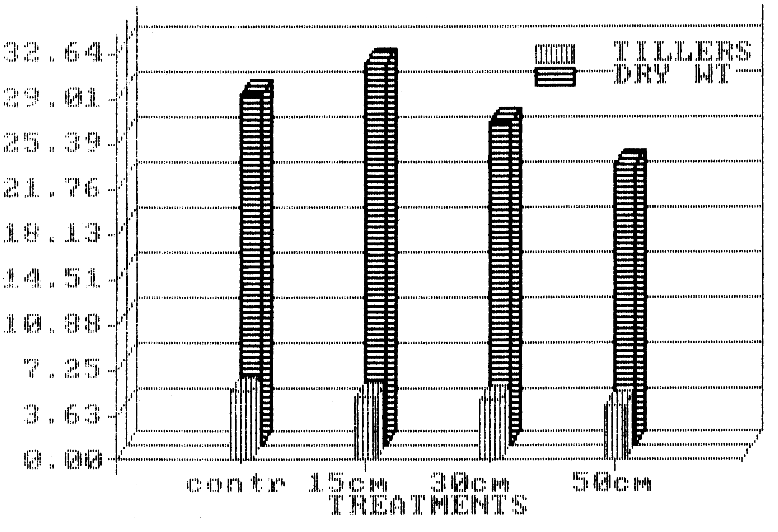


Figure 3. Final harvest dry weights per wild rice plant and number of tillers per plant of wild rice plants lowered during the second floating-leaf stage.

Based on our field research, we had anticipated a high mortality when the plants were lowered. This did not happen. Not one plant died, and even those plants lowered 50 cm were able to survive and were extremely vigorous.

The final harvest dry weights of the plants lowered during the second floating-leaf stage are shown in Figure 3. The wieght and number of tillers decrease with increasing depth treatment, but even at the 50 cm treatment, the plants weigh an average of 23 g and have an average of 4 tillers per plant.

These results show that depth fluctuations have little effect on wild rice survival and growth. Yet in the field, survival is limited, and growth is adversely affected. The only difference to field conditions was that the rice plants were heavily fertilized so there would be no nutrient stress to the plants. This suggests that it is not simply water level fluctuations which cause mortality; rather, it is water depth increases under nutrient stress conditions. The implication is that it may be possible to manage for water depth increases by ensuring there is no nutrient stress on the plants. We are currently examining this possibility.

Nutrient Dynamics

One of the major problems in wild rice lakes is that nutrients are depleted as a result of intensive management. Large amounts of biomass are produced in the central portions of these lakes each year (Fig. 4), and in the spring the straw is swept to shore by the wind, removing large amounts of nutrients.

Under controlled greenhouse conditions, it has been shown that wild rice growth responds best in such nutrient-depleted soils when slow-release fertilizers are used (Lee, 1983). Under field conditions, production was also enhanced when slow-release fertilizers were added to an intensively managed lake while there was no effect on unmanaged lakes (Lee, 1985) (Figure 5). In the managed lake, the weights per plant and the number of seeds per panicle increased with higher fertilizer concentration.

This suggests that nutrient depletion is occurring in these managed lakes and that it may be possible to reverse this situation by adding these fertilizers. However, there are a great number of technical, environmental, and economic difficulties with this technique. The equipment does not exist to add the required nutrients to the bottom sediments without releasing any nutrients to the water column. The cost of the slow-release fertilizer is also much greater than that of conventional fertilizers. Although nutrient replenishment may become a necessity if a stable annual production is desired, considerably more basic research is required before this becomes a common wild rice cultivation technique.

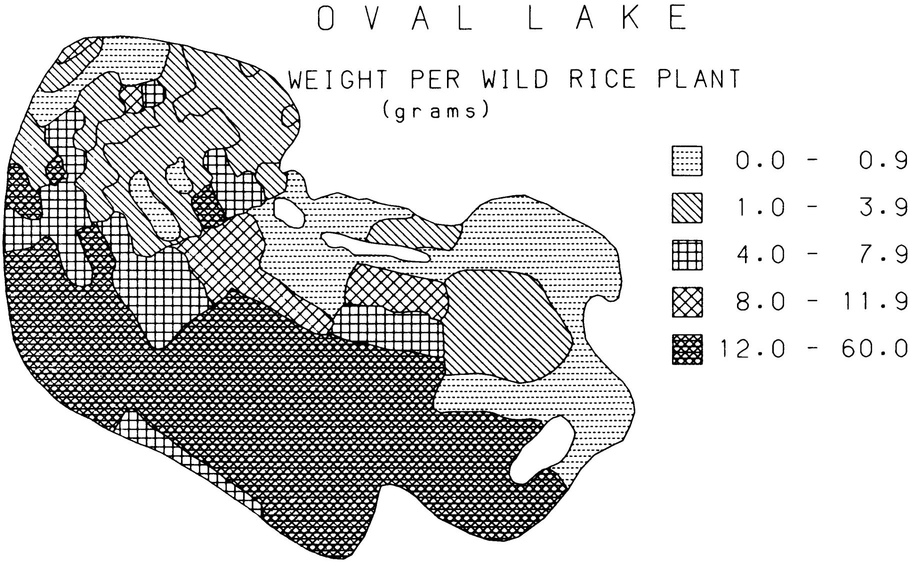


Figure 4. Wild rice production in a typical wild rice lake. The biomass accumulated in the central portions of the lake is swept to shore in the spring.

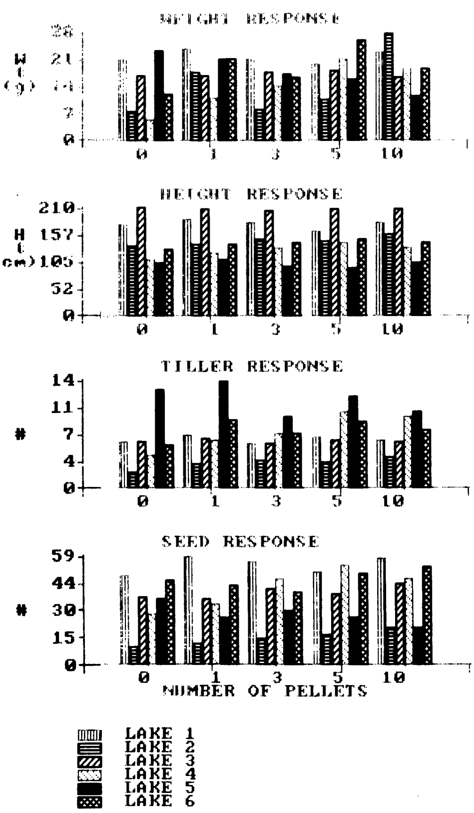


Figure 5. Plant performance characteristic for the field experiments at six lakes.

Weed Control Research

Our weed control research is primarily concerned with eliminating competent emergent aquatic macrophytes. The most common problem species is Eleocharis spp.

We have tried various techniques to control this weed using mechanical methods. Periodic cutting of the stems during the growing season had no effect on this plant. A second theory was that the rhizomes in winter depend on the culms protruding through the ice for their supply of oxygen: if the culms were cut off at ice level, the plants would die. This technique was reportedly used with success for removing Typha spp. on prairie sloughs in western Canada. We tested this idea by pulling a cutting bar behind a snowmobile. No damage was observed with the Eleocharis and the technique was abandoned.

Best success was achieved in controlling these emergent macrophytes by using herbicides which are applied with a modifed brush-on type of applicator (Fig. 6). The herbicide can be applied to the target species while wild rice is still in the floating-leaf or submerged stage. The boom, which is saturated with herbicide, can be raised or lowered depending on the height of the competing species. This method gives nearly complete control of the competing species and has no effect on wild rice. This method also has two distinct environmental advantages: (1) no herbicide is directly applied to the water, and (2) amounts required for control are one-hundredth to one-fiftieth the amounts recommended for conventional spraying applications.

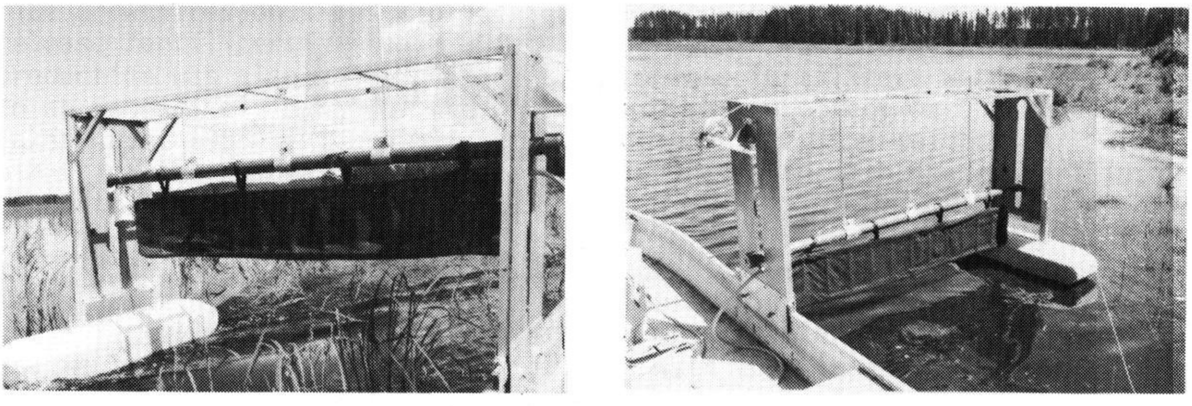


Figure 6. Brush-on herbicide application to control competing emergent aquatic plants.

Stand Establishment Research

Research with Minnesota cultivars has revealed genetic variability for certain morphological and phenological traits (Foster & Rutger, 1980) and phenotypic plasticity in response to temperature and daylength (Oelke et al., 1981). However, little information exists on genetic and phenotypic variation in lake wild rice.

Our stand establishment research is primarily concerned with determining if genetic differences exist among wild rice populations, and if there are any seed sources more suited to such conditions as deeper waters, shorter growing seasons, clay soils, and which can produce higher yields (Counts, 1983, 1984).

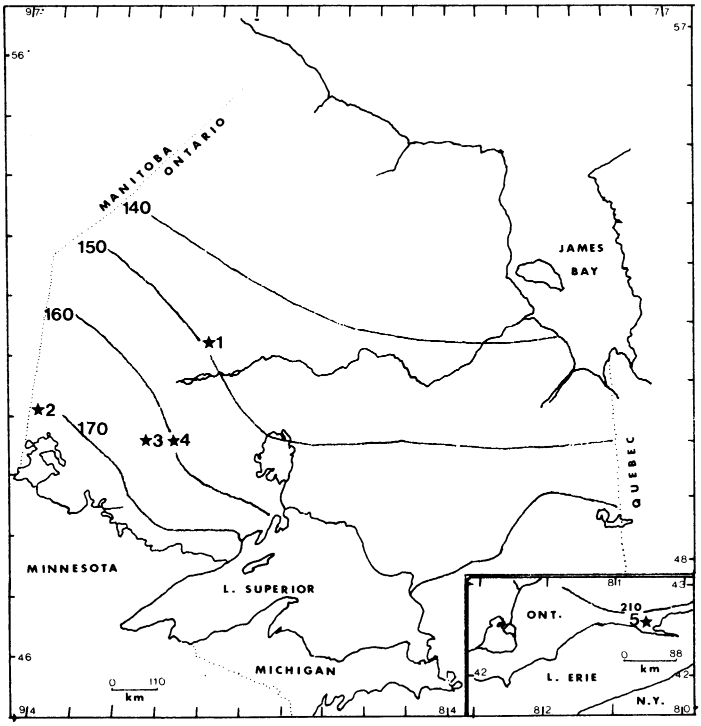
A greenhouse comparison of seed sources from a widespread geographical region (Map 1) grown in nutrient enriched organic and clay soil, revealed that populations varied in the number of surviving plants, number of stems per plant, height, number of female florets per stem, and number of seeds per plant (Fig. 7). The differences were greatest between the most northern and most southern populations. The northern plants tended to be shorter, have fewer seeds, and mature more quickly than the southern plants. There was little variation in plants grown on clay versus organic soil. Since wild rice normally grows poorly on clay soil, this suggests that a deficiency of mineral nutrients, rather than soil texture, is the dominant factor affecting wild rice productivity.

Continued investigations revealed that certain populations are better adapted to deeper water. This trait was demonstrated under greenhouse conditions and further tested by seeding four depth-tolerant populations in the same lake. The four sources were separated from each other by open water to prevent intermixing. Depth tolerances under field conditions were similar to those seen in greenhouse trials, varying from 90 to 120 cm.

Genetic variation, therefore, exists among natural populations of wild rice, and there are very desirable traits in many of these populations which could greatly increase the production of wild rice in lakes. Future research will concentrate on utilizing these characteristics in a wild rice breeding program to develop varieties of wild rice best suited to the environmental conditions at hand.

CONCLUSIONS

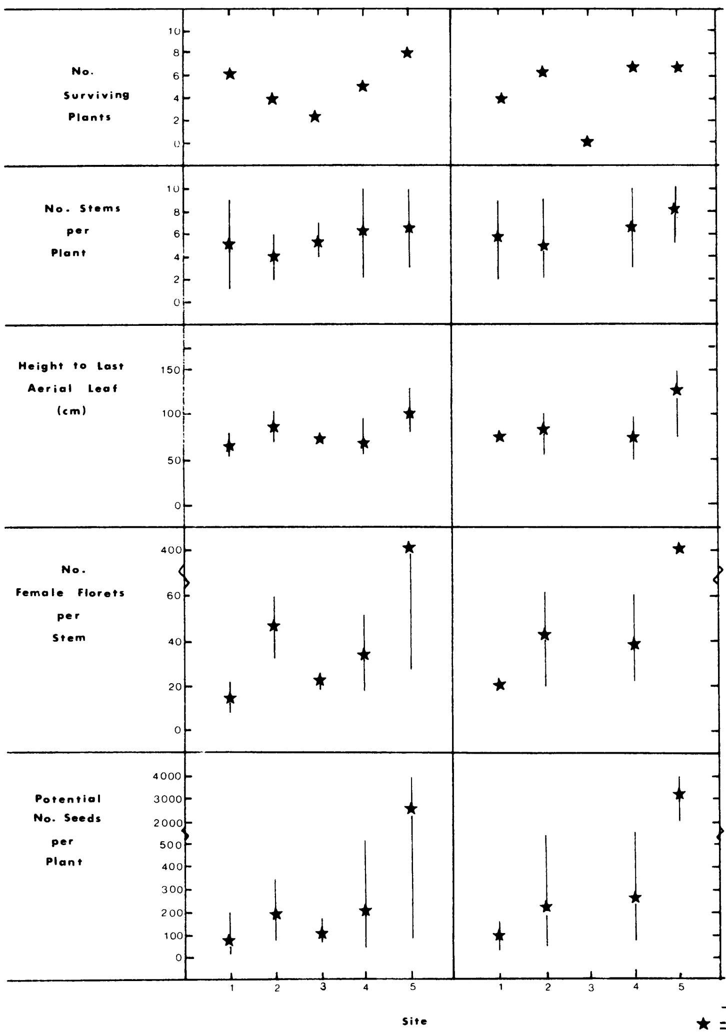
The management of wild rice in lakes is still in its developmental stages. Research has shown that water level fluctuations are not a severe problem under conditions of optimum nutrient levels. The use of slow-release fertilizers can greatly enhance the growth of wild rice in lakes which have been depleted of essential nutrients by intensive cultivation. Control of competing emergent aquatic macrophytes is most effectively accomplished with herbicides applied with a brush-on applicator. Genetic variations exist in wild rice which are expressed in time to maturity, yield, height, and depth tolerance. The significance of these findings is that yields of wild rice in lakes



MAP OF NORTHERN ONTARIO SHOWING SEED SOURCE LOCATIONS AND MEAN ANNUAL LENGTH OF GROWING SEASON (DAYS)

(Inset: North Shore of Lake Erie, Southern Ont.)

Map 1.



Variation in morphological characteristics of five seed sources grown under greenhouse conditions in organic and clay soils.

Figure 7.

can be increased and sustained. However, the application of the results of this research is just beginning. Future research will be concentrated on applying this information to improve wild rice production in Ontario lakes.

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SOIL AND VEGETATION DYNAMICS IN A ROTARY DITCHED MOSQUITO CONTROL IMPOUNDMENT ON THE MERRITT ISLAND NATIONAL WILDLIFE REFUGE

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ABSTRACT

From April, 1981, to July, 1984, changes in soil and vegetation occurred relating to the open marsh water management (OMWM) activities on the Merritt Island National Wildlife Refuge (MINWR). During this period, there was a significant decrease ($P > 0.05$) in four soil elements: soluble salts, phosphorus, potassium, and calcium. Vegetation changes occurred as follows: a significant increase ($P > 0.05$) in spike grass (Distichlis spicata) and total cover, a significant decrease ($P > 0.05$) in perennial glasswort (Salicornia virginica), and significant changes ($P > 0.05$) in annual glassworth (Salicornia biglovii) and saltwort (Batis maritima). Spike grass will respond to soil changes in 1 to 2 years. The data indicated that OMWM strategy had a much greater effect on the soil and vegetation changes than did the rotary ditches. The construction of the rotary ditches had no significant effect on vegetation.

INTRODUCTION

Between 1959 and 1966 approximately 7700 hectares of high salt marsh (e.g., Spartina bakeri, spike grass, saltwort) on the Merritt Island National Wildlife Refuge (MINWR) were impounded to control salt marsh mosquito populations (Godfrey & Wooten, 1979, 1981). This impounding and overflooding significantly changed the ecological relationships of the high salt marsh community.

Mosquito populations of the natural salt marsh were documented as exceeding 100 landings per minute throughout most of the marsh. Following impounding, these landing rates were significantly reduced to an average level of less than 10 landings per minute (Salmela, 1984, personal communications).

The mosquito control impoundment system has significantly changed other ecological relationships as well. Sweet (1976) found that the natural salt marsh community following flooding could evolve into many different plant communities depending on the water depth and salinity. Trost (1964) found that impounding changed the vegetation community which was reflected by bird populations and seriously hurt one, reduced five, had no effect on seven, helped eight, greatly helped twelve, and phenomenally helped two species of birds. Leenhouts and Baker (1982) found significant vegetation changes in another impoundment at the MINWR between 1973 and 1980 associated with dewatering for dusky seaside sparrow (Ammospiza maritima nigrescens) management.

An open marsh water management (OMWM) experiment was conducted within one impoundment of the refuge in 1981. The objectives of this project were to maintain tolerable levels of salt marsh mosquito production and return much of the natural ecological processes of the original salt marsh. OMWM is the hydrologic connection of mosquito breeding sites within the marsh with the open marine system thus allowing larvivorious fish to move throughout the mosquito breeding areas and feed on juvenile mosquitoes. In the case of MINWR, the ditches were constructed with a rotary ditcher. It was anticipated that significant changes in the impoundment's ecological processes would take place. Shisler and Jobbins (1977) found significantly higher total vegetational biomass produced in the mosquito ditched marsh compared to that of the unditched marsh in New Jersey. A monitoring project was developed to evaluate changes in fish and invertebrate populations, water quality, and salt marsh vegetation. This paper reports the changes in soil and vegetation as a result of this OMWM project.

STUDY AREA

The study area was the salt marsh community within mosquito control impoundment T-10-H, a 136 ha impoundment on the MINWR, Brevard County, Florida (Fig. 1). The climate of the area is subtropical with a mean annual temperature and rainfall of 25° C and 150 cm respectively.

The fall of 1980 was the beginnning of a drought which lasted until late summer, 1981. This drought evaporated water from the Indian River and many of the impoundments including impoundment T-10-H. Normal rainfall returned in 1982 through 1984. In December, 1983, a freeze occurred with termperatures reading a low of -5° C and wind speeds of 25 mph for two nights.

The historic, infrequently inundated, high salt marsh was described by Trost (1968) as containing glasswort (Salicornia spp.), saltwort and spike grass along the shore of the Indian River and tributaries. Cordgrass (Spartina bakeri) and scattered stands of black needle rush (Juncus roemerianus) made up the interior of the marsh. In elevated areas, small stands of sea-oxeye (Borrichia frutescens) and groundsel-tree (Baccharis halimifolia) were found. Impoundment T-10-H exhibited these vegetation characteristics before impounding. Since construction of the mosquito control impoundments, wax-myrtle (Myrica cerifera), groundsel-tree, and Brazilian pepper-tree (Schinus terebinthifolius) have extended their range into the salt marsh probably via the dike system. Black mangrove (Avicennia germinans) and white mangrove (Languncularia racemosa) also expanded along both the estuarine and impoundment edge of the dikes.

Impoundment T-10-H began to function as an impoundment by the late 1950's, and by 1962 a permanent dike system enclosed the salt marshes of north Merritt Island. Flooding the impoundments for the control of salt marsh mosquitoes with from 10 to 45 cm of fresh water began as each impoundment was completed. To test the potential artesian water

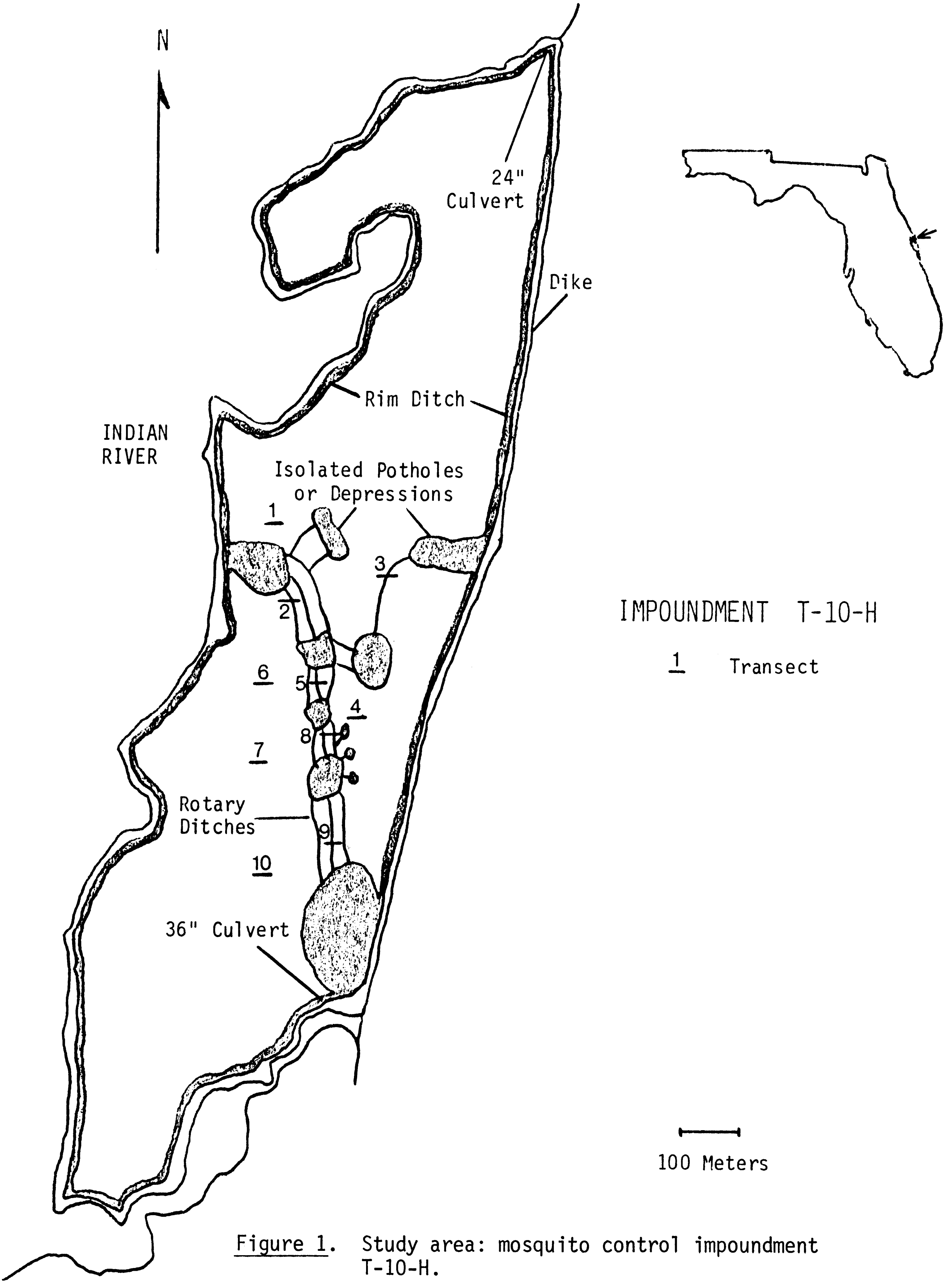

Figure 1. Study area: mosquito control impoundment T-10-H.

supply for a supplemental water supply in north Brevard County, the Brevard Mosquito Control District drilled two brackish water (approximately 20 ppt salt) artesian wells which flowed freely into the impoundment in the late 1950's. The overflooding and additional salt caused the vegetation to change until most of the impoundment's original vegetation was lost (Salmela, 1984, personal communications). In an attempt to revegetate the impoundment, it was dewatered in 1974 and left open to the estuarine system through a 60 cm culvert at the northeastern corner of the impoundment. In 1981, the rotary ditch system and a 90 cm culvert at the southeast corner of the impoundment was installed. In 1983, the rotary ditches were reditched for maintenance.

METHODS

To study the effects of rotary ditching on the salt marsh soil and vegetation dynamics, 10 permanent 30 m line transects were systematically established in an east-west orientation in April, 1981, before ditching. Five of the transects were located in areas selected to be ditched and where the transect would bisect the ditch, and five were located in areas with similar vegetation types that would not be ditched.

Soil

Soil was collected with a standard 20 cm sampler from each transect in 1981, 1983, and 1984, and analyzed for soluble salts, pH, phosphorus (P205), potassium (K20), calcium (CaO), and magnesium (MgO) at the University of Florida, Institute of Agricultural Sciences, Soils Laboratory.

Vegetation

Thirty plots were established per transect by placing a steel frame 20 x 50 cm (inside dimensions) at 1 meter intervals along a steel tape. Percent ground cover of each species found at each plot was recorded for all ten line transects. Class ranges for percent cover were 0-5, 5-25, 25-50, 50-75, 75-95, and 95-100. Mean values for percent cover were calculated using midpoint values for each class: 2.5, 15, 37.5, 62.5, 85, and 97.5 respectively. The presence or absence of a species in each plot determined the frequency of occurrence of the species (Daubenmire, 1959).

Data on the line transects were collected on April, 1981, before ditching and on July, 1981, July, 1983, and July, 1984, after ditching.

Statistical analyses were performed on both soil and vegetation data for each transect, all transects, ditched transects (2, 3, 5, 8, and 9) and unditched transects (1, 4, 6, 7, and 10). Analysis of variance was used to test the difference of each soil element between years for ditched, unditched, and total transect data. The Kruskal-Wallis test was used to test the difference in mean species percent cover between years for each species in the ten transects, all transects, ditched transects, and unditched transects. A null hypothesis, that there was no significant difference ($P < 0.05$) in each soil element or species percent cover (two-sided analysis) for each comparison, was tested.

RESULTS

Soil

Of the six soil elements, only five were tested. Magnesium was not tested because values for most transects reached miximum values. The results of the soil elements are given in Table 1. All tested values except pH showed significant decreases between 1981 and 1984. The 1981 K20 concentration was the only value which was significantly different between ditched and unditched transects. The ditched transects showed significant differences between 1981, 1983, and 1984 in all tested values except pH and the unditched transets showed significant differences between 1981 and 1984 for only CaO.

Vegetation

There were 14 plant species found within the 10 transects (Table 2): 9 in April, 1981; 7 in July, 1981; 13 in July, 1983; and 11 in July, 1984. The dominant vegetation (percent cover > 5%) was made up of two species, perennial glasswort and spike grass. Annual glasswort was codominant in April, 1981, and July, 1983.

From July, 1981, to July, 1984, total ground cover significantly increased from 52.4 to 73.5 percent. The decrease in total ground cover from 57.2 to 52.4 percent from April, 1981, to July, 1981, was not significant. Percent cover of only 4 of the 14 species significantly changed throughout the study area: perennial glasswort, spikegrass, annual glasswort, and saltwort. Percent cover of the 14 plant species from April, 1981, to July, 1984, by transect is summarized in Table 3. Twenty of 49 species comparisons within transects showed significant differences. Spike Grass showed the greatest overall and within transect change. Perennial glasswort and annual glasswort were second and third respectively. Total cover significantly changed in eight of the ten transects.

Table 4 summarizes overall and time interval percent cover change for the three dominant species and total cover between ditched and unditched transects. There were six individual species and one total cover time interval difference, but only annual glasswort showed an overall significant difference.

DISCUSSION

The most significant soil changes came, as expected, in water soluble elements. The greatest soil changes occurred in the ditched transects; this is probably due to the greater water movement in and around the ditches. Only calcium significantly decreased in the unditched transects. It appears that the ditches themselves may speed up any soil element

Table 1. Soil values of the salt marsh study site.

Soil Parameter	Date	Transect 1	2	3	4	5	6	7	8	9	10	TOTAL	Dit.	Undit.
S. Salts	81	161	125	74	28	123	69	36	158	113	18	91	119	62
	83	14	14	10	14	16	13	14	12	23	4	14	15	12
	84	48	30	18	27	16	12	17	22	29	11	23	23	23
PH	81	6.7	6.5	7.4	8.4	7.1	6.4	6.3	7.1	7.0	7.3	7.0	7.0	7.0
	83	6.3	6.5	7.6	8.2	6.7	6.6	6.9	6.8	7.6	7.2	7.0	7.0	7.0
	84	6.7	6.6	6.9	8.0	6.7	6.6	6.6	7.4	6.9	7.0	6.9	6.9	7.0
P2O5	81	53	31	76	65	54	82	10	48	119	56	59	65	53
	83	17	18	35	108	18	16	60	17	33	19	34	24	44
	84	18	9	14	32	14	5	9	37	32	18	18	21	16
K2O	81	1928	1889	1407	491	1928	1031	723	1928	1311	289	1292	*1692	*892
	83	424	404	337	337	482	375	453	404	520	192	393	429	356
	84	547	528	413	336	395	298	365	538	480	230	413	470	355
CaO	81	5599	5599	5599	2519	5151	5599	2799	5599	5599	3415	4748	5509	3986
	83	1904	1792	2352	2408	2800	2912	2576	2072	3136	1288	2324	2430	2217
	84	2462	2016	2016	1848	2632	2184	1904	4088	3024	1736	2272	2755	2026
MgO	81	2667	2667	2667	2667	2667	2667	2667	2667	2667	2667	2667	2667	2667
	83	2667	2667	2133	2667	2667	2667	2667	2667	2667	1120	2579	2640	2358
	84	2667	2667	2531	2667	2667	2597	2667	2667	2667	2065	2586	2640	2533

Soluble salts - parts per thousand; P2O5, K2O, CaO, MgO - pounds per acre

Underlined values indicate significant differences ($P < 0.05$) among years.

Asterisk values (*) indicate significant difference ($P < 0.05$) between ditched and unditched transects.

Table 2. Percent cover over frequency of occurrence of vegetation in the salt marsh study site.

Species	Apr 81	Jul 81	Jul 83	Jul 84
Perennial glasswort (Salicornia virginica)	38.5/63	39.2/61	25.8/64	26.6/65
Spike grass (Distichlis spicata)	11.5/35	6.6/23	18.0/56	41.6/74
Annual glasswort (Salicornia biglovii)	6.4/24	5.8/28	0.2/5	2.1/13
Boneset (Eupatorium rotundifolium)	0.4/2	0.2/2	0.2/2	0.5/4
Black mangrove (Avicennia germinans)	0.2/2	0.1/2	0.6/3	0.7/3
White mangrove (Laguncularia racemosa)	0.1/1	T/T	0.1/1	0/0
Coastal dropseed (Sporobolus virginicus)	0.1/T	0/0	0.4/1	0.4/1
Sea-oxeye (Borrichia frutescens)	T/1	0/0	0.3/1	0.4/2
Saltwort (Batis maritima)	T/T	0.5/2	4.2/7	1.0/4
Sea-purslane (Sesuvium portulacastrum)	0/0	0/0	0.3/5	0.3/7
Groundsel-tree (Bacharis halimifolia)	0/0	0/0	0.1/3	0.0
Sea blite (Suaeda maritima)	0/0	0/0	T/1	0.1/1
Marsh elder (Iva frutescens)	0/0	0/0	0/0	0.2/1
Common purslane (Portulaca oleracea)	0/0	0/0	0.3/1	0/0
TOTAL PERCENT GROUND COVER	57.2	52.4	50.5	73.4

T/T - Less than 0.1 percent cover and a frequency of occurrence of 1 respectively.

Underlined values indicate significant differences ($P < 0.05$) in percent cover among years.

Table 3. Percent cover of vegetation for each transect in the salt marsh study area.

		Transect									
Species	Date	1	2	3	4	5	6	7	8	9	10
Perennial glassowrt	Apr. 81	22.6	12.9	17.1	17.7	59.8	73.9	82.2	26.7	35.0	36.7
	Jul. 81	28.2	12.5	17.2	17.6	60.0	55.5	81.6	37.1	37.4	42.2
	Jul. 83	24.1	9.4	22.4	14.2	37.0	36.0	1.9	60.0	15.5	37.4
	Jul. 84	14.8	29.7	27.0	7.3	13.8	40.4	61.7	1.1	43.2	27.0
Spike grass	Apr. 81	6.6	14.1	13.7	0.2	31.0	0.7	0.7	28.4	0.6	18.7
	Jul. 81	4.4	9.7	8.1	0.1	26.6	0.4	0.3	16.6	0.1	0.0
	Jul. 83	16.8	26.8	15.3	0.3	32.4	40.8	27.1	11.2	3.9	5.8
	Jul. 84	56.0	45.2	31.2	2.1	79.9	57.4	26.9	58.1	20.7	38.3
Annual glasswort	Apr. 81	1.4	12.2	6.9	19.3	0.0	8.4	0.6	0.0	1.5	5.5
	Jul. 81	2.8	21.3	8.6	9.4	0.0	6.8	0.8	0.0	5.1	4.7
	Jul. 83	0.0	0.4	0.2	1.3	0.0	0.0	0.0	0.0	0.0	0.2
	Jul. 84	0.0	3.2	4.2	8.4	0.0	0.0	2.6	0.0	1.7	1.4
Boneset	Apr. 81	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	3.8
	Jul. 81	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.1
	Jul. 83	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.6
	Jul. 84	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	5.2
Black mangrove	Apr. 81	0.0	0.0	2.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Jul. 81	0.0	0.0	1.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Jul. 83	0.0	0.0	6.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Jul. 84	0.0	0.0	3.7	0.0	0.0	0.0	3.0	0.0	0.0	0.1
White mangrove	Apr. 81	0.5	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0
	Jul. 81	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.0
	Jul. 83	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.2	0.1
	Jul. 84	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Coastal dropseed	Apr. 81	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.2
	Jul. 81	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Jul. 83	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	4.1
	Jul. 84	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	4.6
Sea-oxeye	Apr. 81	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0
	Jul. 81	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Jul. 83	0.0	0.0	0.0	3.1	0.0	0.0	0.0	0.0	0.0	0.0
	Jul. 84	0.0	0.0	0.0	4.2	0.0	0.0	0.0	0.0	0.0	0.0
Saltwort	Apr. 81	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1
	Jul. 81	0.0	0.0	0.0	0.1	0.0	0.0	0.0	4.4	0.0	0.1
	Jul. 83	0.0	0.0	0.0	5.4	0.0	0.1	7.8	5.7	0.0	0.5
	Jul. 84	0.0	0.0	0.0	9.6	0.0	0.2	0.1	0.0	0.0	0.5

Table 3. Continued.

Species	Date	Transect 1	2	3	4	5	6	7	8	9	10
Sea-purslane	Apr. 81	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Jul. 81	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Jul. 83	0.0	0.0	0.0	2.4	0.0	0.0	0.1	0.2	0.0	0.6
	Jul. 84	0.0	0.0	0.0	1.2	0.0	0.0	0.7	0.0	0.0	0.7
Groundsel-tree	Apr. 81	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Jul. 81	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Jul. 83	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1
	Jul. 84	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Sea blite	Apr. 81	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Jul. 81	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Jul. 83	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0
	Jul. 84	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0
Marsh elder	Apr. 81	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Jul. 81	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Jul. 83	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Jul. 84	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.6
Common purslane	Apr. 81	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Jul. 81	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Jul. 83	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.7
	Jul. 84	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
TOTAL COVER	Apr. 81	30.7	39.3	39.4	37.1	90.4	83.1	83.4	59.8	37.1	66.1
	Jul. 81	35.4	43.6	35.6	27.0	87.0	62.8	82.8	57.2	42.6	52.1
	Jul. 83	42.5	36.7	44.2	25.4	71.0	76.4	37.0	70.9	19.6	54.1
	Jul. 84	70.0	74.9	63.9	33.7	93.8	97.8	93.5	64.2	63.8	79.2

Underlined values indicate significant differences (P > 0.5) among years.

Table 4. A comparison in percent cover change in spike grass, perennial glasswort, annual glasswort, and total cover between 5 ditched and 5 unditched transects in the salt marsh study area.

	TIME INTERVAL			
SPECIES	Apr81-Jul81	Jul81-Jul83	Jul83-Jul84	Overall
Spike grass				
Ditched	-5.3	+5.7	+29.2	+9.8
Unditched	-4.3	+17.1	+17.9	+10.2
Perennial glasswort				
Ditched	+2.5	-4.0	-5.9	-2.4
Unditched	+1.0	-22.8	+7.5	-2.4
Annual glasswort				
Ditched	+2.9	-6.9	+5.2	+0.4
Unditched	-2.1	-4.6	+2.2	-1.5
TOTAL COVER				
Ditched	+0.2	-10.4	+22.6	+4.5
Unditched	-13.4	-4.9	+22.6	+3.1

(+) values indicate an increase and (-) values indicate a decrease in percent cover for the given time interval.

Underlined values indicate significant differences ($P < 0.05$) between ditched and unditched values for the given time interval.

change caused by a particular management strategy but have little overall effect. The greatest soil effect would be caused by the type of management strategy selected for a particular impoundment (e.g., OMWM vs. fresh water flooding).

The effects of the decrease in soil salinity was evidenced in the vegetation. Spike grass, less salt tolerant, increased whereas perennial and annual glasswort decreased. The increase in spike grass occurred a year following the significant decrease in soil salinity and perennial and annual glasswort indicating that there is a one to two year lag in vegetation response to a decrease in soil salinity. Table 4 indicates that spike grass responded sooner in the unditched transects than the ditched transects probably reflects the salt water flowing into the impoundment and through the ditches under the OMWM management strategy. Spike grass can probably significantly increase within one growing season with the addition of fresh water. Results from experiments in impoundment T-24-C, three miles south of the study area, also reflect the one year time frame. Rollins (1973) found that two flushings with fresh water is sufficient to decrease soil salinity for changing vegetation species in California impoundments. One season of rain appears sufficient to significantly leach soil salts from impoundments in east central Florida.

The construction of the rotary ditches had no significant effect on vegetation in the study area. There was a significant difference between ditched and unditched in total cover from April to July, 1981 (before and after ditching), but there was a slight increase (+0.2%) for the ditched transects and a decrease (-13.4%) in the unditched transects (Table 4). This indicates that the variability in the vegetation is much greater than the effects of ditching.

There were significant differences between ditched and unditched transects in vegetation changes during time intervals for all three dominant species and total cover; however, only annual glasswort exhibited significant differences between ditched and unditched transects which may be the result of its being an annual species. This indicates that the ditches have very little effect on future successional changes.

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WETLANDS RECLAMATION USING SAND-CLAY MIX FROM PHOSPHATE MINES

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ABSTRACT

In response to the need to establish wetlands following phosphate mining, the authors developed a method for incorporating a perched water table into the reclamation site. A 2:1 sand-clay mix created from mine tailings and phosphatic clay was laid down over contoured mine land and allowed to dewater. The resulting layer became functional, almost impermeable, hardpan when covered with a layer of overburden. Five plots of 1.2 hectares each were established as follows: (1) overburden, (2) overburden with hardwood swamp soil deposited in strips on gradient contours, (3) overburden with pine flatwood soil strips, (4) overburden with marsh topsoil strips, and (5) untreated sand-clay mix. Elevational gradients provided the range of floodplain zones. Woody species were planted in copses to accelerate areas of canopy closure. A permanent grid system over the test plots provided the filed base for a highly cost efficient and statistically sound monitoring program.

INTRODUCTION

Phosphate mining in several peninsular Florida counties has had great and obvious impact on the environment. Removal of overburden without regard to surface conditions has resulted in serious environmental problems which have prompted the passage of a series of control statutes. With the continuing introduction of ever more stringent regulation of land use, new approaches need to be taken to reduce long term environmental impact of transitory use of the land for mining.

Reclamation efforts for wetlands impacted by mining are being shifted from the traditional land-and-lakes approach. Instead, the goals are becoming the reestablishment of wetlands which are subject to intermittent periods of flooding and exposure. Wetlands being displaced by mining are considered to be significant in retention of water from rapid runoff and also to function as significant percolation beds in the recharge of aquifers (Anon., 1967). Further, because many of these wetlands, swamps, and riverine forests contain broad-leafed trees which create a cool air column, precipitation is enhanced. When extensive tracts such as cities have large areas barren of forest cover, localized rainfall is reduced in the region of the rising hot air column (Leopold, 1968). One can observe the paucity of wildlife habitat in recently surface mined land. Thus, the need for an effective method for wetlands restoration in a manner which provides for biological diversity and reestablishes wetlands functions in the ecosystem is clear.

Huck (1979) has demonstrated that Floridian wetlands are derived from perched water tables. These perched tables are usually over almost impervious layer a few decimeters below the soil surface. Overlaying soils saturate quickly and flood until evaporation, lateral movement of water, and slow percolation through breaks in the hardpan again dry the topsoil. Where waters are held in depressions in riverine floodplains and where surface flow is impeded, native vegetation soon gains ascendency over other taxa. Our observations over many years of botanical studies in Florida and the tropics indicate that hydroperiod is highly significant both for the establishment of adapted species and elimination of most undesirable invaders which may be allelopathic and inhibitory to desirable species. Tree ring studies and observed successional histories of vegetation on wetlands now part of the lands of the University of Central Florida show that succession in severely deforested and burned over wetlands is rapid once diaspores establish adapted species (Miller, unpublished, & Poppelton, et al., 1972).

These facts about the nature of Floridian wetlands led us to develop a long term outlook that stimulated our efforts to develop a _modus operandi_ which would approximate a natural landscape in a minimum of time. Our working hypotheses rest upon the concept that succession to desirable wetlands vegetation can be significantly accelerated by finding a proper combination of diaspores and hydrologic regime over a perched water table. Our work began in 1982 when we were asked to design a forested wetlands research project for the CF Industries phosphate mine in northern Hardee County, Florida. It is our purpose in this report to describe the project and to provide a preliminary report of our experiences. We anticipate making future reports with quantitative data derived from the monitoring program.

STUDY SITE

Approximately six hectares of unreclaimed spoil area were prepared as a demonstration suite of wetland habitats in which water levels could be regulated to simulate "normal" rainfall cycles. Contouring of the terrain was accomplished with earth-moving equipment.

Slowing percolation until a natural hardpan or similar layer develops is not a usual part of restoration technology. We viewed perching a water table near the surface as a necessary first step to establishment of wetland vegetation. Lining of farm ponds with a layer of clay is a solution widely employed for making water retaining basins but that was not the solution to our problem because desirable vegetation is slow to establish on clay even when it is kept wet.

As is well known in the industry, clay slime loses water very slowly and must be retained within dikes for many years before it is sufficiently consolidated to be considered solid earth. When mixed with various proportions of sand, however, the dewatering process is much accelerated. After some simple percolation tests, we settled on a sand-clay ratio of 2:1 as being suitable for our needs. The mix was applied to the base overburden surface from a pipeline to a depth of about two feet (6 dm) to yield a layer about a foot thick (3 dm) after dewatering. We expected the drying process to be far enough along in three months to be able to spread a layer of overburden over the sand-clay mix with the usual equipment but an exceptionally wet season delayed the spreading of the overburden (Fig. 1).

MATERIALS AND METHODS

The experimental area was divided into five 1.2 hectare sections, or plots (Fig. 2), put on gradual, slightly irregular surface sloped upward from the "streambed," which always contained some water. Each plot surface was different:

Plot 1. Untreated overburden;

Plot 2. Hardwood swamp topsoil held in reserve storage piles for several months and then spread in strips on overburden more or less along contour lines;

Plot 3. Pine flatwoods topsoil spread in strips on overburden more or less along contour lines;

Plot 4. Pan-lifted and directly transported marsh topsoil spread in strips on overburden more or less along contour lines;

Plot 5. Sand clay mix, untreated, not contoured.

Because each of the plots was the same size and of the same general orientation comparable study sites could be established on a grid with all major corners marked by high stakes. Numbering the individual elements of the grid allowed the development of a planting strategy which provided for a miximum amount of statistically significant data.

Each grid square was planted with tubelings of nursery-grown native species of trees which are commonly encountered in wetland habitats in the area. The species selected were:

1. *Acer rubrum*, red maple
2. *Carya glabra*, pignut hickory
3. *Celtis laevigata*, surgarberry
4. *Fraxinus caroliniana*, popash
5. *Gordonia lasianthus*, loblolly bay
6. *Ilex cassine*, dahoon

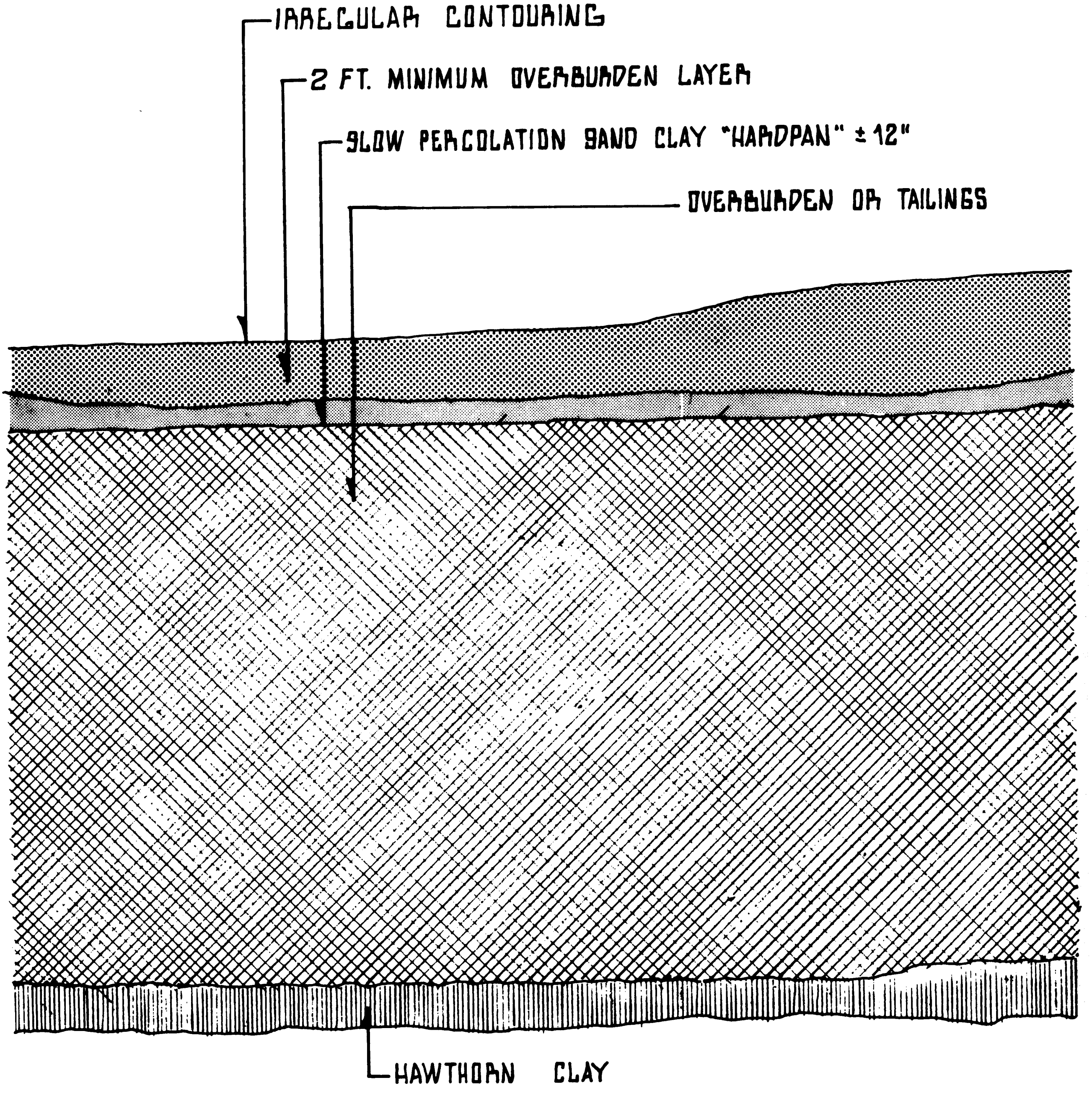

Figure 1. Typical cross-section of reclaimed land for wetland areas.

(2 ft = 6 dm; 12" = 3 dm)

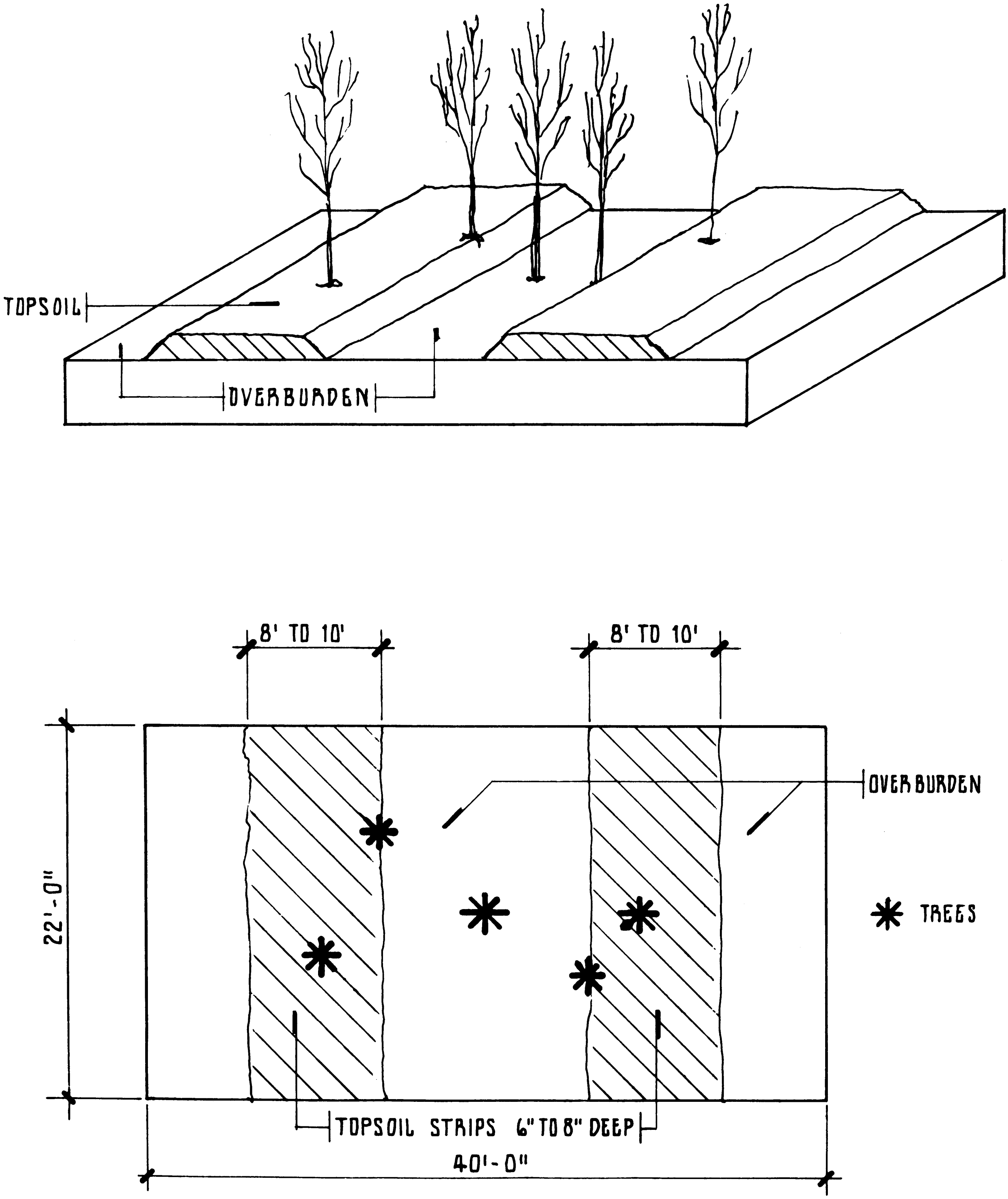

Figure 2. Typical experimental bay. (dimensions are shown in feet; 1 foot = approximately 0.305 meters)

7. Liquidambar styraciflua, sweetgum

8. Magnolia virginiana, sweetbay

9. Nyssa biflora, blackgum

10. Quercus laurifolia, laurel oak

11. Quercus nigra, water oak

DISCUSSION

The grid planting system allowed for testing success of both pure stands of a species and of mixed species plantings. The gradual changes in elevation and distance from the water course allowed for determination of optimum conditions for growth and survival of individual species upon the soil provided in the plot. A single species was planted in a row of grid units from the water's edge to the highest elevation. This created, then, 11 ribbons each with a single species. The remaining rows of grid units were then planted with species mixes in each grid unit. This would allow for observations upon possible allelopathy between or among species.

Grids are usually established for tree planting to provide for a regular spacing of the trees so that they are equidistant. Our primary goal, however, was the establishment of a functional forested wetland as rapidly as possible. Therefore, we planted small copses of five or seven closely spaced trees, about seven feet apart, so that canopy closure would occur in a reasonably short time (Fig. 3). As soon as canopy closure occurs, the environment beneath becomes shaded, more humid, and protective of wildlife inside the copse. The copses are not so widely spaced that total canopy closure will not occur, given time, and our experience working in Florida wetlands has led us to believe that the closed copse canopy will accelerate natural establishment of wetland woody species.

Whenever reclamation is undertaken, weeds tend to invade and to delay the establishment of native species. Weed invasion can be somewhat controlled by planting a mixed cover crop such as Aeschynomene and millet. In addition to weak control, penetration of cover crop roots into the soil imporves the structure and dead roots provide a good source for establishment of the soil biota. Leaf and stem litter encourages invasion of insect scavengers and soil fungi so necessary to achieve natural biogeochemical cycles. We have not undertaken to inoculate the soil beyond the transport to the site of native soils from nearby regions. The evidence to date (e.g., Godley & Callahan, 1983) is that the native soils do introduce, or provide for, establishment of species diversity. We have already noted considerable diversity in the plots where topsoils have been spread on the overburden and that the species' compositions correlate with origin of the topsoil.

We experienced the wettest season in years while waiting for the sand-clay to dry out. Subsequently, after planting, we experienced a drought which severely stressed the tubelings. Recently, the site was invaded by wild hogs and portions severely disturbed. Despite the

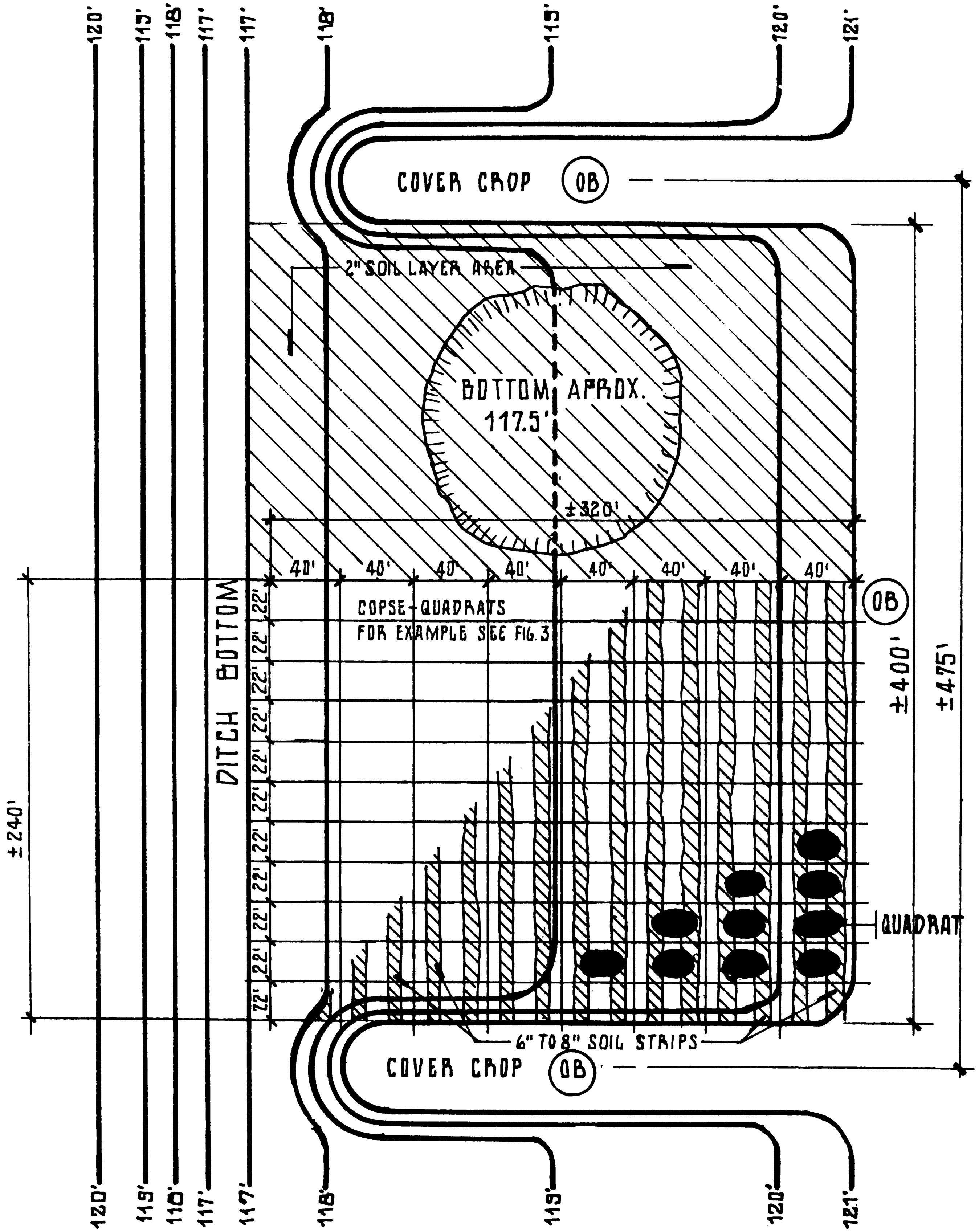

Figure 3. Typical copse quadrat. (dimensions are in English units: 6"-8" = 15-21 cm; 8'-10' = 2.4 - 3.1 m; 22' = 6.7m; 40' = 12.2m)

setbacks and the occurrence of the unexpected, our research plots have provided preliminary information and will continue to yield useful data. We have begun to monitor our experimental plots on a schedule which allows observation of seasonal changes and the progress of the establishment of biota. Analysis of the data derived from the test plots will assist in developing improvements to our basic reclamation strategy which are applicable in the future. We look forward to reporting on the progres of this project in a more quantitative way at future conferences.

In summary, we have responded to the need for a better wetlands reclamation protocol by creating the necessary perched water table. Over the perched water table we spread five different soil types along a gradient from standing water to the upper limit of high flood stage. Eleven hardwood species were tested against a variety of soil and flood conditions. Trees were planted in copses in order to more quickly establish a suitable subcanopy microclimate needed for establishment of inhabitants of wetland woodlands. Some successes with our approach have been observed and we are sure that much remains to be learned as the system matures and we develop quantitative data from the monitoring program.

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DETERMINING MICROBIAL COMMUNITY EQUILIBRIUM IN DISTURBED WETLAND ECOSYSTEMS

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ABSTRACT

There is considerable theoretical and practical interest in verifying equilibrium conditions in wetlands and other natural systems that have been disturbed or displaced by anthropogenic stress. Similarly, determination of equilibrium conditions in alternative or reclaimed ecosystems, such as wetlands created to replace terrestrial ecosystems following surface mining, is a necessity for regulation and mangement of newly created systems. An important question for alternative ecosystems is "do equilibrium conditions resemble those of ecologically similar natural systems in the same geographic area?" We have used community structure and colonization dynamics of aquatic, eukaryotic microorganisms for this purpose. Photosynthetic species such as motile algae integrate the effects of variations in water quality and are especially reflective of nutrient levels. Heterotrophic forms crop bacteria and process detrital materials and reflect the relative state of detrital processing in sediments. In studies of 21 natural and reclaimed Florida lakes, we found that microbial colonization of artificial substrates could be used to estimate recovery rates of lakes following reclamation activities. Microbial community structure differed in the youngest lakes studied, but lakes greater than two years old showed little difference from natural lakes in terms of community composition or colonization dynamics. Young lakes had high levels of aluminum and selenium and low pH. Microbial species are sensitive to unnatural disturbances such as inputs of excessive nutrients or toxic substances and can provide one type of evidence for ecosystem recovery following stress.

INTRODUCTION

A great deal of interest exists in developing wetland ecosystems of many kinds, such as small lakes, marshes, wet prairies, bogs, and other related wetlands, on sites disturbed by human activities. This interest has been primarily in response to difficulties in restoring disturbed ecosystems because of the removal of valuable subsurface materials that results in permanent, water-holding depressions. In addition, wetland acreage has been on the decline throught the history of human settlement of North America.

The purpose of the investigation reported here was to utilize microbial community dynamics as measures of disturbed ecosystem recovery. This study was carried out in the phosphate mining region of central Florida where surface mining annually results in the creation of a large number of reclaimed lakes. Examination of lakes was the focus of the research, but the design could easily be modified for use in measuring the disturbance of any wetland containing standing or flowing water.

The microbiota of aquatic systems is an attractive target for ecosystem recovery study. Microbial degradation of organic carbon accounts for a major fraction of the carbon and energy flow in most aquatic systems. Much of this activity is mediated by bacteria, but Protozoa, fungi, and micrometazoa also assist in the mineralization of carbon. For example, the vast majority of protozoan species are bacteria-detritus feeders (Pratt and Cairns, 1985). Most microbial communities develop on surfaces, and attached algae on these surfaces may account for as much as 25 percent of primary production in aquatic systems (Wetzel, 1983). We have restricted our analyses to the measurement of protozoan colonization dynamics on artificial substrates. Although we examined only a portion of the community, artificial substrate communities included species inhabiting most of the different kinds of surfaces available.

Since members of microbial communities tend to be similar all over the world and span the range of sensitivities of higher organisms, they can be used as valuable ecological indicators of ecosystem conditions. We hypothesized that conditions in developing ecosystems could be evaluated using microbial colonization of artificial substrates. We postulated that the lack of disperal of a large species pool of microbes or the presence of adverse conditions in human-influenced systems would cause aberrations in the microbial colonization process. This hypothesis had already been validated in a number of ecosystems (e.g., Cairns et al., 1979; Plafkin et al., 1980; Cairns & Henebry, 1982).

METHODS

This study was carried out in Polk County, Florida. Sixteen lakes created following phosphate mining and three natural lakes were examined. In addition, water in two active mine cuts was also sampled. Lakes of mining origin ranged in age from ~2 months to >60 years. Most lakes were <10 years old and, in general, were reclaimed under rules promulgated by the Florida Department of Natural Resources directing the reclamation of surface mined lands. Ages of lakes were obtained from reclamation records. Active mine sites, old unreclaimed lakes, and natural lakes were assigned ages of 0.1, 60, and 100 years, respectively, to simplify statistical analyses and display of data.

Microbial community dynamics were evaluated by placing polyurethane foam artificial substrates in each study lake following the method of Cairns et al. (1979). Sufficient substrates were placed to allow collection of four replicate substrates from each lake after 0.5, 1, 3, 8, and 10 days of colonization. Each collected substrate was placed in a sterile collecting bag and returned immediately to a field laboratory where the substrate was squeezed to remove its contents. The number of species colonizing the substrate after a given exposure time was determined by repeatedly subsampling the substrate contents until an asymptotic number of species had been identified. Colonizing protozoan species were identified using standard protozoological references.

In addition to biological collections, a variety of physical and chemical measurements was made in each system on each sampling day. Standard physical and chemical measurements (temperature, pH, conductivity, dissolved oxygen, hardness) were supported by measurements of nitrogen, phosphorus, and sulphur nutrients; organic carbon; and trace metals.

Colonization data were fitted to the MacArthur-Wilson (1967) equilibrium model:

$$S_t = S_{eq}(1-e^{-Gt}),$$

using nonlinear least square regression. Using the number of species (S) at any given time (t), it is possible to estimate the equilibrium species number (S_{eq}) and the colonization rate (G). Equilibrium species number and colonization rate may be used as derived variables assigning a single representative number to each study lake for use in further analysis. Simple correlation procedures were used to relate biological and physicochemical factors to lake age.

RESULTS

Several factors differed among study lakes. Certain chemical factors showed some relationship to lake age and were useful for tracking the recovery process. Newly reclaimed lakes had extremely low pH (Fig. 1). In addition, levels of trace metals were elevated in the newly reclaimed systems. Aluminum levels were quite high (Fig. 2), and selenium levels were high in both newly reclaimed lakes and active mine sites (Fig. 3). Selenium levels approached federal chronic standards for water quality and Florida class III standards (e.g., Fernald & Patton, 1984). Low pH appears to be the result of pyrite oxidation (Boody et al., 1984). Filling of reclaimed mine sites with rainwater, followed by acid-producing oxidation processes, probably resulted in the solubilization of aluminum. With the exception of elevated metal levels in newly reclaimed systems, trace metals generally were in the range of trace metals in natural lakes.

Planting undertaken to stabilize regraded banks around newly reclaimed lakes was related to high nitrate levels in the newly reclaimed systems (Fig. 4), presumably from inputs of fertilizers in runoff. We also noted elevated levels of sulfate and fluoride in some systems, but no consistent pattern differentiated lakes. Natural lakes sampled concurrently had higher levels of potassium than reclaimed lakes.

Evaluation of protozoan colonization dynamics showed that newly reclaimed lakes had comparatively simple protozoan communities. Colonization in the most recently reclaimed lakes was much lower in terms of estimated equilibrium species number than that found in older reclaimed, unrecliamed, and natural lakes (Fig. 5). Colonization in systems other than those most recently reclaimed was very rapid and extensive. Equilibrium species numbers were much higher than those reported for a variety of other lakes (e.g., Plafkin et al., 1980). Similarly, evaluation of the total number of different species sampled from the various lakes demonstrated a similar pattern: those lakes most recently disturbed had lower

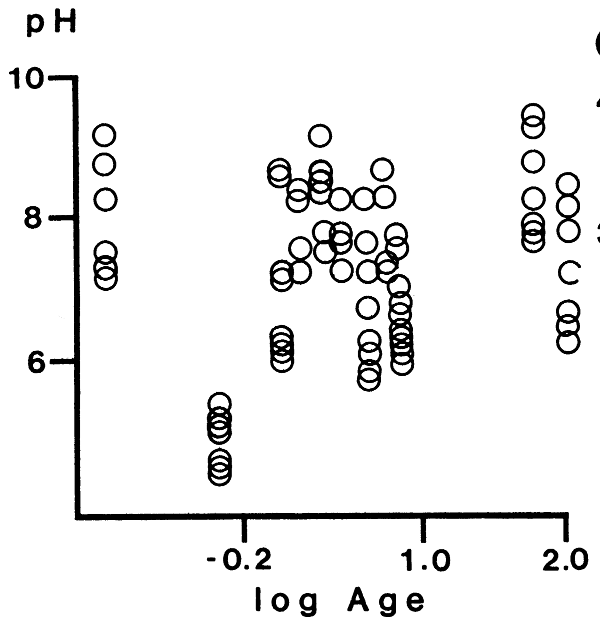

Fig. 1. Relationship of pH to the logarithm of age of study lakes.

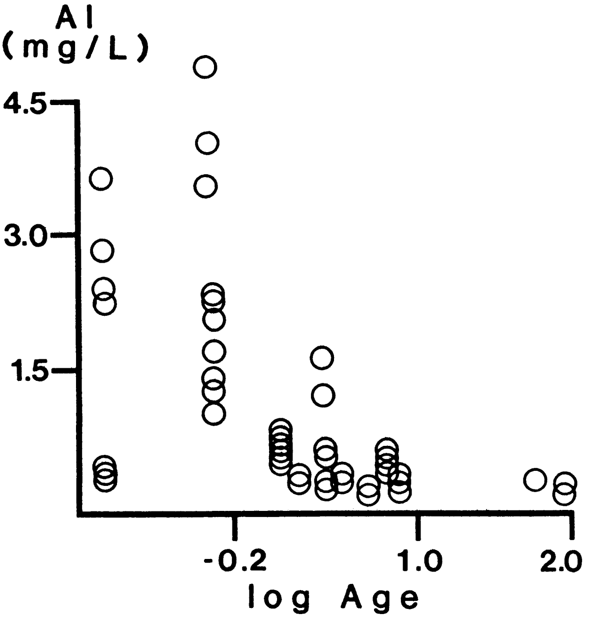

Fig. 2. Relationship of aluminum and lake age in study lakes.

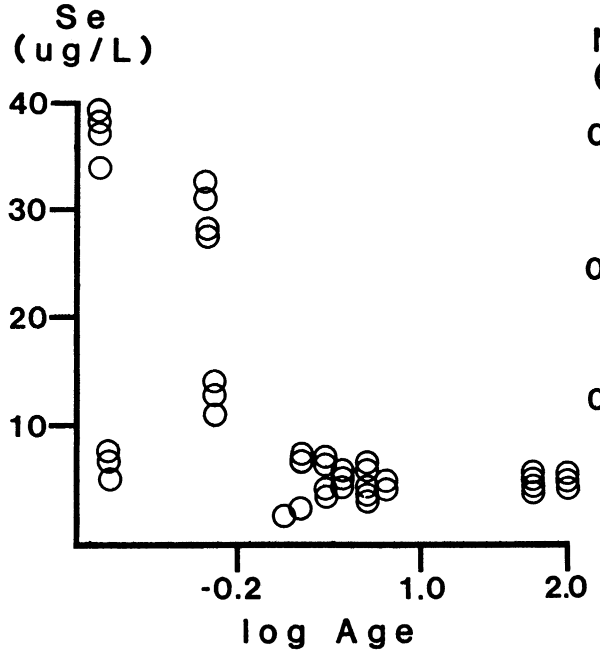

Fig. 3. Relationship of selenium and lake age in study lakes.

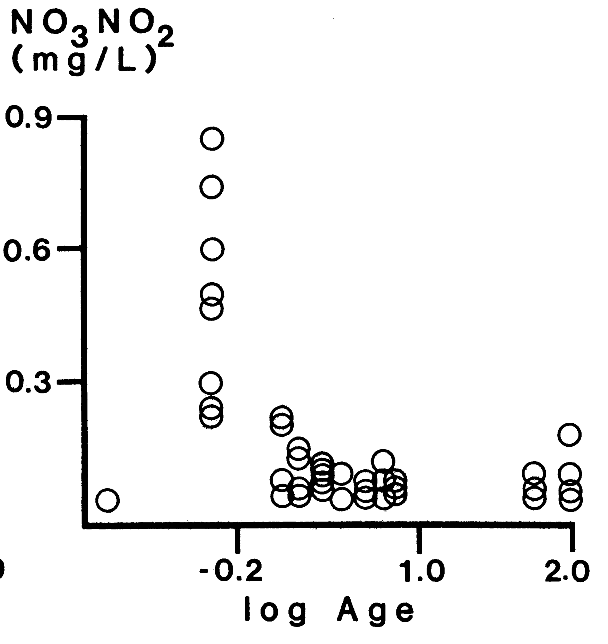

Fig. 4. Relationship of nitrate (as N) and lake age in study lakes.

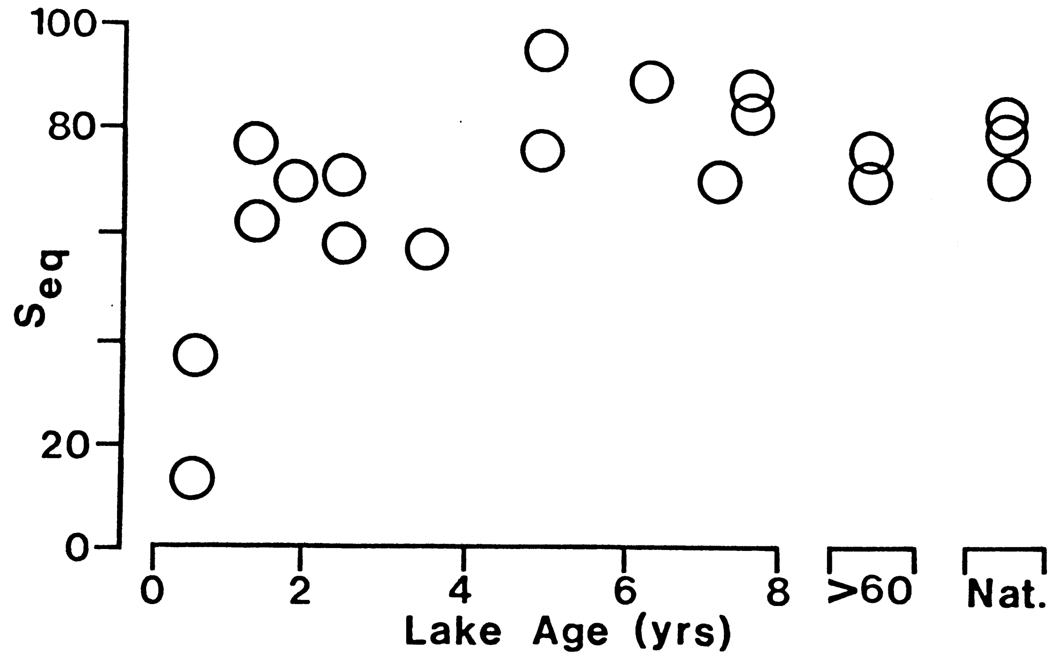

Fig. 5. Change in equilibrium species number based on protozoan colonization of artificial substrates with lake age.

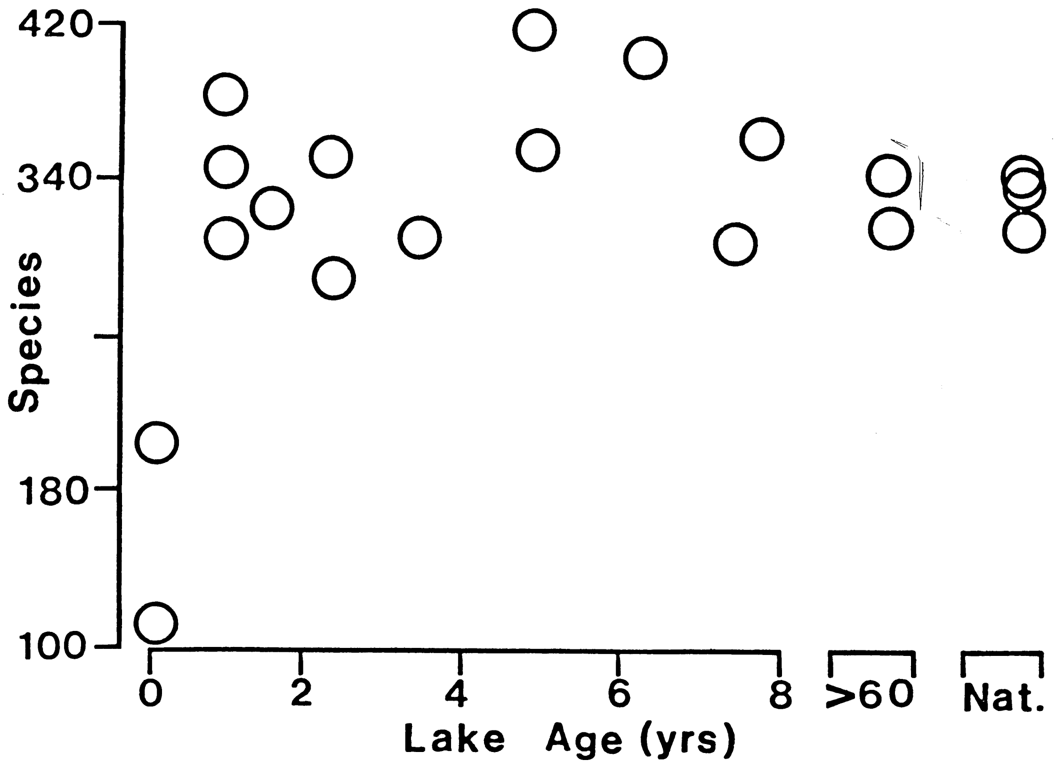

Fig. 6. Change in total protozoan species sampled (an estimate of the total species pool) with lake age.

numbers of total species than other lakes (Fig. 6).

Equilibrium species numbers were significantly correlated with differences in various chemical parameters. In addition to the obviously unusual metal and pH levels in newly reclaimed lakes, equilibrium species numbers were significantly related to carbon to nitrogen (C:N) ratio (Table 1). Total organic carbon levels tended to be somewhat lower in the youngest systems, and, as mentioned previously, nitrate levels were elevated in newly reclaimed lakes.

Table 1. Correlation of chemical factors and equilibrium species (S_{eq}) number in study lakes, summer, 1984.

Factor	r	Significance
Aluminum	-0.68	$p<0.001$
Selenium	-0.51	<0.001
Nitrate-nitrite	-0.51	<0.001
Carbon:nitrogen ratio	0.52	<0.001

Space does not permit further elaboration of results. However, it should be noted that a variety of other measures might be useful in examining a large data set. We used a variety of multivariate procedures to analyze further and more specifically the data set. It is also possible to examine the biological structure in terms of functional groups, as mentioned previously. These analyses can serve to test interpretations based on the simpler relationships among lake ages, chemical composition, and number of species. Such differences are discussed briefly below.

DISCUSSION

Reclamation activities undertaken in the phosphate mining region of central Florida are essentially rehabilitation measures (sensu Magnuson et al., 1980). No attempt is made to restore the original ecosystems, and reclamation laws direct industries to develop ecosystems capable of supporting fish and wildlife. This necessitates both rehabilitation of uplands and the management of water bodies created in depressions remaining after regrading the disturbed mine areas. Based on our analyses, some of which have not been presented here, microbial community structure and dynamics in reclaimed systems fall within the range of natural lakes and old unreclaimed lakes after ~2 years of aging. This apparent recovery corresponds to the loss of toxic materials from the water columns of the reclaimed lakes.

Measurement of colonization dynamics is one way of evaluating the structure and function of communities within the reclaimed ecosystems. Other studies (e.g., Boody et al., 1984) have evaluated other functional parameters such as primary productivity. Other data that might relate to processes within the systems are lacking. Our inference of recovery is based on knowledge that colonization dynamics for protozoans are sensitive to nutrients loads (Cairns et al., 1979; Pratt, 1984), to toxic inputs (Niederlehner et al., 1985), and primary production (Henebry & Cairns, 1984). Direct measurements of other functional processes, such as flows of nutrients or carbon, would provide valuable additional information useful for comparing lakes of different types. Since this study has shown that newly reclaimed lakes apparently recover quite rapidly, a suite of structural and functional measurements might suffice as a measure of adequate recovery of the system providing a time span of two to three years is allowed for communities to develop in these newly created systems.

It should be noted that reclamation laws, such as those directly regulating the phosphate industry in central Florida and laws with similar puposes such as the federal Surface Mining Control and Reclamation Act (SMCRA) regulating coal mining operations, are relatively new governmental experiments in mandating the creation of alternative ecosystems. Unlike the reclamation laws of Florida, SMCRA requires that land use patterns be retained post-mining. Florida regulations provide much more latitude in the construction of different types of ecosystems and encourage experimental manipulation of reclaimed systems. This may relate to the rapidity with which the Florida ecosystems recover. For example, Friederich (1975) estimated that pH in a lake formed in a reclaimed coal mine did not return to normal for ~10 years. Despite the latitude available to industry reclamation officers in Florida, changing reclamation rules have made comparison of reclaimed systems difficult. Reclamation rules specify the slope of the littoral zone. Since most newly reclaimed lakes are small (<50 ha), many lakes may have areas of shallow water (<1 meter) representing nearly half the surface area of the reclaimed lake. We noted in our work that the newer reclaimed lakes were all surrounded by dense macrophytic vegetation (chiefly *Typha*), while older lakes with steeper littoral zones were much more variable in their macrophytes. It is unclear how these newer lakes will age.

We have applied several other analytic techniques to our larger data set in an effort to understand which biological and physico-chemical factors differentiate younger and older lakes. Community analyses using a variety of multivariate procedures have demonstrated essentially the same pattern elucidated by the colonization analysis. That is, community composition among the reclaimed lakes older than two years generally falls within the range of natural and older reclaimed lakes.

Analyses of protozoan communities can provide valuable information concerning ecosystem conditions. For a healthy protozoan community to develop, nutrient conditions must be suitable to support photosynthetic flagellates. An adequate bacterial-detrital mileu must be present to support the vast majority of protozoan species that exploit this resource. Primary production and detrital processing are essential processes in every ecosystem, and an experienced investigator can easily distinguish a community with unusual composition and colonization dynamics from one with a comparatively normal system. It is useful to have natural systems available for comparison since geographic, geologic, and water quality variations often preclude *a priori* understanding of systems in a particular geographical area. We were surprised at the rapidity of colonization and the large number of species we observed at equilibrium. Although these experiments may not follow the traditional methods for measuring the processes these organisms mediate, colonization analyses provide a functional measure of community response to the availability of new habitat. Structural measurements provide an opportunity to compare systems for their resident species. Protozoan communities typically have a core of common species (Yongue & Cairns, 1971; Pratt et al., in press). It should be noted, however, that species turnover is comparatively rapid in the Protozoa (Schoener, 1983). Simple analyses of colonization rate and comparisons of community similarity are relatively inexpensive and can be performed quite rapidly.

ACKNOWLEDGEMENTS

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WETLANDS CREATION TECHNIQUES FOR HEAVY CONSTRUCTION EQUIPMENT

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ABSTRACT

An upland site was excavated in October/November, 1984, to create four wetlands totalling 40 hectares adjacent to a natural slough in Desoto County, Florida. Earth-moving equipment (Caterpillar 631 Wheel Tractor-Scrapers and D8 and D9 Bulldozers) was used to excavate the site and haul spoil to a deposit area. Two percent of the return trips from the spoil deposit area were diverted to remove and transport soils and plant material from nearby isolated marshes. The soil from these donor marshes was spread in each of the new wetlands as a mulch and source of propagation material for wetland plant species. One of the four wetlands, comprising 20 hectares, was chosen as the experimental wetland and divided into three areas. Two areas were mulched, each with soil from a different type of donor marsh. The third area was left unmulched as a control. The experimental wetland was subsequently flooded with well water; the other wetlands received no irrigation. Establishment of vegetation in the flooded wetland has been initially successful in the mulched areas, but not successful in the unmulched area. No significant germination has occurred to date in unflooded areas because of the lack of precipitation since construction. Using heavy construction equipment proved successful and economical both for removing mulch from donor wetlands and for spreading it on the excavated site.

INTRODUCTION

This paper is a preliminary report on construction and vegetation techniques for wetland creation, and the results of the first six months of an experimental project utilizing those techniques. The study project was initiated in October, 1984, with the excavation and vegetation of four new wetlands totalling 40 hectares (100 acres), on a 120-hectare (300-acre) study site in Desoto County, Florida.

The new wetlands were designed to simulate the shape and seasonal hydroperiod of natural wetlands in the area, based on surveys of ground and normal wet season water surface elevations for existing wetlands in the vicinity of the study site. Each new wetland system was provided with a control structure to regulate the overflow level at the outfall point leading to an adjacent natural slough system.

Vegetation of the new wetlands was accomplished by the application of "mulch" consisting of soils and plant material removed from donor wetlands. At the time of this writing, the study site has yet to receive wet season rains; however, one of the excavated areas received water from a flowing artesian well. We have focused our study on this area (referred to as the "experimental wetland"), because the presence of water has allowed the development of a wetland community in spite of a particularly dry period. The other excavated areas remained dry, with the water table well below the ground surface. If expected seasonal rains occur during June through October, 1985, normal water table elevations will flood all of the new wetland areas, and supplemental irrigation from the artesian well should not be required.

The experimental wetland is approximately 20 hectares, created by the excavation of approximately 180,000 cubic meters (235,000 cubic yards) of soil. A 30.5 meter (100 foot) broad littoral zone slopes at a 1 percent gradient to a deeper central zone. With average wet season water surface elevations of 9.9 m (32.5 feet) above mean sea level, the central zone will have a water depth of approximately 50 cm (20 inches).

During the construction, two different kinds of mulching treatment were applied over separate portions of the experimental wetland. A third portion was left unmulched as a control, but was lightly seeded with rice (<u>Oryza sativa</u>). This paper presents preliminary data comparing the mulching treatments, as well as describing the construction techniques. The remaining three excavated areas, totalling 20 hectares, also received varying mulching treatments and will be the subject of later investigations after normal wet season water levels have occurred.

SITE DESCRIPTION

The site is located on both sides of a natural slough in Desoto County, 19 kilometers (12 miles) southwest of Arcadia, Florida. The site is relatively undisturbed and the water table has not been lowered by drainage alterations. Wetlands occur adjacent to the slough and overflow into the slough during periods of high water. Figure 1 depicts the study site prior to construction.

Soils adjacent to the slough are primarily Oldsmar Fine Sand. Oldsmar is a series with an aquic moisture regime. The water table is within 25 cm (10 inches) of the surface from one to three months of the year, and below 100 cm (40 inches) only during extended dry periods.

Most of the excavated site was a pine-palmetto (<u>Pinus elliottii-Serenoa repens</u>) community, but a portion was improved pasture. The pine-palmetto community included a substantial amount of tarflower (<u>Befaria racemosa</u>) and gallberry (<u>Ilex glabra</u>). The pasture was predominantly Bahia grass (<u>Paspalum notatum</u>).

Figure 1: Aerial photograph of the study site prior to wetland construction.

CONSTRUCTION METHODS

Construction occurred in October-November, 1984. The area to be excavated was carefully staked and flagged before construction machinery began operating on the site. Haul roads and spoil areas were designated to keep haul time and environmental damage to a minimum. Supervisors accompanied bulldozer operators as the clearing of vegetation began around the perimeter of each site. The sites were cleared and prepared for the use of Caterpillar 631 Wheel Tractor-Scrapers. The Caterpillar 631's moved the overburden material from the excavation site to the spoil disposal area. Caterpillar D8 and D9 bulldozers were used to push the scrapers in soft or wet material. The scrapers were fitted with oversize tires to allow them to work more easily in soft soils.

On return trips from the spoil disposal site to the excavation sites, some of the scrapers were diverted to isolated wetlands which were used as a source of mulching material. Two wetland types were used as donor sources: (1) shallow, flat wetlands dominated by St. John's wort (*Hypericum fasciculatum*); and (2) marshes dominated by pickerelweed (*Pontedaria lanceolata*). These donor wetlands were completely isolated from the slough drainage and all other surface waters.

In the St. John's wort marsh, scrapers pushed with the help of bulldozers removed the dark A horizon of the wetland soils, which is only about 15 cm (6 inches) deep. The scaped material was hauled to areas which had been excavated to final grade, where it was dropped and spread with bulldozers.

Backhoes were used to load other scrapers with organic soils from the pickerelweed marsh. The backhoe excavated to the point at which the soils became mineral, a depth ranging from 50 to 150 cm. The scrapers brought in clean fill behind the backhoe to provide a solid haul road as the backhoe worked across the marsh. The organic soil was hauled to excavated areas ready for mulching, where it was spread with bulldozers.

EXPERIMENTAL METHODS

The experimental wetland received three mulching treatments on separate areas, each exceeding five hectares: (1) mulching with A horizon soils from St. John's wort marsh; (2) mulching with deep organic soils from pickerelweed marsh, and (3) no mulching, but light seeding with rice.

After the mulch was spread as evenly as possible to an average depth of 10 cm (4 inches) over areas with an expected water depth of 0 to 50 cm (0 to 20 inches) during maximum flooding, water from an artesian well was diverted to the wetland. The well water was tested, and chloride and dissolved solds concentrations were found to be well within acceptable ranges for supporting plant life, even after some concentration by evaporation. Flow from the well was not sufficient to immediately flood the area, nor was it sufficient to raise water levels to the control level of the outfall structure. It did, however, provide enough water to cover over 90 percent of the bottom. None of the mulch became completely dry during the interim between spreading and flooding. Figure 2 depicts the site following flooding of the experimental wetland.

In April, 1985, five months after completion of the construction, line transect sampling was conducted in each of the treatment areas. Thirty-meter transects were laid out from above the water line toward the deeper water in the center. Transect Line 1 was established in the area mulched with organic soils from the pickerelweed marsh, Transect Line 2 in the area mulched with A horizon soils from the St. John's wort marsh, and Transect Line 3 in the unmulched control area. Plants were counted if their above-ground parts intersected the transect line, and identified to species whenever possible. Several grasses were not sufficiently mature for identification.

RESULTS

The use of Caterpillar 631 Wheel Tractor-Scrapers proved practical and efficient for removing the A horizon soil and plant material from shallow, flat wetlands with little or no organic accumulation. Scrapers were able to move into the wetland unloaded, stand momentarily, pick up soil with the help of a bulldozer, and continue moving out of the wetland once loaded. Virtually no time was lost for mired-down scrapers.

The donor St. John's wort marsh will recover, but the composition after recovery is an open question at this point. Rows of vegetation were left between scraped areas consisting of exposed mineral soils that are about 15 cm (6 inches) lower than adjacent undisturbed vegetation. Due to the lack of precipitation, recovery to date is not indicative of the wetland vegetation expected to colonize with the resumption of normal wet season water levels. Although it is unlikely that St. John's wort will dominate the colonizing vegetation, the irregular substrate and water depth may encourage a greater diversity of plant species. Since the organic soils from the pickerelweed marsh were completely removed to create the new wetland, the donor marsh is not expected to recover.

Of the two kinds of mulching material used, that from the St. John's wort marsh was easier to spread. Since it was primarily sand, it was easily worked with the bulldozers and could be spread more thinly than the heavy organic soils from the pickerelweed marsh. The pickerelweed marsh, however, produced six times more soil per unit area than did the St. John's wort marsh because of the greater average depth of excavation.

Vegetation of the mulched experimental wetland has been initially successful. The transects are graphically depicted in Figures 3-5. Transect data show that the mulched sites supported similar diverse communities of wetland plants, in spite of the marked differences between their donor wetlands. Transect 2, for instance, was dominated by pickerelweed in spite of the mulch being from the St. John's wort marsh. In the mulched areas, composition may be more closely correlated to current conditions such as water depth than to mulch source. The deep organic soil from the pickerelweed marsh appears to be just as viable a mulch source as the A horizon soil from the St. John's wort marsh, even though the proportion of live plant material was greater in the latter.

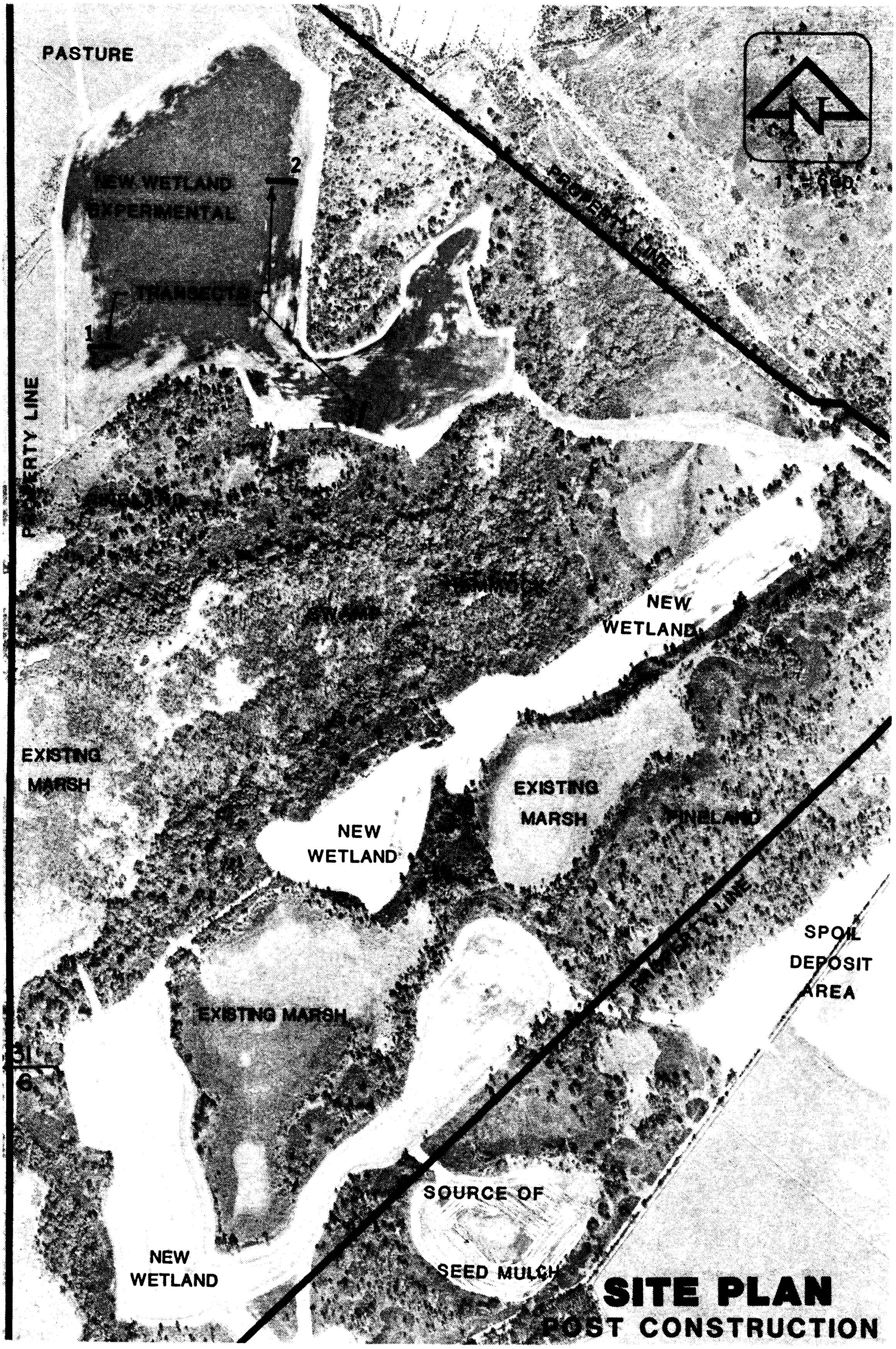
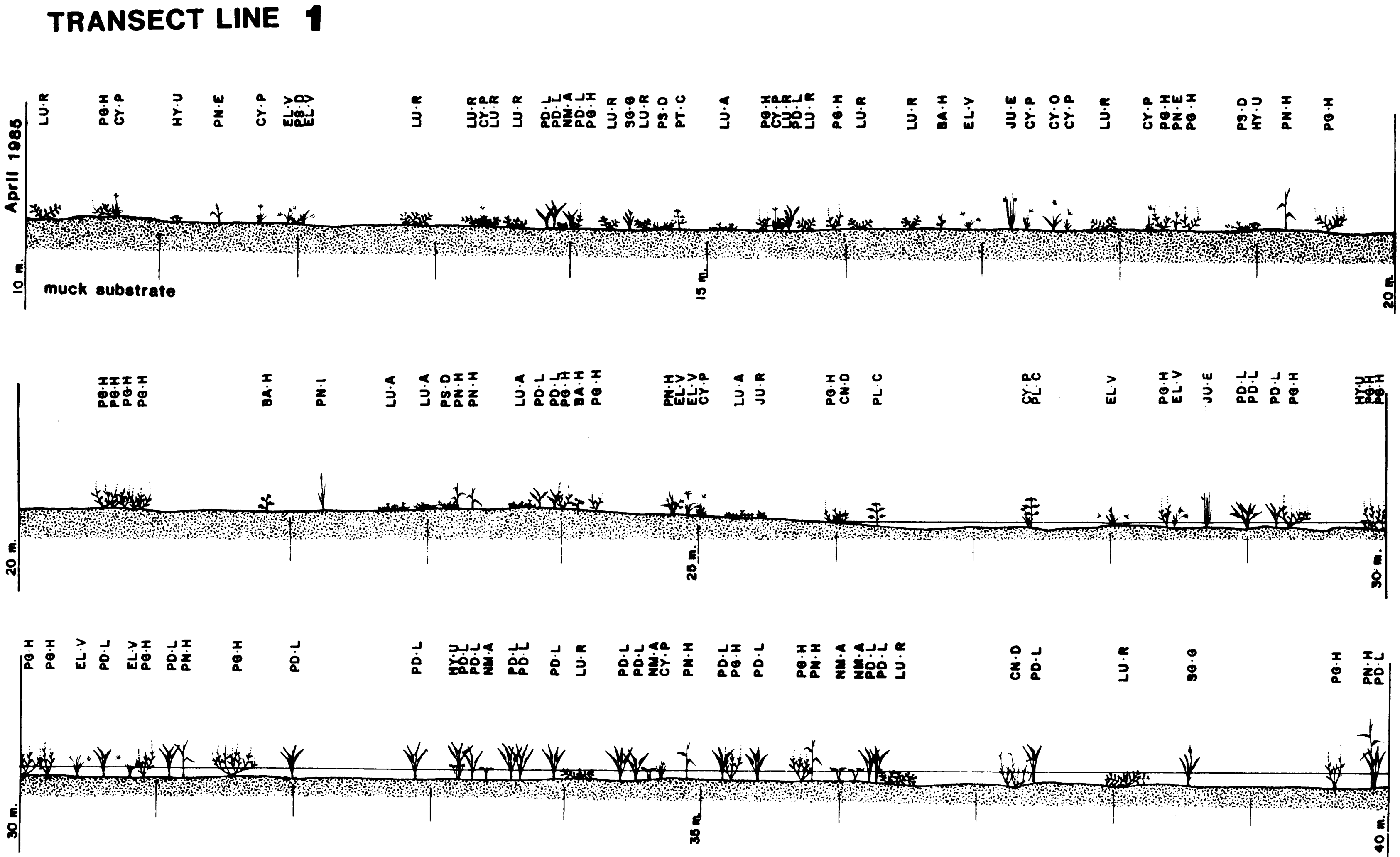


Figure 2: Aerial photograph of the study site after construction of wetlands.

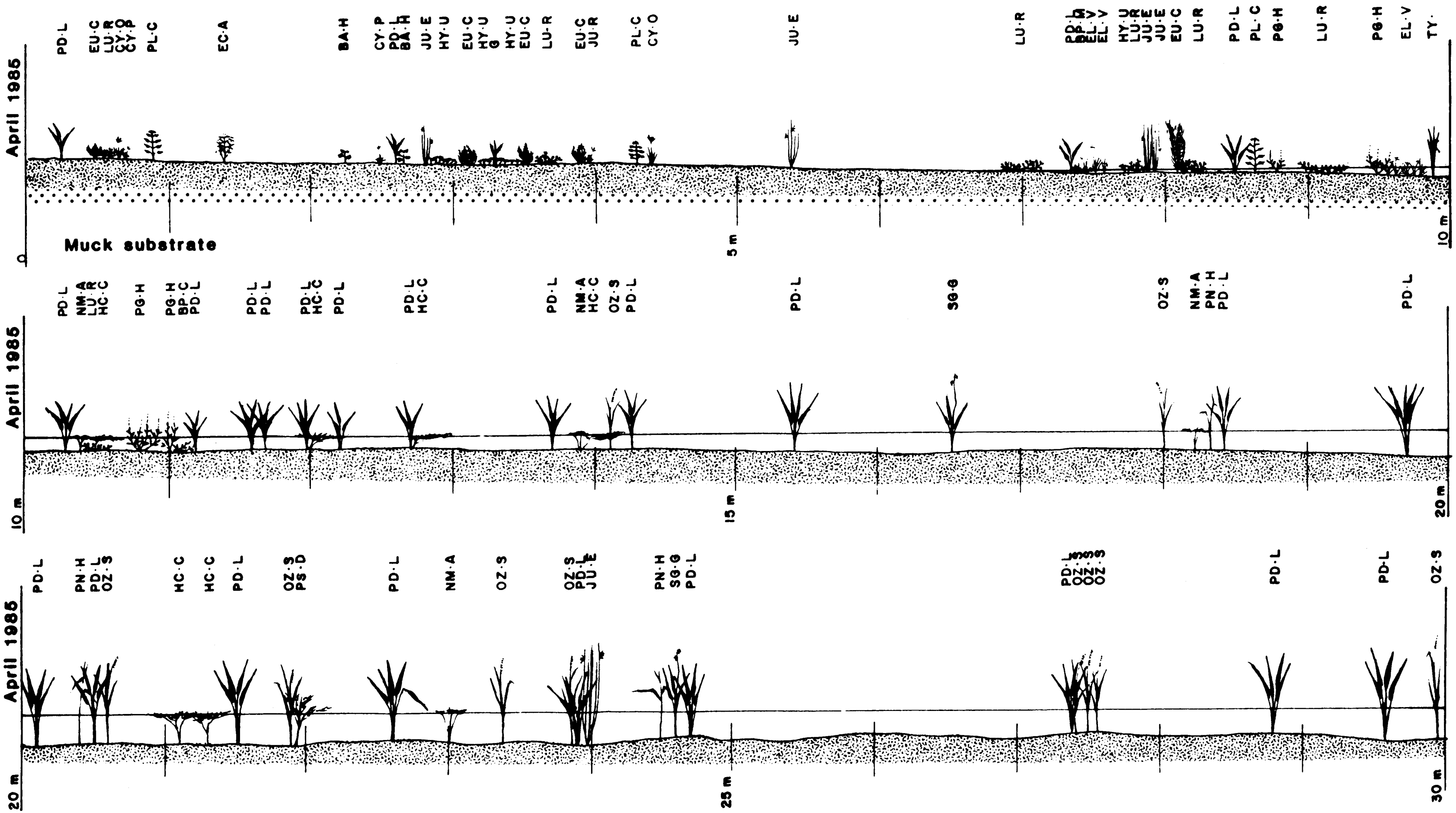
TRANSECT LINE 1



BA-H	*Baccharis halimifolia*	HY-U	*Hydrocotyle umbellata*	NM-A	*Nymphoides aquatica*	PN-H	*Panicum hemitomon*
CN-D	*Cynodon dactylon*	JU-E	*Juncus effusus*	PD-L	*Pontederia lanceolata*	PN-L	*Panicum lancearium*
CY-O	*Cyperus odoratus*	JU-R	*Juncus repens*	PG-H	*Polygonum hydropiperoides*	PS-D	*Paspalum dissectum*
CY-P	*Cyperus polystachyos*	LU-A	*Ludwigia arcuata*	PL-C	*Pluchea camphorata*	PT-C	*Ptilimnium capillaceum*
EL-V	*Eleocharis vivipara*	LU-R	*Ludwigia repens*	PN-E	*Panicum erectifolium*	SG-G	*Sagittaria graminea*

Figure 3: Profile drawing of Transect 1 through area mulched from a pickerelweed marsh.

TRANSECT LINE 2



BA-H	*Baccharis halimifolia*
BP-C	*Bacopa caroliniana*
BP-M	*Bacopa monnieri*
CY-O	*Cyperus odoratus*
CY-P	*Cyperus polystachyos*
CE-A	*Eclipta alba*
EL-V	*Eleocharis vivipara*
EU-C	*Eupatorium capillifolium*
G	unidentified grass
HC-C	*Hydrochloa caroliniensis*
HY-U	*Hydrocotyle umbellata*
JU-E	*Juncus effusus*
JU-R	*Juncus repens*
LU-R	*Ludwigia repens*
NM-A	*Nymphoides aquatica*
OZ-S	*Oryza-sativa*
PD-L	*Pontederia lanceolata*
PG-H	*Polygonum hydropiperoides*
PL-C	*Pluchea camphorata*
PN-H	*Panicum hemitomon*
PS-O	*Paspalum dissectum*
SG-G	*Sagittaria graminea*
TY-	*Typha sp.*

Figure 4: Profile drawing of Transect 2 through area mulched from a St. John's wort marsh.

TRANSECT LINE 3

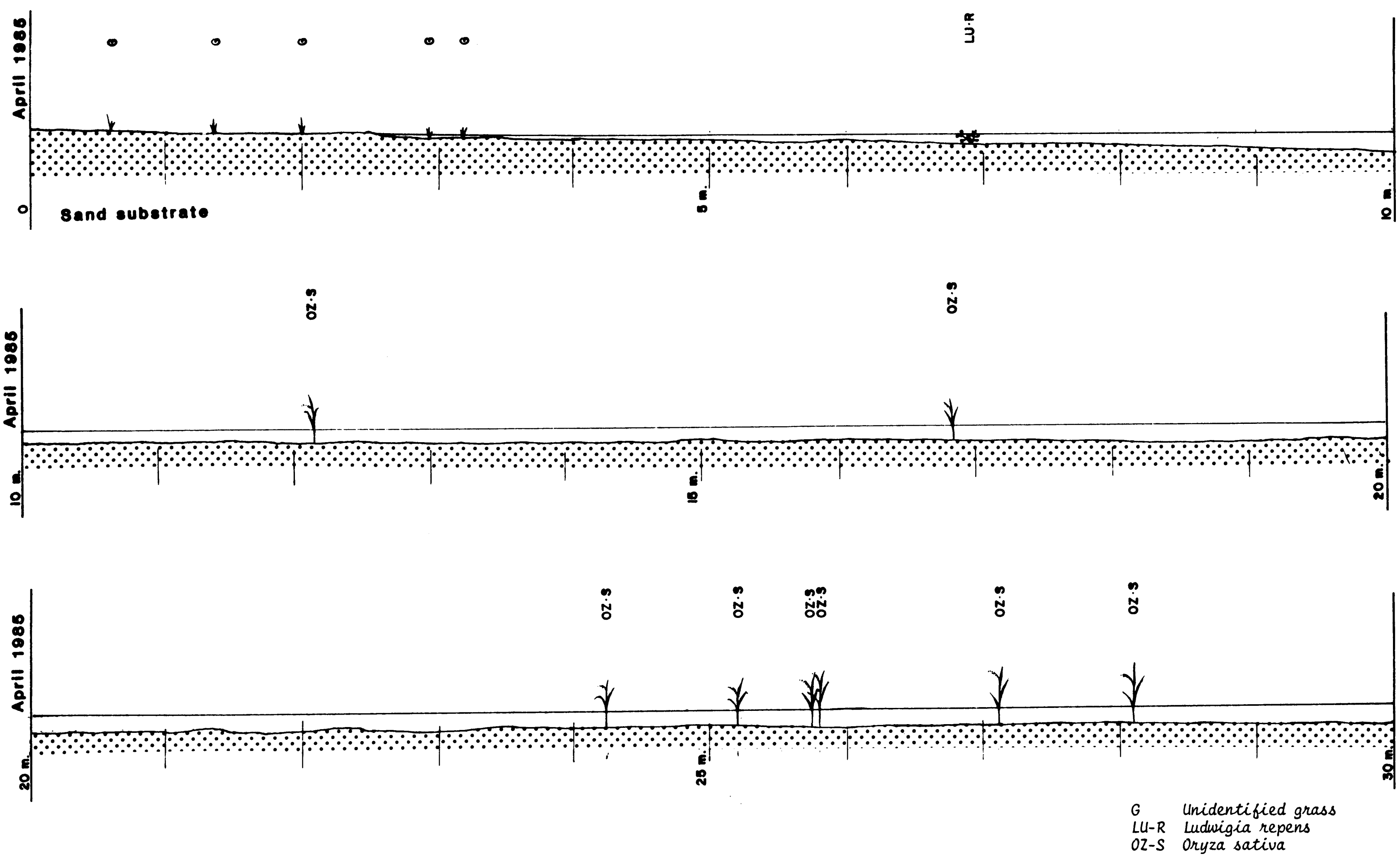


Figure 5: Profile drawing of Transect 3 through an unmulched area seeded with rice.

The results from the mulched sites (Transects 1 and 2) are compared to the control area which received no mulch (Transect 3). The transects are graphically depicted in Figures 3-5. The mulched sites supported diverse communities of wetland plants, but did not resemble their respective donor wetlands. Transect 2, for instance, was dominated by pickerelweed in spite of the mulch being from the St. John's wort marsh. The control area supported only sparse vegetation, dominated by rice which had been planted during construction. The immediate disparity in plant biomass and diversity between mulched and unmulched areas strongly suggest the value of the mulching technique. We caution the reader that these data represent a transitional community only five months after construction. For that reason, we have not attempted to draw inferences about a community composition and diversity. After a longer recovery period, we should be better able to quantify the results of these techniques.

DISCUSSION

The need for wetland creation techniques compatible with heavy construction operations is growing. High labor costs limit the practicability of many techniques which otherwise produce good results. Further, the opportunity for wetland restoration and creation frequently occurs in conjunction with heavy construction or land development operations. The wetland vegetation technique reported in this paper was made economically feasible by integrating it with the excavation work. By using the earth-moving equipment to load and haul mulch on their return trips from the spoil deposit area, the incremental cost of wetland vegetation over the cost of excavation was minimal.

The cost of return-trip mulching was very small partly because of the minimal amount of time added to the roundtrip haul time, but more importantly because only 2 percent of the hauls were diverted for mulch. This percentage was based on donor size and location, and provided only partial coverage in the other three new wetlands. A 3 percent diversion of hauls would have provided adequate mulch for the entire 40-hectare project. This would not be a 3 percent increase in construction cost, but rather an incremental cost on 3 percent of the construction.

In addition to the applicability of this technique to wetland creation or mitigation required by regulatory action, these small incremental costs would make mulching of new stormwater retention ponds feasible as well, provided donor wetlands are available. The preliminary results indicate that there is little, if any, difference in result from the two types of mulch used. If this indication holds true and is corroborated by other projects, the need for mulch could be more readily satisfied. Since haul distance is the largest cost factor, whether hauling earth or mulch, it is critical to the economy of the mulching project that donor wetlands be located close to the haul line between the excavation site and the spoil disposal site.

The extent to which damage or destruction of donor wetlands affects their suitability as donor sources depends on the value of these wetlands, their status with regulatory agencies, and their ultimate fate. There are many construction projects in which certain wetlands are substantially altered or removed. These wetlands are obviously excellent candidates for donor wetlands. The donor wetlands used in this project, for example, will ultimately be removed for future development. With proper permits from appropriate regulatory agencies, other wetlands could also be used, provided that a good balance is reached between the benefit to the created wetlands and the impact to the donor.

This report is preliminary and the study incomplete. We are following the recovery of the St. John's wort marsh that was scraped for mulch. The three new wetlands that had not been flooded as of June, 1985, will provide some insight into the viability of mulch which has been subject to extreme drying for several months.

Also, in these wetlands, we have been experimenting with a "mulch-stretching" technique, which consists of applying the organic soils, which are difficult to spread, in strips alternating with strips of bare mineral soils. This technique requires no further spreading. We hope to evaluate the efficacy of this technique for inoculating the excavated site with wetland plants, using only one-half the mulch required for full coverage. Monitoring and transect sampling will continue at regular intervals in both the initial experimental wetland and the other three. Additional data should allow more definite conclusions as to the relative success and feasibility of different mulch types, spreading techniques, and irrigation regimes, as well as the long-term viability of the project.

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