KELLY, M.

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the role of aquatic plants in florida's lakes and rivers

KELLY, M.

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THE ROLE OF AQUATIC PLANTS IN FLORIDA'S LAKES AND RIVERS

Proceedings of the
Second Annual Meeting

FLORIDA LAKE MANAGEMENT SOCIETY

September 26-27, 1990
Orlando, Florida

Edited By
Martin Kelly, Editor FLMS
September, 1991

FORWARD

"I'm going to speak my mind, because I have nothing to lose." - S.I. Hayakawa

I've just pasted Brad Cook's last TSI equations on the bottom of page 29. There ought to be some kind of editor's rule on how many equations someone can stick in a single table. What the heck am I talking about you're probably asking. I just thought as editor of this historic, "collector's edition" proceedings of the FLMS 2nd Annual Meeting, it was my duty to share some thoughts with you.

Let me digress briefly. Look at the inside margin of this page -notice the dashed vertical line? Well, if you don't think these comments have any place in a proceedings of this calibre just cut along the dotted line.

This is a collector's edition. I hope it's not like my copy of Fossil Magazine, the first volume was the last. I guess that makes it more valuable, but I paid for an annual subscription and only got one issue before the magazine folded. I hope the FLMS has a proceedings next year and the year after that and the year after that and ..... Well, you get my drift.

I remember the first volume of Limnology and Oceanography (issue #2, I don't think I saw issue #1). Well, anyway, some guy named Odum had authored a paper about how to measure primary productivity using diurnal fluxes in dissolved oxygen. Another guy named Verduin talked about primary production in lakes, while some other fellow named Ryther explained how to measure it. That was about 35 years ago. A few of the guys with papers in that near inaugural edition went on to make it kind of big in limnology circles. They've done a lot of research, published a lot of papers and sent a few graduate students on their way. I wonder what contributions the authors of papers in our humble proceedings will make. While our proceedings may not be another L&O, I'm hopeful that the authors in this issue and the co-workers cited will be major players in Florida limnology; helping us to understand, protect and preserve our aquatic resources.

It has almost been fun putting this Proceedings together. I've gained some respect for real editors. This is where I make the point that these Proceedings were only edited, not peer reviewed. This is appropriate for a proceedings. If you give a paper at our annual meeting and submit a copy for publication, then you are included in the Proceedings. In any case, the contributors to this Proceedings should be commended for taking the time and making the additional effort to put their thoughts and research on paper (and in numerous figures and tables). Thank You!

Some free advice. Don't ever send in a map with a legend: 1 inch = 1 mile. Once we make a photo-reduction, your river is only 1/4 as long as it use to be. I won't mention any names, but we used the white-out on page 34. Do it like Vince (page 23), except use something besides a crayon.

Although the main theme of our 1990 meeting was aquatic plants and their management, we don't have a paper on bio-control using aquatic insects or on mechanical harvesting. Maybe our only major weak point. But we did see what a problem aquatic macrophytes can be and how chemicals are being used to control them (Langeland et al., p. 1), how they effect the biota (Bartodziej, p. 33; Thayer, p. 51) and water quality (Williams and Cook, p.21), and how they can be used to treat stormwater (LaRock et al.). Remember in English class, where you read a six line poem and then the teacher told you to write a short three page essay (in 45 minutes) discussing the poem's true hidden meaning? Give me a break! How can you write three pages about six lines unless you're a politician (everybody knows they can talk for hours about nothing). Well, check out Karen's three pages (K. Brown, pp. 17-19). She'll explain how you can take advantage of APIRS; you'll be writing whole volumes about something in no time. And finally, be sure to read the paper, "Importance of Isolated Wetlands in Upland Landscapes" by Linda LaClaire and Richard Franz, it was selected to receive the Society's first "best paper" award.

It is, of course, the hope of the FLMS that you will find this proceedings useful. Likewise, it is our hope that you will become involved in our society, if you are not yet active. You can contact me directly (Marty Kelly, SWFWMD, 7601 Highway 301 North, Tampa, FL 33637; 813/985-7481) or Richard Coleman, Executive Director (203 Lake Pansy, Winter Haven, FL 33881; 813/956-3771.

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Selective Hydrilla Management in Upper Myakka Lake

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Abstract

The aquatic herbicide fluridone was applied to Upper Myakka Lake each year from 1988 thru 1990 to manage hydrilla, which grows to problem proportions in the lake. Populations of Mexican waterlily, coontail and hydrilla were monitored to determine the impact of the herbicide applications to the nontarget Mexican waterlily and coontail and the effectiveness of the herbicide to control hydrilla in this system, which is subject to high rates of water exchange. Detrimental impacts of the herbicide to Mexican waterlily and coontail were not observed. Hydrilla frequency decreased by as much as 60% as a result of the herbicide applications in 1988 and 1989 but little effect was observed in 1990.

Introduction

Upper Myakka Lake is located totally within the boundaries of Myakka State Park (Sarasota Co.), one of Florida's largest and most frequented parks by both out of state and resident visitors. At 395 ha (Shafer 1986), the lake is one of the park's major resources, especially to resident visitors who utilize the lake's sportfishery. The lake has "tremendous regional significance in a rapidly growing area where freshwater fishing opportunities are limited" (Champeau 1988). Visitors also enjoy observing the lakes abundant wildlife from canoes, small power boats or from the 73 passenger, "Gator Gal" airboat.

Upper Myakka Lake and the Myakka River, its main tributary, are located in the Gulf Coastal Lowlands physiographic province and in regions dominated by deposits of the highly phosphatic Hawthorne formation (Puri and Vernon 1964). As influenced by geologic characteristics, the lake is naturally eutrophic and the lakewater is moderately hard. Canfield (1981) reported an average total phos-

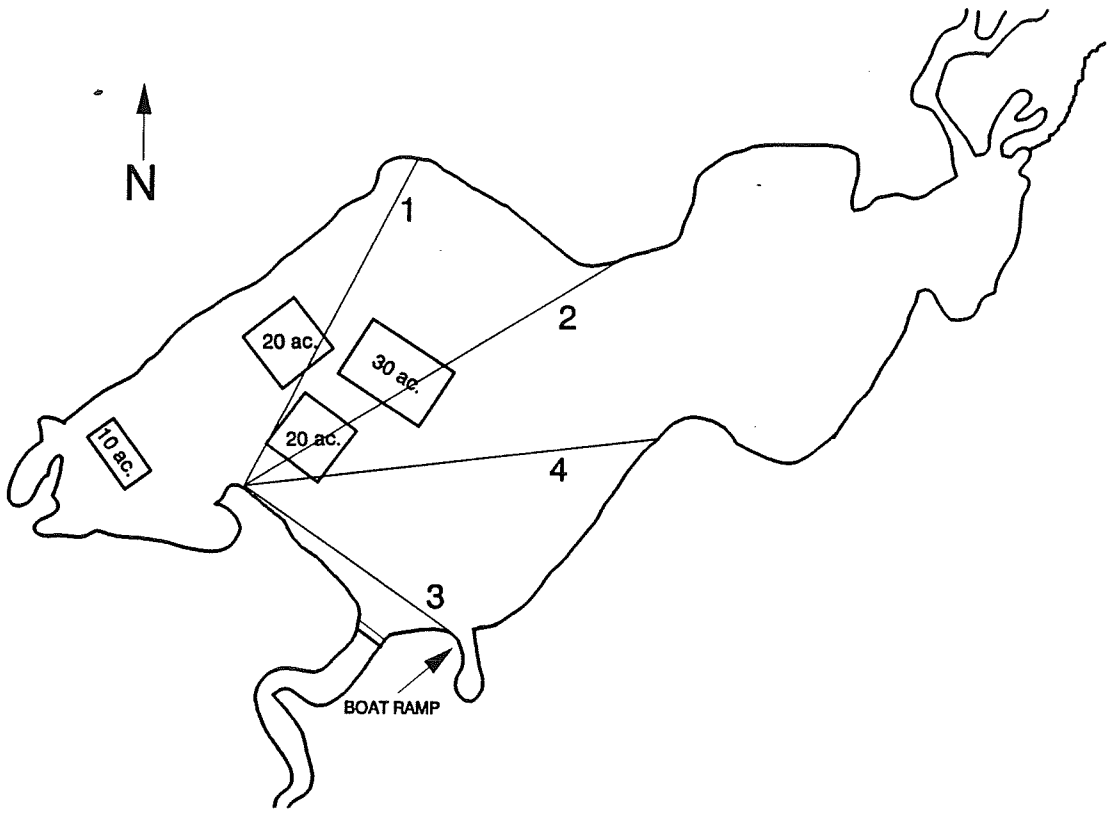

Figure 1. Location of plots in Upper Myakka Lake where Sonar 5P was applied at a rate of 1.68 kg fluridone/ha (1.5 lb/ac) in April 1988.

phorus concentration of 207 mg/m^3, pH of 7.6, total alkalinity of 30 mg/l as $CaCO_3$, total hardness of 80 mg/l as $CaCO_3$ and specific conductance of 201 umhos/cm. The lake is also influenced by non point sources of nutrient inputs from agricultural activities in the drainage basin (priede-sedgwick, inc. 1983).

The high amounts of nutrients and resulting high productivity not only allow for an excellent fishery and abundance of waterfowl and wading birds, but also a diverse assemblage of aquatic vegetation that are enjoyed by fisherman, naturalists, and casual observers. Fifteen species of aquatic angiosperms and ferns were reported in 1988 (Champeau 1988).

Water chemistry and shallow depth have allowed for prolific growth of hydrilla (*Hydrilla verticilla*). This aggressive submersed aquatic angiosperm, which is native to warmer regions of the Old World, was first identified in the United States in the Miami and Crystal Rivers, Florida in 1959 (Blackburn et al. 1969) and has spread throughout the state to cover more than 20,000 ha of water bodies (Schardt and Nall 1988). Hydrilla causes major detrimental impacts to recreational water use by interfering with boating, replacing native plant populations, and degrading fisheries and wildlife habitat. Hydrilla is currently managed with specially designed mechanical harvesters, biologically with triploid grass carp (*Ctenopharyngodon idella*) and with several herbicides that are registered for aquatic sites. Site characteristics, intended water uses and economics, among other criteria, determine which method(s) is used.

Hydrilla was first found in Upper Myakka Lake in the mid 1960s (Champeau 1988). Use of mechanical harvesters or triploid grass carp are not feasible in this lake for various reasons. Hydrilla management in the lake by the Southwest Florida

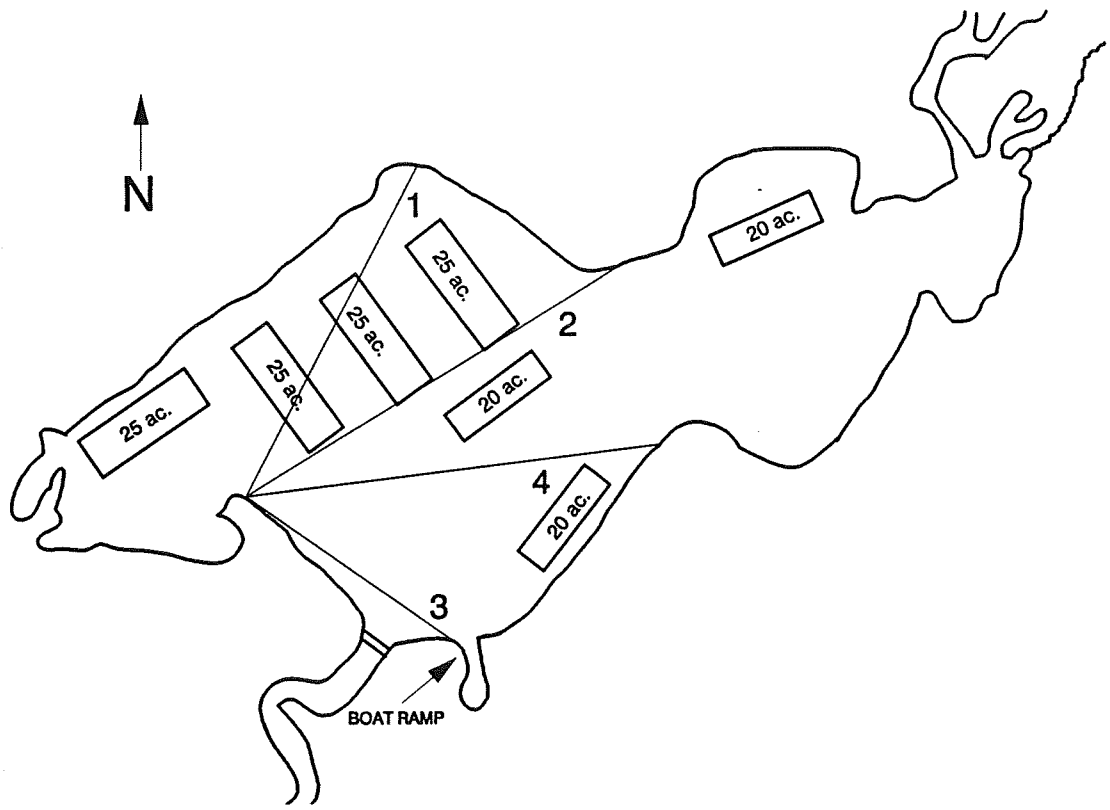

Figure 2. Location of plots in Upper Myakka Lake where Sonar5P or Sonar a.s. was applied at a rate of 1.68 kg fluridone/ha (1.5 lb/ac) in April 1989. All 25-acre plots and the adjacent 20-acre plot were treated with Sonar 5P. The remaining 20-acre plots were treated with Sonar a.s.

Water Management District (SWFWMD), through 1987, has entailed use of the contact herbicide, endothall, to treat boat trails, or sometimes larger areas, and has occurred only when economically and politically feasible (i.e., when outboard motors were not allowed on the lake federal cost sharing funds were not available). Hydrilla regrowth occurred soon after these herbicide applications. The result has been inadequate hydrilla control through 1987 that has resulted in inhibited navigation and public utilization of the fishery resource and reduced quality of largemouth bass (*Micropterus salmoides*) and black crappie (*Pomoxis nigromaculatus*) (Champeau 1988).

A potential new aquatic herbicide, fluridone (1-methyl-3-phenyl-5-[3-(trifluoromethyl) phenyl[-4(1H)-pyridinone), was evaluated in Florida Lakes during 1981 through 1985 to determine the feasibility for registering it for aquatic use. During this testing it was determined that fluridone was very effective for controlling hydrilla and several other aquatic plants and that some native aquatic plants were tolerant. Fluridone was registered by EPA for aquatic use in 1985 and it has been used for hydrilla control in some large Florida lakes such as Lake Harris, Lake Istokpoga and Lake Kissimmee, where at least a full year of control was observed.

Experimental applications of fluridone were made in Upper Myakka Lake in 1984. According to Florida Department of Natural Resources survey records, hydrilla in Upper and Lower Myakka Lakes and the river section between them decreased from 950 acres in 1984 to 160 acres in 1985 and 250 acres in 1986. This suggests that fluridone was effective. However, interpretation is difficult because the Sonar application was followed by applications of endothall and diquat.

According to Florida Park Service (FPS) guidelines, a major consideration in determining management

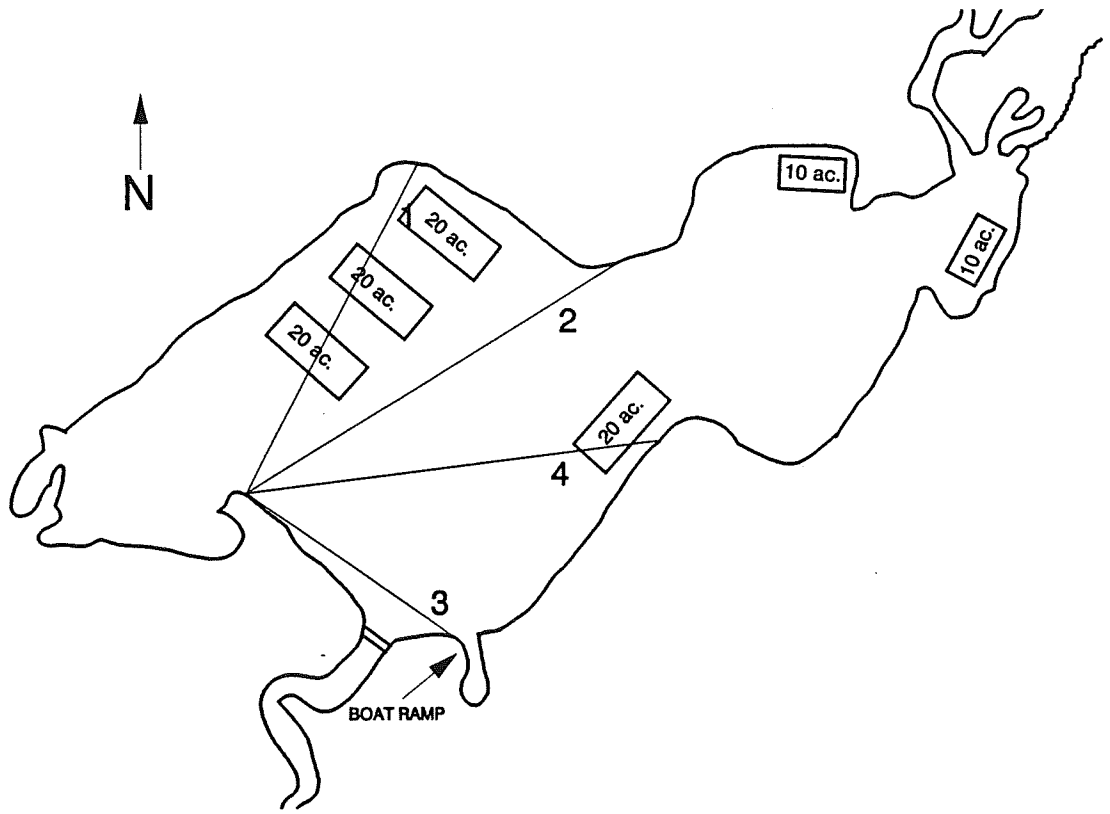

Figure 3. Location of plots in Upper Mayakka Lake where Sonar 5P was applied at a rate of 1.68 kg fluridone/ha (1.5 lb/ac) in March 1990.

measures to be taken with regard to exotic plants is the parks natural qualities, and priorities are given to those exotic species that are a threat to the natural environment of the area. These guidelines are consistent with aggressive hydrilla management in Upper Myakka Lake. However, FPS guidelines also stipulate that no pesticide will be used where there is basis for belief that water quality will be degraded or fish, or wildlife, their food chain or other components of the natural environment will be threatened. This caused concern over the use of fluridone for hydrilla control in Upper Myakka lake because it was feared that the herbicide would not only control hydrilla but also native and preferred aquatic plants in the lake.

Of special concern was the abundant population of Mexican waterlily (*Nymphaea mexicana*) in the lake. Although Long and Lakela (1971) consider Mexican waterlily non-native to Florida, it was observed in the state in the early 1800s (Gray 1876) and others believe that it is a native species probably established well before European colonization (Richard Wunderlin and Bruce Hansen, University of South Florida Herbarium, personnel communication). These plants are considered to be an important resource to the park because their showy yellow flowers are a major aesthetic attraction and the plants are probably important to fisheries, waterfowl and wading birds.

In the spring of 1988, a study was begun to determine the impacts of fluridone applications on Mexican waterlily and coontail (*Ceratophyllum demersum*) in Upper Myakka Lake. This data would be essential to develop an effective management program for hydrilla in the lake.

Materials and Methods

Sonar 5P (5% clay pellet formulation of fluridone) was applied to 32.4 ha (80 ac) in Upper Myakka

lake, at a rate of 1.68 kg ai/ha (1.5 lb/ac) in May 1988 (Fig. 1). Aquathol K (40.3% dipotassium salt of endothall liquid formulation) was applied to 24.3 ha (60 ac) at a rate of 16.8 kg ae/ha (15 lb ae/ac) to the area in front of the boat ramp but not within the Sonar 5P plots (Fig. 1) in June 1988. In April, 1989, 48.6 ha (120 ac) of the lake were treated with Sonar 5P at a rate of 1.68 kg ai/ac and 16.2 ha (40 ac) were treated with Sonar A.S. (41.7% aqueous suspension of fluridone) at the same rate (Fig. 2). Sonar 5P was applied to 40.5 ha (100 ac) of the lake in May, 1990 (Fig. 3).

Responses of Mexican waterlily, coontail and hydrilla to the Sonar 5P and Sonar A.S. applications were measured by determining presence or absence of the plants with a grapple, at four diametrically opposed positions, at 31 m intervals, along four line transects (Figs. 1-3). Sampling was conducted throughout the waterbody, rather than strictly within the actual treated areas, because it was assumed that fluridone mixed throughout the lake resulting in uniform effects. Frequencies of the plants were averaged among transects and compared among sampling times. Only sample locations that contained Mexican waterlily at the beginning of the study were used for analysis of this plant, in order to reduce variability. All statistical analysis were performed with the aid of the Statistical Analysis System (SAS Institute Inc.).

Sonar concentration was measured in lakewater by HPLC (Fox et al., in press) analysis one month following herbicide application in 1990.

Results and Discussion

Visual symptoms (predominance of red and yellow pigments) typical of physiological responses to fluridone (inhibition of carotenoid pigment synthesis and subsequent bleaching of chlorophyll) were evident on some leaves of Mexican waterlily and subsequent mortality of some leaves was observed. However, these leaves were rapidly replaced by newly produced leaves and the frequency of Mexican waterlily did not change during the three years of the study (Fig. 4). In fact, there is a trend toward increasing frequency of Mexican waterlily following the second Sonar application in April of 1989 through June of 1990. Fragrant waterlily (*Nymphaea odorata*) is reported to be sensitive to Sonar and can be controlled in enclosed ponds at recommended application rates (David Tarver, personal communication) that result in up to 120 ppb. However, assuming that all the fluridone applied was dissolved in the lakes water in the applications and diffused throughout the lake, the maximum potential lakewide concentration in Upper Myakka

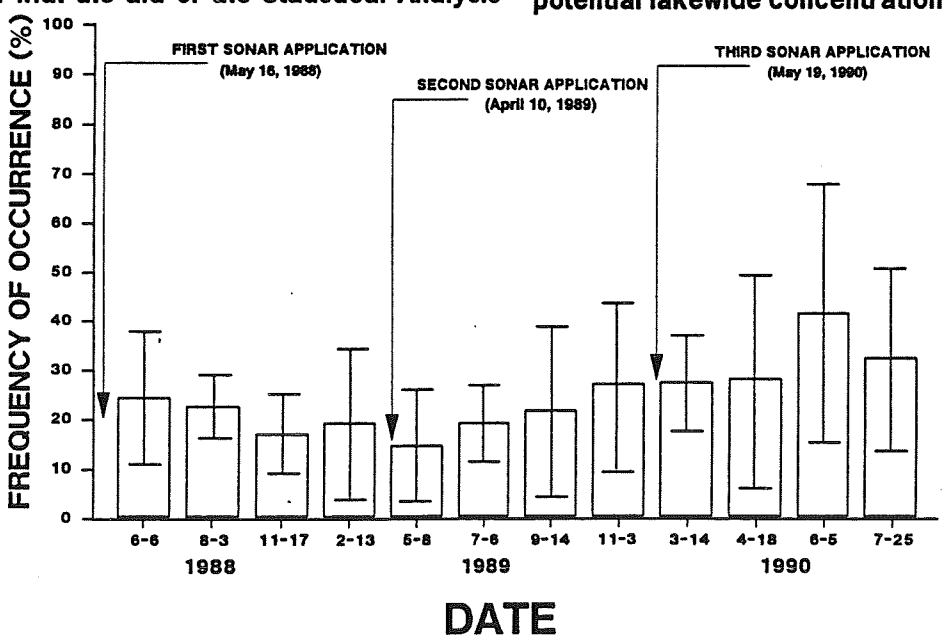

Figure 4. Occurrence of Mexican waterlily in Upper Myakka Lake during 1988-90 while fluridone was used for hydrilla management.

Lake would have been only 9, 18, and 11 ppb in 1988, 1989, and 1990 respectively.

Coontail was present at low frequencies and did not change during the three year period (Fig. 5). Although it is considered sensitive to Sonar at recommended application rates in ponds (David Tarver, personal communication), coontail is apparently tolerant to the concentrations and length of exposure to fluridone in this study. Coontail was also unaffected by Sonar applications in Lake Okeechobee (Langeland et al. 1988).

Frequency of hydrilla occurrence was reduced from over 80% to less than 25% two months following Sonar application in 1988 (Fig. 6). However, regrowth occurred throughout the year and by May, 1989 hydrilla frequency had returned to the level that was observed prior to Sonar application in 1988. Hydrilla occurrence in August, 1988 was predominantly in areas that were not covered by lakewater during herbicide application but became submerged as lake water rose during the rainy season. Hydrilla produces tubers that are dormant but remain viable in moist soil and can sprout when water returns (Haller et al. 1976). Sprouting of tubers in these areas was responsible for hydrilla re-establishment. It is explainable because as waters rose, tubers were stimulated to sprout and at the same time fluridone was flushed from the lake. Re-occurrence of hydrilla throughout the lake may have resulted to some extent from fragments originating from these areas but occurred, for the most part, from sprouting of tubers, which were able to survive after fluridone disappeared, in deeper waters.

Two months following herbicide application in 1989, hydrilla frequency was reduced to less than 20% occurrence, did not increase throughout 1989 and did not approach 1989 pretreatment levels until March, 1990. Hydrilla begins producing tubers in response to short photoperiods in early fall continues through early winter and again produces tubers in early spring (Haller et al. 1976). Therefore, new tubers would have been produced during 1988. However, reduced regrowth following herbicide application in 1989 suggests that good hydrilla

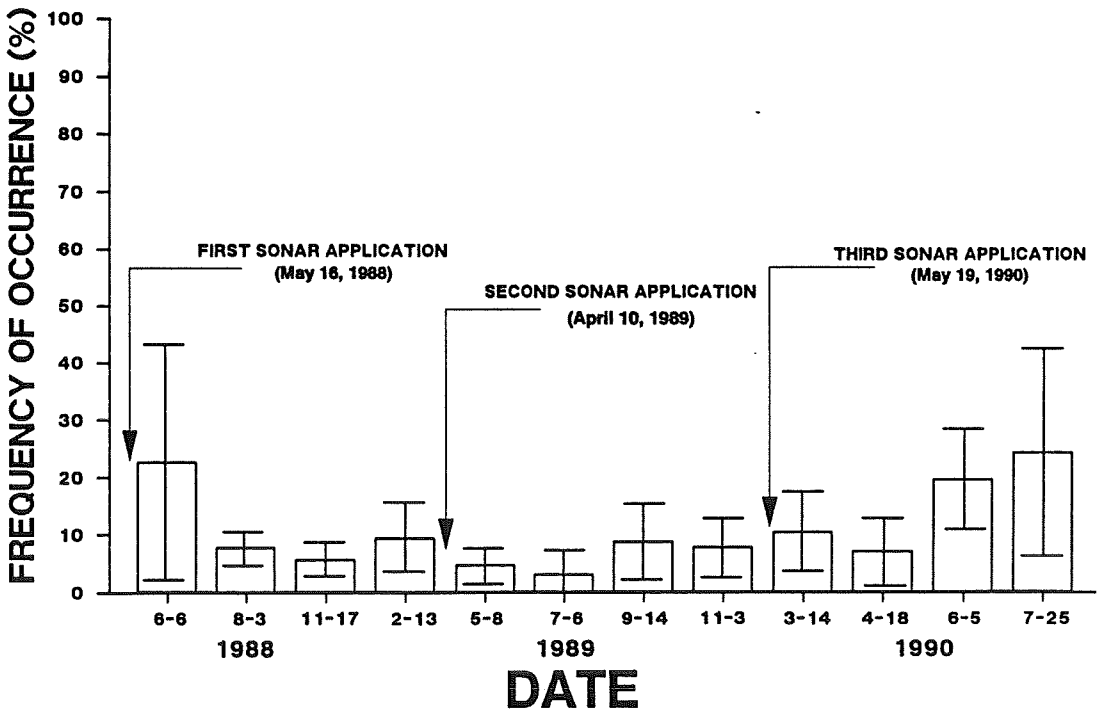

Figure 5. Occurrence of coontail in Upper Myakka Lake during 1988-90 while fluridone was used for hydrilla management.

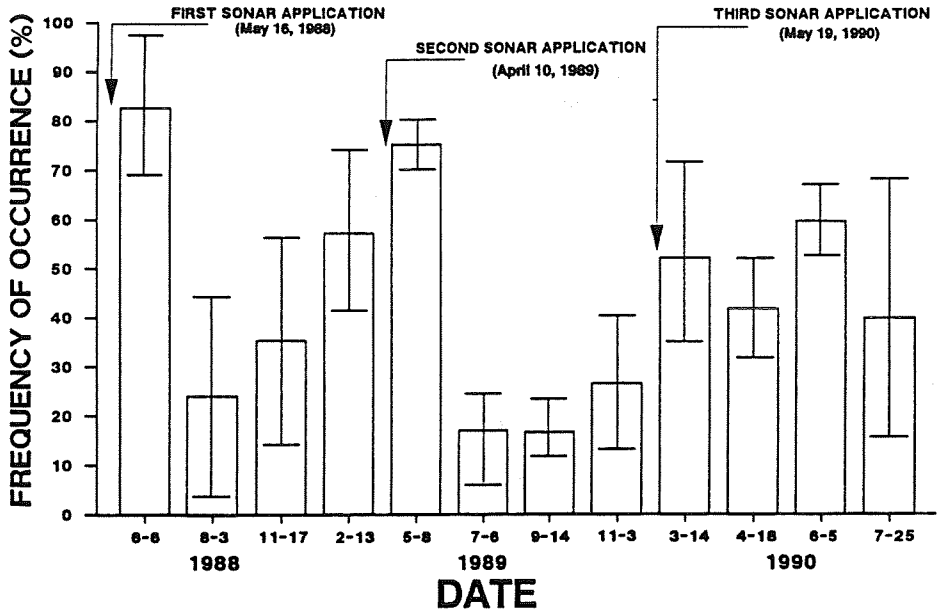

Figure 6. Response of hydrilla to application of fluridone in Upper Myakka Lake.

control in two consecutive years has an additive affect and this may be because tuber density was reduced as a result of the 1988 control.

Frequency of hydrilla occurrence did not change following herbicide application in 1990 but, in general, remained below levels that existed prior to Sonar application in 1988, through July 1990. Lack of hydrilla response to 1990 Sonar application probably resulted because of too short a contact time to the herbicide when rainfall increased flow rates of tributaries and flushed the herbicide out of the lake. This hypothesis is supported by the low fluridone concentration (2.15 ppb) measured in lakewater one month after Sonar was applied.

Conclusions

This study demonstrated that herbicides can be used to manage hydrilla in Upper Myakka Lake without detrimental impacts to native or non-target vegetation. During 1988-89, successful management of hydrilla resulted in increased navigation and fisherman access for most of the year and population structures of largemouth bass, bluegill and redear sunfish had significantly higher percentages of quality-size fish (Champeau 1990). Mexican waterlily populations, along with other native aquatic plants were maintained for the aesthetic enjoyment of park visitors and provided habitat for fish and other wildlife.

Hydrilla cannot be controlled in Upper Myakka lake for longer than one growing season by a single Sonar application (at least at the amounts applied during this study). Although hydraulic budgets are not available for the lake, this is probably due to loss of the herbicide when seasonal rains cause exchange of lakewater. Likewise, when rain occurs soon after herbicide application, hydrilla may be unaffected.

Future studies should focus on determining rates of water exchange in the Upper Myakka Lake/Myakka River/Lower Myakka Lake system toward developing slow release or split Sonar application schedules that will optimize hydrilla control and, perhaps the use of other herbicides along with fluridone.

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Importance of Isolated Wetlands in Upland Landscapes

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Abstract

Perched, isolated wetlands in upland areas in the southeastern coastal plain are essential for a group of specialized vertebrates that tolerate highly variable hydroperiods. These ponds are particularly important as breeding sites for several upland amphibians with eggs and/or larvae that are vulnerable to fish predators. Several of these vertebrates are listed as declining species by the Florida Committee on Rare and Endangered Plants and Animals and/or the Florida Game and Fresh Water Fish Commission. In order to protect the ponds and their unique fauna, it is necessary to understand the role of wetland soils, specialized vegetation, and hydroperiod in the formation and maintenance of this habitat.

Introduction

Generally overlooked, isolated ephemeral ponds are important components of upland landscapes in the southeastern United States. Their importance has been underestimated because of their generally small size and because these sites are frequently isolated with respect to one another. A high level of connectedness has been given high value in some wetland habitat evaluations with a low value given to isolated wetlands (Brown & Starnes 1983). This has led to a disregard for the unique character of these wetlands and the flora and fauna that inhabit them.

As a result of their isolation in an otherwise droughty environment, isolated ephemeral ponds are used as watering places, sources of food, and in certain cases for nesting. Snakes, turtles, wading birds and mammals frequently use these ponds. Most importantly, these wetlands are essential breeding sites for a significant number of upland amphibians, in particular striped newts (*Notophthalmus perstriatus*), mole salamanders (*Ambystoma talpoideum*), barking treefrogs (*Hyla gratiosa*), and gopher frogs (*Rana areolata*). These species have eggs and/or larvae that are vulnerable to fish predation. They are able to successfully inhabit

droughty uplands as a part of a group of specialized vertebrates that are adapted to highly variable hydroperiods. Several of these vertebrates are listed as declining species by the Florida Committee on Rare and Endangered Plants and Animals and/or the Florida Game and Fresh Water Fish Commission. In order to gain protection for this unique fauna, it is imperative that their breeding ponds are protected and the ecology of these ponds is understood.

We have recently begun a study of isolated, ephemeral ponds in three Florida counties to better understand the relationship between these wetlands and their use by amphibians. Ponds known to be breeding sites for the gopher frog (*R. areolata*) and the striped newt (*N. perstriatus*) were selected. Our study sites are contained within the Apalachicola National Forest in Leon County, the Ocala National Forest in Marion and Putnam Counties, the Katherine Ordway/Swisher Memorial Sanctuary and the Welaka Research and Education Center in Putnam County. Each study site consists of 2-6 ponds characterized as round, shallow depressions that irregularly fill with water. They are sinkhole in origin and tend to be associated with sand ridges. The ponds are surrounded by upland habitats of high pine, scrubby flatwoods, or scrub.

Isolated, ephemeral ponds support a very different plant and animal community than those found in larger, more permanent wetlands. Permanent wetlands may dry in exceptional years and in such cases, most of the permanent water fauna will be extirpated. Recolonization by fish and other permanent water fauna will only be possible when high water connects the wetland to others where the fauna is present. In temporary waterbodies, selection has occurred for species adapted to a cycle of drying and re-filling. Such an environment provides a unique habitat for plants and animals as a result of isolation and varying hydrology. For them, stability of water level is not a beneficial condition. In our study, we are looking at the role of pond soils, specialized vegetation, and hydrology in the formation and maintenance of this habitat.

At the beginning of 1990, we began an assessment of the physical and botanical aspects of our study ponds. Those holding water were sampled for amphibian larvae. Vegetation and soil were analyzed along transects within dry pond basins. From the data collected thus far, we are able to construct a preliminary picture for isolated, ephemeral ponds contained within upland habitats of north-central Florida.

VEGETATION

The plants encountered within ephemeral pond basins reveal a range of adaptations in tolerating inundation and represent a succession that results in zonation of the vegetation (Lippert and Jameson 1964). The gradually decreasing water depth typical of ephemeral pond drying often results in zones of vegetation in concentric rings of different species and vegetative types. Both the frequency and duration of pond filling is reflected in the vegetation of the concentric bands. When the pond is filled, the inner zone often contains floating-leaved plants, then submergents, tall and short emergents at the pond edge and lastly water-tolerant shrubs or trees in some transitional zones (Weller 1978). As water levels change, the bands of vegetation move back and forth across the pond basin in a reflection of soil moisture conditions. After the pond dries, emergent grasses fill the previously flooded portion and the submergents senesce or adjust their growth to a terrestrial form.

The vegetation of the ponds we have analyzed typically form a pattern as shown in Figure 1. Maidencane (*Panicum hemitomon*) is an obligate wetland species (Reed 1988) that has been found in all of the ponds. It prefers very moist soil and thus the upslope location of the *P. hemitomon* band represents the edge of the previously flooded portion of the pond. Further upslope, a ring of sandweed (*Hypericum fasciculatum*) represents the next level of soil moisture. It is considered a facultative wetland plant in the National List of Plants Occurring in Wetlands with a probability approaching 99% of usually occurring in wetlands

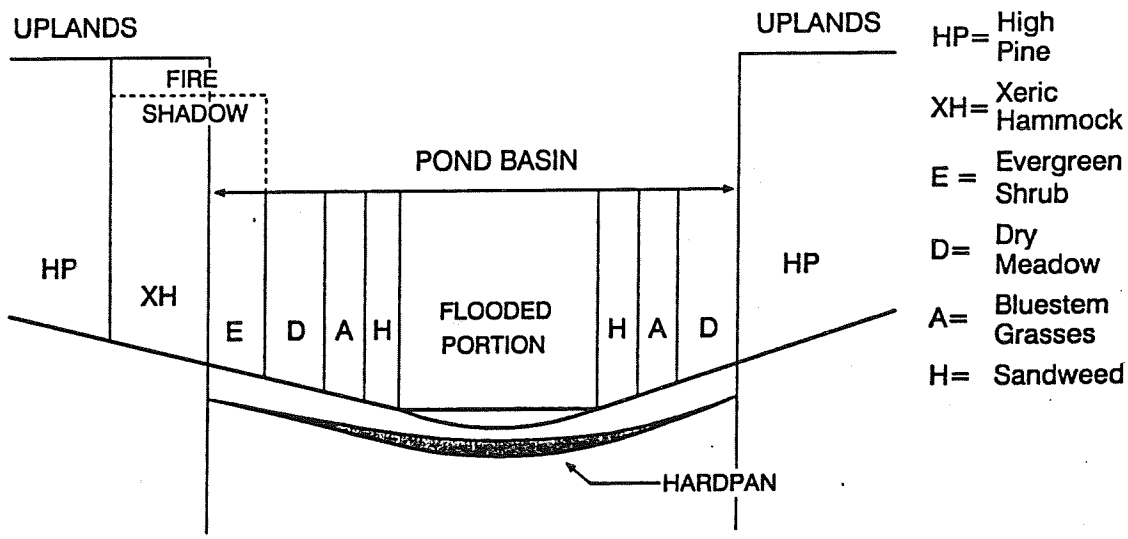

Figure 1. Vegetational structure of pond basin with associated hardpan.

(Reed 1988). This category of wetland plant requires less soil moisture than an obligate wetland species. Bluestem grass (*Andropogon virginicus*) dominates the next ring of vegetation in many of our ponds. Its status as a plant in the next level of soil moisture is reflected in a listing as a plant equally likely to be found in non-wetlands as wetlands. Continuing upslope, vegetation is found in dry meadows consisting of species occurring equally in wetlands and non-wetlands and species occurring generally in uplands. The species composition of each of our study ponds differs depending on existing soil moisture conditions, hydroperiod and surrounding upland type, but the general pattern of banding remains consistent.

An additional feature of several ponds is the presence of a fire shadow along the upslope wetland/upland boundary (Fig. 1). The vegetation within the fire shadow contains fire intolerant evergreen shrubs (*Ilex, Vaccinium, Myrica, Ceratiola*) and sometimes xeric oak hammock woodland zones. Ponds that have been burned completely from upland margin across to opposing upland margin lack this vegetation. However, under certain conditions, particularly when the ponds are filled, fire will slow and burn out as it goes down slope into the depression basin. This allows invasion of less fire resistant species on the protected side.

The amount of disturbance or pertebation within the pond basin has a direct effect on the diversity and species composition of the vegetation. Human perturbations such as the use of isolated, ephemeral ponds by off-road vehicle enthusiasts tends to reduce the diversity of vegetation found in them (pers. obs.). Also, plowing into the dry meadows along pond margins can result in the invasion of other plant communities, including sand cordgrass (*Spartina bakeri*) meadows and components of xeric oak hammocks. On the other hand, our initial pond surveys indicate that disturbance by fire tends to increase the species diversity and maintain original habitat components.

SOILS AND HYDROLOGY

The vegetation of isolated, ephemeral ponds is directly related to the underlying soils. Most of our study ponds are located in regions of sand ridges and have the sandy soils corresponding to those of the surrounding uplands. However, organic matter has accumulated in the pond basin and influenced the formation of soil surface horizons that can be classified as hydric soils. If the pond has been filled for several seasons, organic material will accumulate on top of the surface sands. As the pond alternately dries and fills, a subsurface organic hardpan can form below the top layers of sand. This is a result of leaching of organic acids from the dead vegetation above. The sand grains become coated with organic material and begin to adhere to each other. This results in a dense layer that acts as a confining bed by preventing water from draining into the sand layers below. Above and below this organic hardpan the sands are streaked with organic matter. Because of the gradual drying of ephemeral ponds and their occurrence in depressions, the center of the pond holds water longest. As a result, the accumulation of organic matter is thickest in the middle and becomes thinner towards the outer edges of the basin. The corresponding variation in soil moisture retaining capacity has a direct effect on the bands of vegetation that form as described above. Our initial soil surveys correspond with this pattern.

Hydrology is probably the single most important determinant for the establishment and maintenance of isolated, ephemeral ponds (Mitsch and Gosselink 1987). Water depth, flow patterns, and duration and frequency of flooding all influence the biochemistry of the soils and the ultimate selection of the biota. Precipitation is the most important source of water for these wetlands. The amount of precipitation required to re-fill a pond once it has dried depends on the amount of soil moisture, the permeability of the hardpan and the slope of the surrounding pond basin. The more saturated the soil, the smaller the input of rainfall is required to refill a pond. If an organic hardpan has formed, it will maintain soil saturation for a longer period of time.

As the pond fills, there is a pulse of primary productivity as nutrients enter the system. Flooding the soil can also increase the availability of nutrients; phosphorous has been shown to be more soluble under anaerobic conditions (Patrick and Khalid 1974). Over time, plants within the pond basin grow and senesce. Organic matter accumulates on the pond bottom because of slowed decomposition under anaerobic conditions. But, as the pond dries, aerobic decomposers will begin rapidly breaking down the organic matter. Fungi and bacteria colonize the detritus also. Their contribution in terms of additional protein content acts to increase the nutrient value of detritus upon pond re-filling beyond that of permanent ponds (Barlocher et al. 1978). The resulting nutrients will then become available upon the next flooding event. Thus, a building up of nutrient supply occurs and through time the combination of biotic and abiotic elements enriches the pond environment.

AMPHIBIAN USE

In the southeastern coastal plain, ten species of anurans and five species of salamanders breed primarily or exclusively in small, isolated, ephemeral wetlands. In addition, at least ten other species of anurans and seven species of salamanders utilize the habitats opportunistically as well as more permanent sites (Moler and Franz 1987). Table 1 shows the breeding habitats utilized by frogs and toads in Florida. The obligate ephemeral pond breeders cannot withstand the greater range of predation, both vertebrate and invertebrate, encountered in more permanent wetlands (Wilbur 1980). The ephemeral nature of the pond also appears to be an integral component in successful reproduction of ephemeral pond breeders. Two studies (Caldwell 1987, Pechmann et al. 1987) have shown that these amphibians were able to reproduce successfully only in those seasons when the site had dried completely and then refilled. This suggests that the increased availability of nutrients that occurs with pond filling is important in promoting rapid growth and high population densities of larval amphibians. Also, the drying of the pond eliminates possible fish and other large predators

so that when the pond fills the larval amphibians have an initial size advantage over invertebrate predators (Wilbur 1980).

Several studies have quantified the species composition and density of breeding amphibians that can be supported by very small breeding ponds. Pechmann et al. (1987) documented the metamorphosis of 75,644 juvenile amphibians of fifteen species from a 1 ha ephemeral pond in South Carolina. Wharton (1978) found that annually, 1600 salamanders and 3800 frogs and toads moved to a small (<30m diameter) gum pond in Georgia. Dodd and Charest (1988) monitored the use of an ephemeral pond (<0.2 ha) at the Ordway Preserve in Putnam County, Florida over a period of 27 months and captured 2870 individual frogs of 14 species and 1354 individual salamanders of two species.

We have been able to gain only a preliminary understanding of which amphibian species breed in our study ponds due to the current drought in the state of Florida. However, we did find larval amphibians in five of our study ponds in Leon and Marion counties in 1990. In Leon County, four ponds held water during the winter breeding season. In these we found a larval amphibian community composed of the barking tree frog (*H. gratiosa*), spring peeper (*H. crucifer*), southern leopard frog (*R. utricularia*), gopher frog (*R. areolata*), southern cricket frog (*Acris gryllus*), striped newt (*N. perstriatus*), central newt (*N. viridescens*) and mole salamander (*A. talpoideum*). In addition, one of our ponds in the Ocala National Forest has intermittently held water from March, 1990 until the present (October, 1990). During March and late May surveys, larval striped newts (*N. perstriatus*) were found. When the pond was surveyed in September for evidence of late summer breeding, we found tadpoles of the frog species *H. gratiosa, H. femoralis,* and *A. gryllus*, but no newts were present. During the ensuing study we will be able to determine a more complete list of the species using these isolated, ephemeral ponds.

IMPLICATIONS FOR CONSERVATION

Ephemeral ponds are integral parts of upland landscapes. They contribute to ecosystem diversity by providing rich resource patches imbedded within the surrounding habitat (Laney 1988). The vegetation and soils accumulate nutrients and the ponds provide a source of water in droughty environments. As a circular patch, they provide a high interior to edge habitat ratio which may increase the species diversity and foraging efficiency of animals within the pond basin (Forman and Godron 1986). The elimination of these wetlands acts to fragment the surrounding uplands and may reduce or prevent species migration along corridors or exchange between isolated patches.

Upland amphibians rely on ephemeral wetlands and are important elements in the food chains of terrestrial predators and wetland bird species. They can contribute significantly to the biomass of terrestrial ecosystems and in some cases amphibians have been shown to be the dominant vertebrate group (Burton and Likens 1975, Gosz et al. 1978). They also represent one of the few vertebrate biotic mechanisms for the transport of nutrients out of eutrophic waterbodies and into terrestrial ecosystems (Wassersug 1975). Most species of upland amphibians (except for terrestrial breeders) are dependent on isolated ephemeral ponds for the completion of their life cycle. They are susceptible to habitat fragmentation, degradation of suitable upland sites, and most importantly to the loss or degradation of their breeding sites. Different amphibian species breed at different times of the year; therefore, multiple ponds with different filling regimes may be necessary to allow for successful reproduction of all species. Recent attention has been focused on amphibians that are declining worldwide and their potential as sensitive bioindicators of environmental change. Their decline is an indication of deeper ecosystem deterioration and decay of the food web (Blaustein and Wake 1990, Borchelt 1990, Vitt et al. 1990). The disappearance of upland amphibians from the landscape also has inevitable, major consequences for other components of the ecosystem, since

amphibians form important links in upland food webs.

Future studies will focus on describing the biotic and abiotic components of isolated, ephemeral ponds. The mechanisms for pond development and maintenance are not well understood and more information is necessary for future successful restoration or reconstruction. A greater appreciation and understanding of the ephemeral pond breeding sites of amphibians is necessary to ensure amphibian survival and the protection of the upland faunal community.

Table 1. Breeding habitats utilized by frogs and toads in Florida (modified from Moler and Franz 1987).

	Species	Small Isolated Wetlands	Open Marsh	Riparian & Forested Wetlands
*	Eastern spadefoot (*Scaphiopus holbrookii*)	X		
*	Oak toad (*Bufo quercicus*)	X		
*	Barking treefrog (*Hyla gratiosa*)	X		
	Squirrel treefrog (*H. squirella*)	X		
*	Pinewoods treefrog (*H. femoralis*)	X		
*	Little grass frog (*Limnaoedus ocularis*)	X		
	Ornate chorus frog (*Pseudacris ornata*)	X		
	Southern chorus frog (*P. nigrita*)	X		
*	Gopher frog (*Rana areolata*)	X		
*	E. narromouth toad (*Gastrophryne carolinensis*)	X	+	
	Spring peeper (*H. crucifer*)	+		+
*	Southern toad (*B. terrestris*)	+	+	+
	Green treefrog (*H. cinerea*)	+	+	+
	Gray treefrog (*H. chrysoscelis*)	+		+
*	S. cricket frog (*Acris gryllus*)	+	+	+
	Bronze frog (*R. clamitans*)	+		+
	Pig frog (*R. grylio*)	+	+	+
	Bullfrog (*R. catesbeiana*)	+		+
*	S. leopard frog (*R. utricularia*)	+	+	+
	Carpenter frog (*R. virgatipes*)	+	+	+
	Fowler's toad (*B. woodhousei*)			+
	Bird-voiced treefrog (*H. avivoca*)			+
	Pine Barrens treefrog (*H. andersonii*)			+
	Upland chorus frog (*P. triseriata*)	+		+
	N. cricket frog (*A. crepitans*)			+
	River frog (*R. heckscheri*)	+		X
	Bog frog (*R. okaloosae*)			X

X = principal or exclusive breeding habitat
+ = breeding habitats utilized
* = typical frog components of upland habitats

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The Aquatic Plant Information Retrieval System (APIRS)

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The Aquatic Plant Information Retrieval System is part of the Center for Aquatic Plants at the University of Florida in Gainesville. With a name as long as the Aquatic Plant Information Retrieval System, we need an acronym and, unfortunately, that comes out to

APIRS

What is APIRS? APIRS is a computerized collection of scientific literature about aquatic plants. We have been collecting literature, cataloging it and entering it into a database for about ten years. We now have more than 29,000 references pertaining to aquatic plants. References can be books, articles, abstracts, or any other type of scientific information about aquatic plants, although our collection is primarily journal articles and books. Our 29,000 references include over 2,500 on water hyacinths, over 1,300 on hydrilla, over 2,100 on Myriophyllum, over 1,800 on cattails, and over 2,300 on duckweeds.

Obviously, a lot of these references are on controlling these plants. There are over 3,400 on chemical control, over 2,300 on biological control, over 750 on mechanical control.

As we all know, however, the study of aquatic plants includes many other areas of interest besides controlling weeds. We have over 2,500 references on utilization, over 2,400 on wetlands, over 1,600 on pollution control and wastewater treatment, over 450 on aquatic plants and waterfowl, over 300 on created or artificial wetlands, and over 120 on aquascaping or revegetation.

It has taken a great deal of work to amass over 29,000 literature references about aquatic plants, not to mention the fact that we have hard copies of about 98% of them stored in our library. We search for titles in journals, proceedings, publications lists, book reviews, publishers announcements and in other databases. When a relevant title is found, we obtain the article, book or report from our university library, from the author, from the research center or from other library sources. Each article is then read and cataloged into plant names, subject categories, and keywords. This information is entered into the database and the article is filed in with our hard copy collection.

What do we do with all of this information? The purpose of collecting, cataloging and entering these references into a computer is to be able to perform literature searches specific to a requestors needs. When a request is received, we run a computer search of the database using the plant species, categories and keywords provided by the user. For example, if someone requests a literature search on the biological control of water lettuce using insects, we enter *Pistia* and biological control and insects into the database. The computer searches through all 29,000 references and produces a list of documents containing these key terms. There are 73 references for this search. The selected references are printed out as a

bibliography, with each one including the title, date of publication, author, citation and the various categories, keywords and plant names that describe the content of the reference. The printout is mailed to the requestor usually within one or two days.

How is this system useful to you? If you manage a water body and have just spotted a weed that needs to be controlled, you can call or write to request a literature search on controlling the plant, with chemicals or fish or insects or pathogens or machines. We will send you a bibliographic list of the most recent literature available. If your field is pollution control, you can request a literature search on heavy metal uptake by aquatic plants. If you are a researcher, you can request a literature search on any aquatic plant and its morphology, physiology, taxonomy or reproduction. We have information on distribution, cultural control, economics, integrated control, productivity, remote sensing, utilization, eutrophication, plant succession, toxic plants, host plants, aquatic herbicide toxicology, and more. If it has anything to do with aquatic plants, we have it or we will make every effort to get it.

How much does it cost to use this resource? Nothing but the cost of a phone call or a postage stamp. However, we are an information sharing system, so we rely on contributions of reprints. So put us on your mailing list. For every research article, report or book that someone sends us, that is one less that we have to track down through libraries, research centers, etc. Many of our 29,000 references on aquatic plants have been contributed by the authors who, in turn, have benefitted from the use of the APIRS.

Where do you get copies of those references you want to read? If you can not find them through your local library resources, we can send out photocopies of up to twenty or so articles. If you need more than that, you are welcome to visit our office to read or photocopy what you need.

Who uses the database? Anyone who wants to. We help aquatic plant managers, researchers, professors, extension agents, government agency personnel, students, and anyone else interested in aquatic plants. So far this year, we have produced about 600 bibliographies for 300 users, listing a total of over 22,000 citations. We have sent out almost 700 copies of articles. We have also collected, cataloged and added almost 2,000 documents to the database.

For those of you interested in the hardware/software aspect of the system, we use the BRS Search program which runs on the Xenix Operating System. The database is stored on a 230 mb hard disk on a microcomputer, and is backed up on tape cartridge. We can search through all 29,000 citations in the database in a couple of seconds and view selected references onscreen or in print. We are not yet interactive with remote users but that is a goal of ours for the near future.

OTHER SERVICES OF THE APIRS

AQUAPHYTE - The staff of the APIRS also produces the semi-annual newsletter, AQUAPHYTE, which contains articles, conference announcements, news items, book reviews, samplings of recent additions to the database, and other information relevant to aquatic plant research and management. Subscription to AQUAPHYTE is free.

AQUATIC PLANT ILLUSTRATIONS - Just recently, we started a collection of aquatic plant illustrations. Ms. Laura Line has produced over 50 botanical drawings, and they are now available to use for publications, teaching aids, library use, or public information material such as pamphlets or signs.

VIDEO PROGRAMS - Another endeavor of the APIRS is the educational videotape program. For the past two years, we have been collecting video footage on everything from herbicide applications to revegetation projects to research efforts and more. We have relied on the help of numerous aquatic plant managers, researchers, and agency personnel to shoot over 300 hours of raw video footage. So far,

three video programs have been released. CALIBRATION - A FIELD APPROACH features Dr. Bill Haller teaching herbicide equipment calibration techniques. ISTOKPOGA - LAKE OF LEGENDS tells about one of Floridas largest lakes and the treatment of its massive hydrilla infestation. FLORIDAS AQUATIC PLANT STORY acquaints general public audiences with the benefits of aquatic plants, the problems caused by some exotic aquatic weeds, and the major methods of aquatic plant management. FLORIDAS AQUATIC PLANT STORY has been shown on several television and cable stations in Florida, has been sent to many public agencies, and has been distributed to more than 50 elementary and secondary school science teachers and coordinators in the Florida school system. The video has also been used in environmental education forums, training programs and public education meetings. We are currently working with Aquatic Weed Extension Specialist Dr. Ken Langeland on a second calibration program that emphasizes the mathematics required to pass the certified pesticide applicator examination. We are also working on a series of aquatic plant identification videos, featuring University of Florida Extension Botanist Dr. David Hall. Dr. Hall will teach the basic taxonomic information necessary to identify about 100 of Floridas aquatic plants. The series will be presented in four programs: floating and floating-leaved plants, submersed plants, emersed plants, and grasses, sedges and rushes. All video programs are free to public agencies in Florida. For others, copies may be purchased or borrowed from the APIRS.

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To contribute reprints or request a literature search from the database, to be added to the mailing list for AQUAPHYTE, to request aquatic plant illustrations, or for more information on videotape programs, please contact the APIRS, 7922 N.W. 71st Street, Gainesville, Florida 32606, 904/392-1799.

Water Quality in the Littoral and Limnetic Zones of Lake Kissimmee

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ABSTRACT

Healthy communities of rooted aquatic plants have often been assumed to exert several beneficial effects on lake water quality. Aquatic plant communities have been found to lessen the potential for shoreline erosion and resultant turbidity by providing a physical barrier which dissipates wave energy, and to improve water clarity since macrophytes utilize nutrients for growth that otherwise could stimulate algal production. The purpose of this study was to analyze the effects littoral plant communities have on various physical and chemical water quality parameters of a central Florida lake.

From 1977 to 1986, quarterly water samples were collected and analyzed from four sites on an 800 m transect in Lake Kissimmee, Florida. The transect began at the lakes edge and extended 50 m beyond the littoral zone into open water. While most parameters measured showed little change in relation to littoral zone plant communities, there were statistically significant increases ($P < 0.05$) in unfiltered turbidity, total phosphorus, and chlorophyll a levels in open water as compared to the water column in the lakes littoral zone. These were accompanied by a significant reduction in open water secchi disc transparency. Trophic State Index (TSI) values indicated a shift in productivity within the littoral zone. The two landward locations were noted to be within the mesotrophic or mesotrophic/eutrophic range, the lakeward edge of the littoral zone was borderline eutrophic, while open water was classified as being in a moderately eutrophic state.

INTRODUCTION

Annual fluctuations in lake water level, both natural and artificial, stimulate growth of rooted aquatic macrophytes in Florida lakes (Holcomb and Wegener 1971). These plant communities have been shown to be biologically beneficial and necessary to production of numerous aquatic invertebrates (Wegener et al. 1974, Moyer and Williams 1982), and both forage and sportfish species (Wegener et al. 1973, Wegener and Williams 1974). In turn these invertebrates and fish provide a substantial food resource for wildlife, most notably wading birds. Plant communities directly serve as food supplies, and also offer waterfowl and wading birds protected nesting and rearing areas safe from

terrestrial predators. In addition, healthy sportfish populations associated with broad littoral areas are capable of supporting moderate to heavy levels of angling pressure, and serve as both recreational and economic assets to local communities.

In spite of these well documented and widely accepted values, recent questions have been raised concerning both the need for and biological value of rooted aquatic plants in Florida's lake systems. Canfield et al. (1984) stated that although increased macrophyte abundance lowers lake chlorophyll a levels, this does not suggest that macrophytes should be used to improve overall lake water quality unless there are large amounts of aquatic macrophytes present in the lake.

This study was initiated following the 1977 extreme drawdown of Lake Kissimmee (Osceola County, Florida) that was conducted by the Florida Game and Fresh Water Fish Commission to consolidate exposed lakebottom sediments, improve aquatic plant communities and associated aquatic habitat, and stimulate fish production. Under natural hydrologic conditions, water levels fluctuated an average of 1.2 m/yr over a vertical range of 3.8 m. Since 1961, water control structures have limited the lakes annual fluctuation range to 1.1 m between elevations 16.0 m MSL (meters above mean sea level) and 14.9 m MSL. During the drawdown, the water level was lowered nearly 3 m from high pool stage; this exposed, for three months, the entire lake littoral zone, which covers 45% of the lakes surface area. Aquatic plant communities exhibited dynamic changes over the duration of this study, with major species being eelgrass (*Vallisneria americana*), torpedograss (*Panicum repens*), maidencane (*P. hemitomon*), knotgrass (*Paspalidium geminatum*), fragrant water lily (*Nymphaea odorata*), and spatterdock (*Nuphar luteum*). No efforts were made to quantitatively assess vegetation communities at the water quality transect site.

While a significant body of literature exists concerning nutrient uptake by aquatic plants, assimilative capacities of marshes and lake systems, and nutrient cycling and transformation (Canfield et al. 1983, Wetzel 1983, Twilley et al. 1985, Hammer and Bastian 1989), there is little published work which simply compares physical and chemical characteristics of water quality in vegetated and open water portions of a lake basin. This study was designed to provide this information. Since Florida lakes commonly possess littoral zones which cover 20 to 40% of the lake proper, a thorough understanding of various plant functions is essential to development of long-term lake management strategies.

METHODS

From October 1977 to October 1986, four water quality samples were collected quarterly along a single transect line extending across the littoral zone of Lake Kissimmee, an 18,500 ha. eutrophic lake located in Central Florida. Samples were taken 50 m from the lake shore (T-1), at the midpoint of the littoral zone (T-2), in the littoral zone 50 m from open water (T-3), and in open water 50 m from the lakeward edge of vegetation (T-4) (Fig. 1). All samples were collected using a weighted bottle to provide a composite sample of the water column, and care was taken to avoid disturbance of macrophytes and sediments during sampling. Alkalinity, Secchi disc transparency, pH, and conductivity were determined in the field. Samples were iced and transported to the laboratory, where additional parameters (Table 1) were analyzed according to standard methods (APHA 1985).

Due to the fluctuating water levels, the nearshore transect site (T-1) was transitory within a 50 m distance along the transect. Water depths at the sampling stations ranged between 0.5 m and 3.0 m, depending on location and time of year. Data collected were statistically analyzed for arithmetic means, standard errors, and minimum and maximum values. Statistically significant differences between stations were considered to occur when there was no overlap between samples at 95% confidence limits ($x \pm 2$ SE). These differences were verified and seasonal variations were determined using ANOVA techniques ($P < 0.05$).

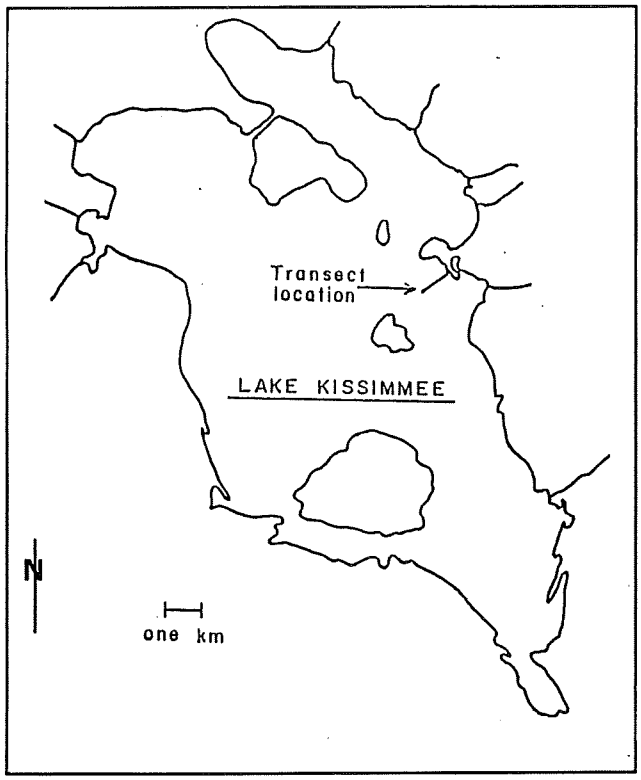

Figure 1. Transect location on Lake Kissimmee (Oceola County, Florida)

RESULTS

Table 1 provides a listing, along with mean values at each transect location, for measured water quality parameters. Sulfate, total phosphorus, chlorophyll a, unfiltered turbidity, and Secchi disc transparency showed statistically significant ($P < 0.05$) changes at various locations along the transect line (Table 2). Two additional values, particulate organic nitrogen and phaeopigments, were higher for the lakeward littoral (T-3) and open water (T-4) sites (Fig. 1). Although these latter increases were not considered significant at $P < 0.05$, both indicated greater production of free-floating algae at these lakeward locations.

SULFATE - Sulfate values (Fig. 2) show significant differences ($P < 0.05$) between T-1 and all other sample sites. No significant difference was found between T-2, T-3 and T-4 (Table 2). The trend is for increasing sulfate values (6.2, 8.7, and 9.6 mg/l, respectively, for T-1, T-2 and T-3), going from the innermost to the outer littoral zone, with the highest value being found in open water (10.2 mg/l). This may be due in a large part to the annual exposure, drying, and oxidation of lake bottom sediments in nearshore areas. Bayley et al. (1986) found that aerobic soil conditions during dry summer periods oxidized reduced sulfur to sulfate.

Temporal variations in sulfate concentrations were analyzed for each location (Table 3). Significant seasonal differences ($P < 0.05$) at T-1 were noted between fall and winter, fall and spring, winter and summer and spring and summer. Sediments along the shallow lakeshore were exposed and dried during the summer months. Higher sulfate values were observed during winter high water periods, which lends support to Bayleys observations. No significant differences were noted at T-2, T-3 and T-4. Seasonal exposure and drying of sediments did not occur at these deeper sample locations, and the significant differences noted at T-1 were not observed.

TOTAL PHOSPHORUS - Total phosphorus data (Fig. 3) indicates that mean values for littoral zone samples (T-1, T-2, and T-3) ranged from 0.05 to 0.06 mg/l. No statistically significant differences were found between T-1 and T-2; however, a significant increase was noted ($P < 0.05$) between T-2 and T-3. The difference between T-1 and T-3 ($P = .06$) fell slightly outside the significant value of $P < .05$. The mean for T-4 (0.09 mg/l) was significantly higher than any of the littoral zone means (Table 2).

Values for overall ranges (Fig. 3) may be misleading for two locations. The nearshore site experienced a highly atypical peak value of 0.23 mg/l, but only 3 of 40 data points equalled or exceeded 0.10 mg/l. This may be due to periodic runoff from surrounding uplands, which are categorized as improved pasture used for cattle grazing. The open water site, while showing a similar range of values, experienced 9 of 40 data points where total phosphorus equalled or exceeded 0.10 mg/l. Although significant differences were documented in total phosphorus concentrations, levels of orthophosphorus were relatively low and stable,

Table 1. Mean values* for water quality parameters.

PARAMETERS	T-1	T-2	T-3	T-4
DEPTH (m)	1.04	1.39	1.52	1.94
pH[2]	6.0-7.9	6.0-7.9	6.4-8.6	6.5-9.1
CONDUCTIVITY (umhos/cm)	148	148	147	146
TOTAL ALKALINITY (as $CaCO_3$)	24	22	23	23
UNFILTERED TURBIDITY (NTU's)[1]	14.1	15.9	17.6	25.4
FILTERED TURBIDITY (NTU's)	10	10	8	8
SECCHI DISC (m)[1]	----	1.1	0.9	0.6
CALCIUM	10.6	10.7	10.6	10.9
MAGNESIUM	3.4	3.5	3.5	3.5
SODIUM	12.3	12.3	12.1	12.1
POTASSIUM	1.8	1.8	1.9	2.0
SULFATE[1]	6.2	8.7	9.6	10.2
NITRATE NITROGEN	0.02	0.05	0.03	0.03
AMMONIA NITROGEN	0.21	0.20	0.20	0.17
TOTAL ORGANIC NITROGEN	1.40	1.37	1.51	1.68
DISSOLVED ORGANIC NITROGEN	1.27	1.28	1.16	1.14
PARTICULATE ORGANIC NITROGEN	0.31	0.28	0.45	0.68
TOTAL PHOSPHORUS (as P)[1]	0.05	0.05	0.06	0.09
ORTHOPHOSPHORUS (as P)	0.01	0.01	0.01	0.01
CHLOROPHYLL a (ug/l)[1]	7.6	13.9	18.7	28.9
PHAEOPIGMENTS (ug/l)	9.6	8.9	16.6	14.1

* Data expressed as mg/l unless otherwise noted

[1] Statistically significant differences between transects at $P<0.05$.

[2] Expressed as a range.

Table 2. F-test values for statistical differences between means at various locations.

	LOCATIONS TESTED					
PARAMETERS	T-1,T-2	T-1,T-3	T-1,T-4	T-2,T-3	T-2,T-4	T-3,T-4
CHLOROPHYLL a	1.267	13.717[a]	59.398[a]	5.638[a]	40.936[a]	16.483[a]
UNFILTERED TURBIDITY	0.560	4.514[a]	52.280[a]	1.902	42.200[a]	26.400[a]
TOTAL PHOSPHORUS	0.015	3.585	35.089[a]	4.060[a]	36.523[a]	16.477[a]
SULPHATE	18.87[a]	30.613[a]	35.723[a]	1.415	2.889	0.279
SECCHI DISC TRANSPARENCY	----	----	----	4.714[a]	40.526[a]	17.875[a]

[a] Statistically significant at $P<0.05$

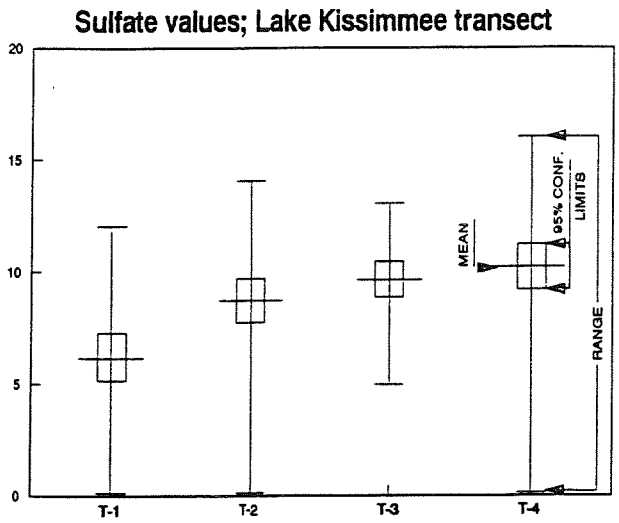

Figure 2. Mean sulfate concentrations (mg/l) for each transect point, 1977-86.

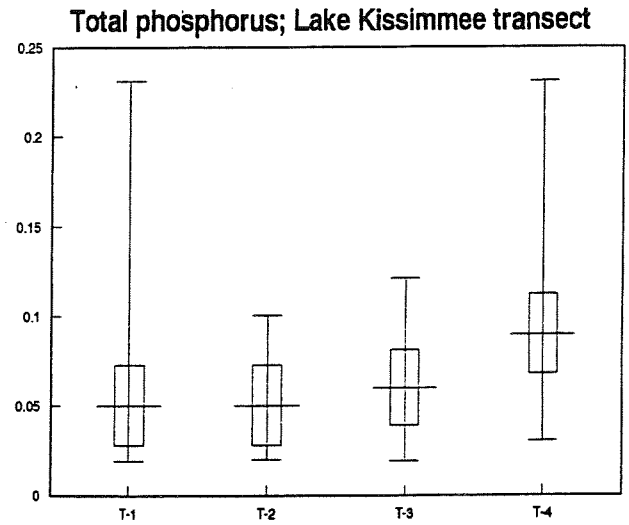

Figure 3. Mean total phosphorus concentrations (mg/l as elemental phosphorus) for each transect point, 1977-86.

0.01 mg/l for all locations (Table 1).

Analysis of seasonal variations in total phosphorus concentrations indicate that no significant temporal differences ($P < 0.05$) occurred at any sample locations. Orthophosphorus concentrations did not show significant seasonal differences (Table 3).

CHLOROPHYLL a - Mean chlorophyll a values increase from T-1 through T-4 (7.6, 13.9, 18.7, and 28.9 µg/l respectively), with the greatest increase occurring between T-3 and T-4 (Fig. 4). All littoral sites have mean values significantly lower than open water. Within the littoral zone, significantly higher mean values were found at T-3 when compared to T-1 and T-2, while the mean for T-2 was intermediate and not significantly different from T-1 (Table 2).

Although T-2 shows a wide upper range (Fig. 4), in fact only one sample in 40 exceeded 35 µg/l, while seven samples exceeded this value at T-4. If the one atypical value (96.2 µg/l) was deleted from the T-2 data base, the mean value could have been lowered by 2.1 µg/l, to 11.8 µg/l; and confidence limits would have been substantially reduced.

Temporal changes in chlorophyll a values (Table 3) indicated that for T-1 significant differences ($P < 0.05$) were found between fall and winter, and between fall and spring. One significant difference was found at T-2, between fall and spring. Significant differences were noted at T-3 between fall and winter, fall and spring and fall and summer; significant differences were found at T-4 between fall and winter.

Chlorophyll a is used as an indicator of algal biomass. Some authors have noted that submerged aquatic macrophytes in lake systems may reduce the phytoplankton production (Hasler and Jones 1949). Canfield et al. (1984) found that in Lake Pearl, Florida chlorophyll a concentrations increased as the percentage of the lakes total volume infested with aquatic macrophytes decreased. These findings lend support to our quantitative assessment of this parameter.

UNFILTERED TURBIDITY - Unfiltered turbidity values were significantly higher in open water than in

Table 3. F-test values for seasonal variations in water quality parameters at various locations.

PARAMETERS	SEASONS TESTED	SAMPLE LOCATIONS T-1	T-2	T-3	T-4
TOTAL PHOSPHORUS	FALL, WINTER	0.221	0.888	2.177	0.439
	FALL, SPRING	0.010	1.553	0.057	0.153
	FALL, SUMMER	1.066	0.171	0.068	0.015
	WINTER, SPRING	0.309	0.110	1.423	0.052
	WINTER, SUMMER	0.333	0.251	2.861	0.551
	SPRING, SUMMER	1.211	0.655	0.235	0.235
ORTHO-PHOSPHORUS	FALL, WINTER	0.237	0.420	0.372	1.863
	FALL, SPRING	0.007	0.253	0.483	0.159
	FALL, SUMMER	2.391	0.416	0.088	0.102
	WINTER, SPRING	1.151	1.282	0.011	0.770
	WINTER, SUMMER	1.152	1.623	0.088	2.547
	SPRING, SUMMER	2.020	0.019	0.151	0.458
UNFILTERED TURBIDITY	FALL, WINTER	0.973	1.017	0.401	3.261
	FALL, SPRING	0.081	0.015	0.158	0.345
	FALL, SUMMER	1.191	1.875	12.309[a]	2.066
	WINTER, SPRING	0.453	0.733	0.047	1.214
	WINTER, SUMMER	1.018	0.153	8.374[a]	0.063
	SPRING, SUMMER	0.616	1.469	9.144[a]	0.642
CHLOROPHYLL a	FALL, WINTER	4.036[a]	1.101	4.132[a]	5.051[a]
	FALL, SPRING	5.232[a]	3.973[a]	9.434[a]	3.614
	FALL, SUMMER	2.060	1.317	8.924[a]	0.180
	WINTER, SPRING	0.114	0.952	1.208	0.041
	WINTER, SUMMER	0.264	0.017	1.031	2.817
	SPRING, SUMMER	0.686	0.675	0.007	1.940
SECCHI DISC	FALL, WINTER	---	0.639	0.981	3.426
	FALL, SPRING	---	0.705	1.833	0.780
	FALL, SUMMER	---	0.302	6.418[a]	0.525
	WINTER, SPRING	---	0.004	0.155	0.719
	WINTER, SUMMER	---	0.051	2.472	1.014
	SPRING, SUMMER	---	0.080	1.314	0.022
SULPHATE	FALL, WINTER	7.438[a]	2.215[a]	0.497	0.708
	FALL, SPRING	7.859[a]	1.980	3.220	0.951
	FALL, SUMMER	0.101	0.768	0.028	0.044
	WINTER, SPRING	0.025	0.001	1.233	0.035
	WINTER, SUMMER	5.423[a]	0.322	0.723	0.993
	SPRING, SUMMER	5.838[a]	0.266	3.631	1.248

[a] Statistically significant at $P<0.05$

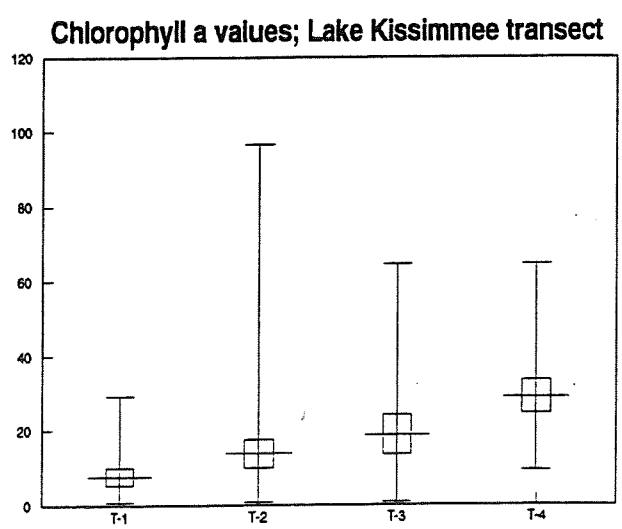

Figure 4. Mean chlorophyll a values (ug/l) for each transect point, 1977-86.

the littoral zone (Fig. 5). The mean value for T-4 (25.4 NTUs) was significantly greater than for T-1, T-2, and T-3 (14.1, 15.9, and 17.6 NTUs, respectively). While no significant differences were noted between T-1 and T-2 or between T-2 and T-3, there was a significant difference between T-1 and T-3 (Table 2).

The upper levels of the ranges at T-1 and T-2 reflect a few atypical samples. Both T-1 and T-2 exhibited a single occasion when unfiltered turbidity exceeded 20 NTUs after March 1978, immediately after the lake had refilled following the extreme drawdown of 1977. Excluding the data points taken prior to March 1978 would have reduced the mean values at T-1 from 14.1 to 12.3 NTUs and at T-2 from 15.9 to 13.8 NTUs. These reductions would have been accompanied by a decrease in confidence limits at both sites.

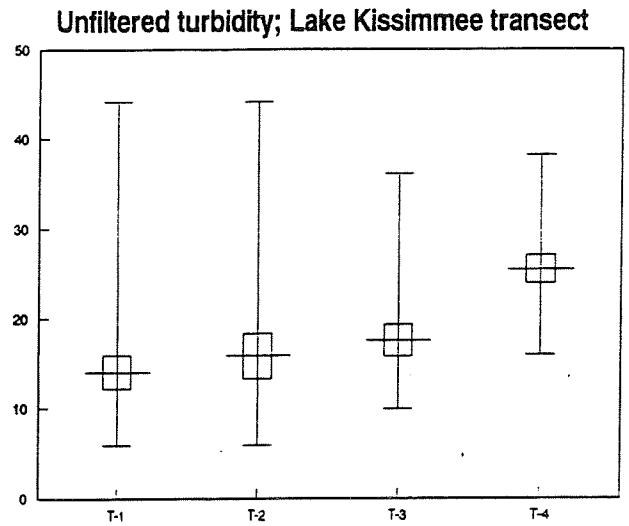

Figure 5. Mean unfiltered turbidity values (NTU's) for each transect point, 1977-86.

Unfiltered turbidity values showed no temporal significant differences ($P < 0.05$) at T-1, T-2 and T-4. Significant differences were found at T-3 occurring between summer and the other three seasons (Table 3).

SECCHI DISC TRANSPARENCY - Secchi disc values (Fig. 6) reflected improved water clarity in the lakes littoral zone. There was a significant difference between T-2 (1.09 m) and T-3 (0.89 m) in the littoral zone. The mean open water transparency (T-4) was significantly reduced (0.64 m), and the upper range of visibility was markedly lower (Table 2).

Data for T-1 is excluded, since the Secchi was most often visible on the lake bottom. Had water depths at T-1 been great enough to allow accurate measurement of light extinction depth, it would likely have been greater than that found at T-2, due to the fact that unfiltered turbidity values indicate less suspended particulate matter present at T-1. The light extinction coefficient was significantly reduced in open water.

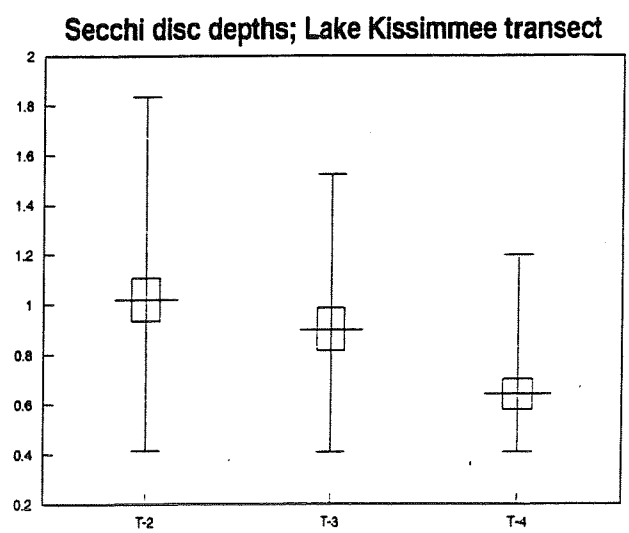

Figure 6. Mean Secchi disc values (m) for each transect point, 1977-86.

No significant seasonal differences ($P < 0.05$) were noted for T-2 and T-4. One significant difference was found between fall and summer data at T-3.

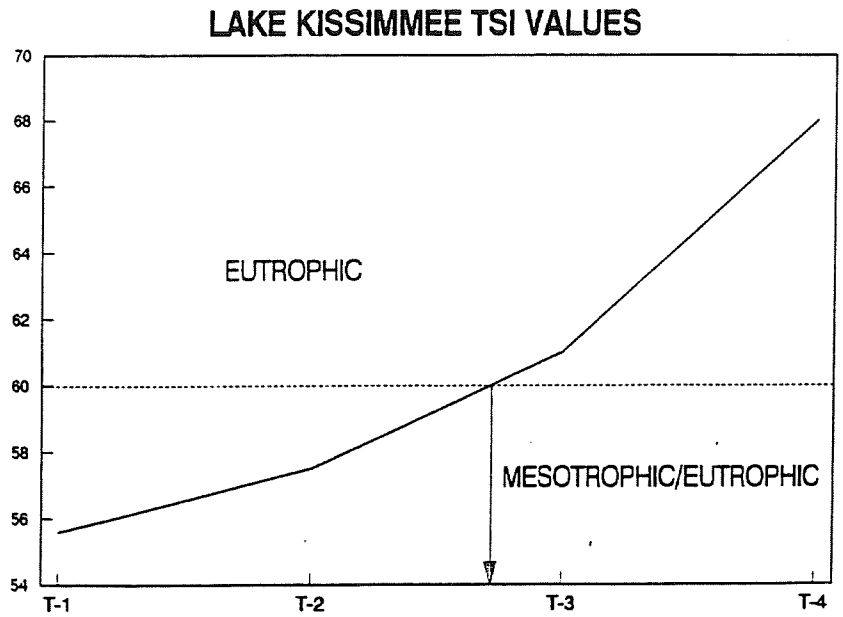

Figure7. Mean Florida Trophic State Index (TSI) for each transect point.

DISCUSSION

Trophic state indices were developed for each site using equations developed for Florida lakes by Huber et al. (1982). These values, along with the calculations used to derive them, are presented in Table 4. TSI values for location T-1 were computed from data obtained when Secchi disc transparency was measured. Average TSI values are graphically represented in Figure 7.

It is noteworthy that TSI average values for location T-1 (56) and T-2 (57) fall within the range considered to define mesotrophic or mesotrophic/ eutrophic systems (TSI AVE range 50-60). T-3 (61) indicates borderline eutrophic status, and T-4 (68) is moderately eutrophic (TSI range 60-80; Fig. 7).

Statistically significant lower total phosphorus and chlorophyll a levels in vegetated areas suggest that a large amount of phosphorus in the littoral water column is contained in rooted aquatic plant biomass, rather than periphyton or free-floating algae. Elevated pH ranges at T-3 and T-4, which likely indicate increased algal production, tend to support this observation. Other authors have found that nutrient and chlorophyll a concentrations can be low in waters with an abundance of macrophytes (Canfield et al. 1983). Lower phaeopigment and particulate organic nitrogen levels (Table 1) at nearshore and mid-littoral locations also suggest that phosphorus is being tied up in biomass other than periphyton. Likely candidates for this include not only vascular aquatic plants but also invertebrate and fish communities. Bottom sediments may also serve as a sink for particulate phosphorus, since these do not resuspend as readily in littoral areas as they do in open water, being subjected to far less wave and current action.

Habitats produced by rooted emergent and submerged aquatic plants, in terms of both physical structure and their attached periphyton communities, have proven in previously cited studies to play a vital role in the production of aquatic invertebrates and healthy well-balanced fish populations. The significant improvements in important water quality parameters documented in this study might serve to sustain or enhance this productivity by providing a level of stability in water quality within the littoral zone of a lake. This stability would be

most important in a eutrophic system where algal blooms, generally defined in Florida as having chlorophyll a values greater than 40 µg/l, create wide diurnal fluctuations in dissolved oxygen values which can cause direct mortality of both fish and susceptible benthic invertebrates. Littoral plant communities could, in extreme instances, serve as refuge areas for fish during times of limnetic oxygen depletion caused by algal blooms. The fact that TSI AVE values in the littoral zone are substantially lower than for open water, and in the case of T-1 and T-2 actually reflect decreased productivity, tends to add credibility to this assumption.

Management of aquatic vegetation in Florida lakes has been accomplished by herbicide treatments, mechanical removal or biological control. This management has ranged from periodic reductions in plant volume to complete eradication of aquatic vegetation. A considerable body of both published and unpublished information exists concerning changes in fisheries and water quality in Florida lakes which have been subjected to eradication of aquatic plants through the use of grass carp (*Ctenopharyngodon idella*; Leslie et al. 1983, Shireman et al. 1984). Some study lakes have undergone measurable deterioration in water quality and fisheries, while others have reportedly shown little change. At present, the emphasis on plant management using triploid grass carp is centered around the concept of herbicide treatment of problem vegetation followed by stocking the minimal number of grass carp required to maintain long-term control over problem plants (e.g., *Hydrilla verticillata*) while leaving desirable species intact.

Urban lake management programs are also beginning to undergo radical changes in reference to littoral zones. In the past, the emphasis of these programs was to eliminate all aquatic vegetation to

Table 4. Trophic State Index values and equations*; Lake Kissimmee, Florida.

	Transect Locations			
	T-1	T-2	T-3	T-4
TN/TP	27	27	25	20
TSICHLa	46	55	59	65
TSISD	62	59	63	73
TSITP	69	69	73	81
TSITN	67	66	68	71
TSINUTR	59	58	61	65
TSIAVE	56	57	61	68

* Equations from Huber et al. 1982

TSICHLa = 10 X (1.68 + 1.44 X LN (CHLa))
TSISD = 10 X (6.0 - 3.0 X LN (SD))
TSITP = 10 X (2.36 X LN (TP X1000) - 2.38)
TSITN = 10 X (5.96 + 2.15 X LN (TN))
TSITNB = 10 X (5.6 + 1.98 X LN (TN))
TSITPB = 10 X (1.86 X LN (TN))
TSINUTR = 0.5 X (TSITNB + TSITPB)
TSIAVE = 1/3 X (TSICHLa +TSIDS + TSINUTR)

TN/TP = Ratio of TN to TP
TSICHAa = TSI based on Chlorophyll a
TSISD = TSI based on Secchi Disc transparency
TSITP = TSI based on total phosphorus
TSITN = TSI based on total nitrogen
TSITNB = TSI based on total nitrogen budget
TSITPB = TSI based on total phosphorus budget
TSINUTR = TSI based on total nutrient budget
TSIAVE = Average TSI

TSIAVE:
< 40 = oligotrophic
40 - 50 = oligotrophic/mesotrophic
50 -60 = mesotrophic/eutrophic
60 - 80 = eutrophic
> 80 = hypereutrophic

provide "clean" shoreline areas. The City of Orlando, Florida has abandoned this approach in favor of eliminating problem species, and subsequently replanting littoral areas with aesthetically pleasing and ecologically valuable native plants. This program has met with widespread public and political support, and hopefully will be adopted by other communities.

CONCLUSIONS

While there is still no consensus of opinion on the impact of aquatic plants on lakewide water quality, this study demonstrates that several important water quality parameters as well as trophic state indices are significantly improved within a lakes littoral zone compared to limnetic areas. While these parameters showed significant changes at a location 50 m from open water, we did not attempt to "fine tune" exactly how far within the littoral zone these changes began to take place. The width of littoral zone required to obtain significant changes in water quality will vary depending on other factors such as lake water quality and trophic state, basin morphometry, size, direction and strength of currents, wave action, plant species present and their densities.

Littoral and limnetic zones with their associated habitats exhibit distinct characteristics which have direct effects on the physical, chemical, and biological aspects of a lake system. Efforts to eradicate or greatly reduce aquatic plant communities through herbicide applications or physical removal could cause deleterious changes to the biota and water quality of the overall lake system. Until the impacts of these factors are better understood, the known and accepted values of aquatic plant communities mandate that they be managed with care to ensure perpetuation of the physical, chemical, and biological benefits they produce.

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Epiphytic Invertebrate Populations on *Hydrilla verticillata* and *Egeria densa* versus Native Submersed Macrophytes

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Abstract

Macroinvertebrates were quantitatively sampled from the exotic macrophytes *Hydrilla verticillata* and *Egeria densa*, and from the native macrophytes *Sagittaria kurziana*, *Vallisneria americana*, and *Potamogeton illinoensis* in a spring-fed, coastal plains river. Total invertebrate density and taxa composition were compared among macrophyte species. Total invertebrate densities were consistently higher ($P<0.05$) on *Sagittaria* compared to densities for other species. The most common invertebrate taxa collected were Chironomidae, Trichoptera, Ephemeroptera, Gastropoda, and Amphipoda. Chironomidae were the most dominant taxa on all plant species and were responsible for higher total numbers of invertebrates on *Sagittaria*. Temporal trends were evident; lowest invertebrate densities occurred during January and February. From plant surface area estimations, it was apparent that surface area was not the cardinal factor in determining the abundance of plant colonizing invertebrates.

Introduction

The macroinvertebrates that colonize and live on macrophytes are numerous (Schramm et al. 1987, Cyr and Downing 1988) and can be more abundant than benthic invertebrates (Needham 1929, Soszka 1975). They are prey to fish and waterfowl (Mittelbach 1984, Keast 1985) and are important in energy transfer (Dall et al. 1984, Kairesalo and Koskimies 1987), especially in aquatic systems with substantial macrophyte growth.

Some macrophyte species have been found to harbor more invertebrates than others (Gerrish and Bristow 1979, Rooke 1986, Cyr and Downing 1988). Krecker (1939) proposed that finely dissected macrophytes, offering more surface area, would have higher invertebrate densities than simple structured, broad leafed macrophytes. Several studies have supported this contention (Rosine 1955, Gerking 1957, Mrachek 1966, Dvorak and Best 1982, Rooke 1986, Schramm et al. 1987), but some early studies (Bownik 1970, Krull 1970) and a recent, well replicated study (Cyr and Downing 1988) found that total abundance of invertebrates was not systematically related to the degree of plant dissection.

Exotic plants are now present in a majority of Floridas freshwater systems (Schardt and Schmitz 1990). Control measures reduce plant biomass, but because of their high resilience, exotics are still

an important component in plant communities. Thus, for fish and waterfowl management, it is useful to determine invertebrate populations associated with exotic macrophytes.

The purpose of this study was to compare epiphytic invertebrate populations on two exotic versus three native submersed macrophytes in a spring-fed coastal plains river. The importance of plant surface area to invertebrate abundance was also assessed.

Methods

Samples of plants and associated invertebrates were collected from the St. Marks River, located in a karst area of Wakulla County (30° 16' N, 84° 08' W). Sampling was conducted in the basin area (Fig. 1), a relatively wide (130 m) and shallow (1 m) portion where the river re-emerges from underground (Bartodziej, in press). Submersed plants abound in this part of the river and often reach the surface.

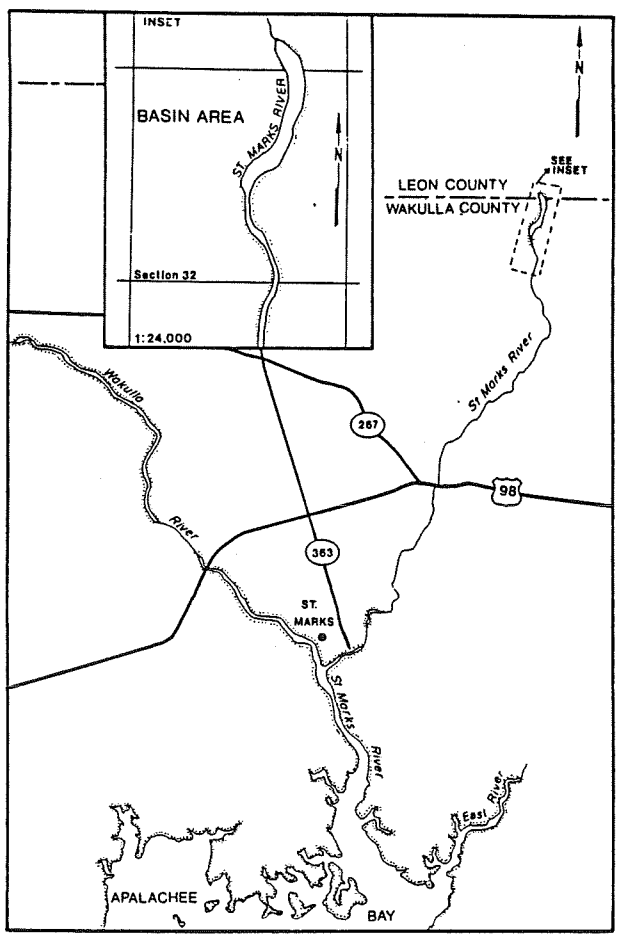

Figure 1. Map of St. Marks River

Monotypic stands of exotic, *Hydrilla verticillata* and *Egeria densa*, and native, *Sagittaria kurziana*, *Potamogeton illinoensis*, and *Vallisneria americana*, plants were sampled monthly from August, 1989 to May, 1990 (Fig. 2). For each species, five randomly chosen stands were sampled each month. Stands that were sampled and of the same species were at least 50 m apart. A watertight, hinged plastic box sampler (2 l) was placed under plant stems within a stand and securely closed (similar to Downing 1986). Stems protruding outside the box were clipped. Contents were emptied into a U.S. Standard number 45 plastic sieve bucket, consolidated, and then bagged in 70 percent isopropanol. Sampling depth ranged from 0.5 m to the surface. In the laboratory, each piece of plant material was thoroughly rinsed over a sieve to remove attached invertebrates. After washing, plants were blotted to absorb excess water and weighed to the nearest 0.1 g. Invertebrates were identified, usually to order, and counted under a dissecting microscope (15x).

Fifteen samples of each species were set aside after weighing to determine plant surface area. On *Hydrilla*, *Egeria*, and *Potamogeton*, leaves were removed from stems and photocopied. Stems were sliced into cross-sections and measured with an ocular micrometer. Plant stem surface areas were computed geometrically. Photocopies of *Vallisneria* and *Sagittaria* and leaves from other species were digitized for each sample. Wet weight to surface area regressions were calculated for each species.

Invertebrate abundances were expressed as total number per 50 g wet weight of plant and total number per 2000 cm^2 of plant surface area. Analysis of variance tests were performed on log_{10} transformed data. Transformations were necessary because of nonconstant variance; means were compared using a Tukeys test.

Water samples were collected at four stations in the basin area. Total nitrogen and phosphorus analysis followed standard techniques (APHA 1985). Water temperature, specific conductance, and oxygen were measured with YSI field meters.

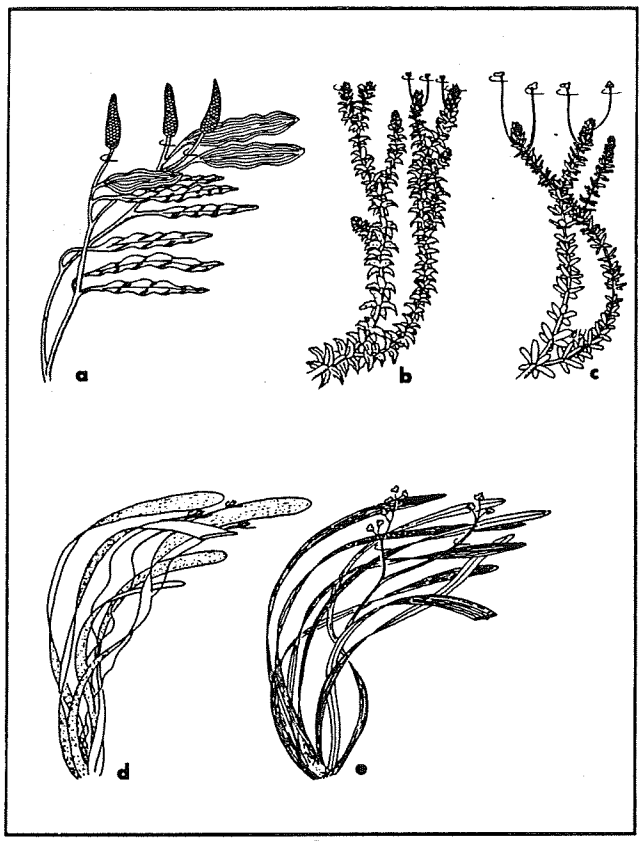

Figure 2. Submersed macrophyte species sampled from the St. Marks River. (a) *Potamogeton illinoensis*, **(b)** *Hydrilla verticillata*, **(c)** *Egeria densa*, **(d)** *Vallisneria americana*, **(e)** *Sagittaria americana*.

Results and Discussion

Water quality was typical of a spring-fed river in this region (Fernald and Patton 1984). Water exiting the spring was nutrient rich, highly alkaline, and relatively cool (Table 1). Water was oxygen poor at the spring head, but concentrations increased ($P<0.05$) down-river. During baseflow conditions, water clarity was high. But shortly after a substantial rain, surrounding wetlands contributed tannic water to the spring. At such times, the input of acid swamp water caused water clarity and alkalinity to decrease and total nitrogen to increase.

Abundances of epiphytic invertebrates differed among plant species. From August through January, more invertebrates were associated with *Sagittaria*. Although variances were large, total numbers of invertebrates on *Sagittaria* were found to be consistently higher ($P<0.05$) than densities on other species (Table 2). For this period, five replicates for each species were used to generate means. However, because of time constraints, three out of five samples were processed for February through May. Variances were generally higher than those around means for previous months; differences in invertebrate abundance among plant species were not evident ($P>0.05$).

The most common invertebrate taxa collected were Chironomidae, Trichoptera, Emphemeroptera, Gastropoda, and Amphipoda. Chironomids were the most dominant taxa, and were responsible for the higher total numbers of invertebrates on *Sagittaria*. The percent composition of chironomids for August was over 68 percent for all plant species (Table 3). During winter, the chironomid population decreased, while generally, populations of other taxa also decreased, but to a lesser degree. The percent composition of chironomids decreased in December for all plant species. In spring, all taxa increased in abundance, but especially chironomids; percent composition values were similar to those for August. That chironomids were the most abundant epiphytic taxa is consistent with findings from other studies (Gerking 1957, Martin and Shireman 1976, Schramm et al. 1987).

The abundance of certain invertebrate taxa differed ($P<0.05$) among plant species. However, consistent temporal trends in densities were not detected. Balciunas and Minno (1984) found a gastropod species (*Elimia floridensis*) to be the most abundant taxa on *Hydrilla* in the Wacissa River, 20 km east of the St. Marks River. Martin and Shireman (1986) also collected relatively high numbers of gastropods on *Hydrilla* in a lake. *Elimia floridensis* was abundant on *Hydrilla* in the St. Marks River, but densities were not consistently higher ($P<0.05$) than densities on other species. Schramm et al. (1987) detected more amphipods (*Hyalella*) on finely divided submersed plants than on simple structured plants. During fall, amphipods

Table 1. Ranges for selected water quality parameters, August 1989 through May 1990. All stations were located in the basin area of the St. Marks River; Station 1 was at the spring head, Station 2 = 0.1 km, Station 3 =1.2 km, and station 4 = 2.6 km below the spring.

Location	Temperature (C)	Dissolved Oxygen (mg/l)	Total Nitrogen (mg/l)	Total Phosphorus (ug/l)	Conductance (umhos/cm)
Spring Head	16.9 - 22.0	1.4 - 3.7	0.16 - 0.45	43 - 93	133 - 424
Station #2 (0.1 km)	17.0 - 21.9	1.8 - 3.7	0.14 - 0.29	44 - 75	133 - 298
Station #3 (1.2 km)	17.0 - 22.8	3.3 - 5.4	0.13 - 0.33	49 - 72	136 - 398
Station #4 (2.6km)	16.4 - 22.4	3.4 - 5.7	0.15 - 0.38	51 - 78	133 - 271

Table 2. Mean epiphytic invertebrate density (total number per 50 gram wet weight macrophyte) on five submersed plant species collected from the St. Marks River, August 1989 through May 1990. Standard deviations are in parenthesis. For each month means with the same letter sesignator are not different (P>0.05). ANOVA and Tukeys mean comparison tests were performed on transformed (log_{10}) data.

Month	Hydrilla verticillata	Egeria densa	Potamogeton illinoensis	Vallisneria americana	Sagittaria kurziana
August	294 (345) a	480 (295) a	288 (199) a	61 (60) b	491 (185) a
September	263 (206) b	608 (865) ab	666 (426) ab	263 (185) b	2529 (2681) a
October	166 (104) a	239 (180) a	243 (120) ab	104 (38) a	840 (710) b
November	90 (73) c	141 (140) bc	375 (24) ab	164 (170) bc	887 (325) a
December	68 (53) a	84 (58) a	270 (240) a	121 (61) a	273 (198) a
January	66 (38) a	56 (17) a	48 (56) a	25 (20) a	316 (225) b
February	24 (16) a	26 (18) a	113 (100) a	34 (46) a	329 (443) a
March	197 (222) a	494 (494) a	361 (241) a	570 (692) a	795 (189) a
April	1066 (923) a	1552 (601) a	914 (193) a	327 (351) a	1007 (478) a
May	1729 (1793)	4035 (2977)	1511 (1589) a	1356 (636) a	2394 (2044) a

(*Gammarus* and *Hyalella*) were more abundant (P<0.05) on *Hydrilla, Egeria*, and *Potamogeton* than on the simple structured *Vallisneria* and *Sagittaria*, supporting Schramms results. But during other periods, amphipod abundance was similar on dissected and simple structured species.

Invertebrate abundance varied temporally, even though water temperature was buffered by spring inputs and did not change seasonally to a great degree (Table 1). Total numbers of invertebrates decreased on all species from late summer to winter and then increased in spring (Fig. 3). All taxa were lowest in number during January and February. Temporal differences in epiphytic invertebrate abundance have been reported by others (Gerking 1957, Krull 1970, Schramm et al. 1987). Invertebrate life cycle patterns are influenced by physical conditions (i.e., temperature, photoperiod), food availability, and competition and predation (Merritt and Cummins 1984). We noticed in the field, and when washing plants, that the abundance of periphytic filamentous algae decreased substantially in winter. Common Chironomidae (*Dicrotendipes*), Trichoptera (Hydroptilidae) and Ephemeroptera (*Baetis*) collected are functionally able to consume algae (Merritt and Cummins 1984). It is plausible that the abundance of certain invertebrate taxa was correlated with fluctuations in epiphytic algal biomass.

Surface area per unit plant weight varied among species. *Egeria* had the highest surface area and *Hydrilla, Vallisneria*, and *Sagittaria* had similarly lower surface areas (Table 4). Although *Egeria* and *Hydrilla* were structurally similar, *Egeria* had larger leaves and a higher density of leaves per stem, which accounted for the difference in surface area between species. Expressing total number of invertebrates by plant surface area decreased abundance values for *Egeria* and *Potamogeton* relative to the other species (Fig. 3). Again, invertebrate densities were consistantly higher (P<0.05) on *Sagittaria* than other species during the first half of sampling.

These data do not support the hypothesis (Krecker 1939) that invertebrate abundance is positively correlated with plant dissection. The simple structured *Sagittaria* had more epiphytic invertebrates

Table 3. Percent community composition of epiphytic invertebrates on macrophytes for August and December 1989, and May 1990.

Taxon	August					December					May				
	Hyd	Ege	Pot	Val	Sag	Hyd	Ege	Pot	Val	Sag	Hyd	Ege	Pot	Val	Sag
Chiron.	69	91	80	79	92	13	44	11	18	61	88	94	93	88	89
Trich.	1	1	1	3	0	15	1	13	34	20	5	1	2	4	7
Ephem.	22	3	11	8	7	39	13	65	32	14	5	3	3	4	2
Gastro.	1	2	6	8	1	13	5	2	1	0	1	1	0	1	1
Amphi.	1	2	2	2	0	10	28	2	1	1	0	0	0	0	0
Other	6	1	0	0	0	10	9	7	14	4	1	1	2	3	1

Chiron. = Chironomidae; Trich. = Trichoptera; Ephem. = Ephemeroptera; Gastro. = Gastropoda; Amphi. = Amphipoda; Other = other taxa.

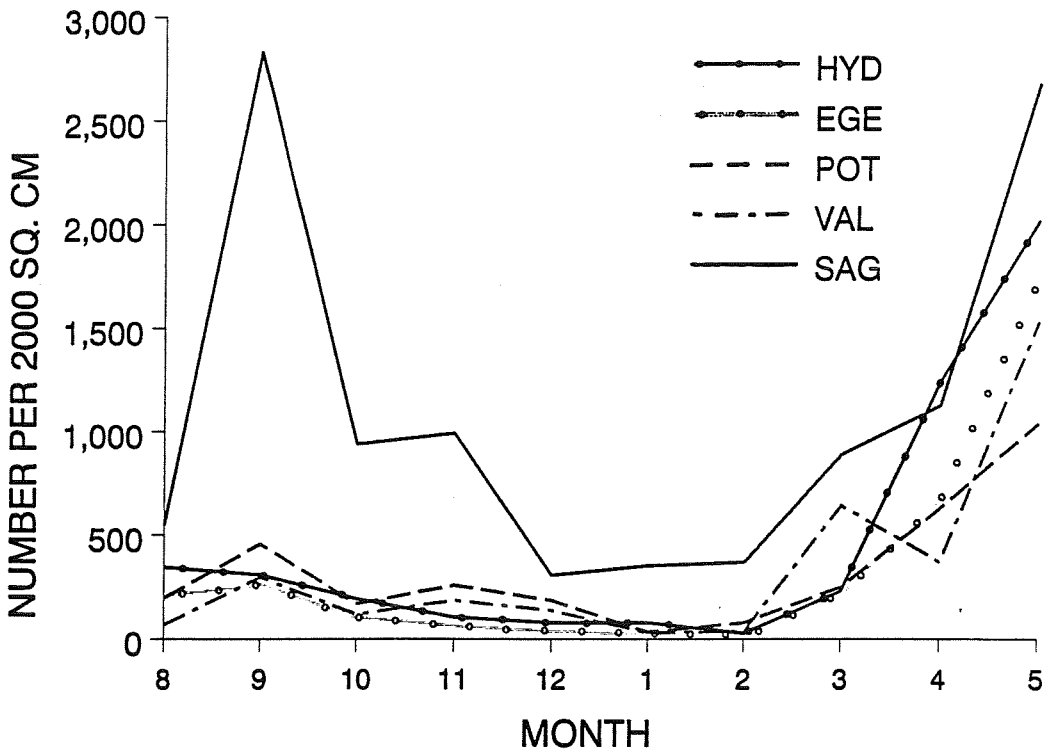

Figure 3. Mean density (total number per 2000 cm^2 of plant surface area) of epiphytic invertebrates on five submersed macrophytes, August 1989 through may 1990.

than *Egeria*, which was most dissected and provided the greatest amount of colonizable surface area. Apparently, surface area was not the cardinal factor in determining the abundance of plant colonizing invertebrates in the St. Marks River. Other factors such as periphytic algal abundance and community composition, plant morphology, surface texture, nutrient content of plant tissue, and defensive chemicals can also influence invertebrate abundance (Cyr and Downing 1988). It is plausible that differences in one or more of these factors accounted for the relatively high numbers of invertebrates on *Sagittaria*.

Knowledge of plant-invertebrate associations may be useful in the management of game species. Most invertebrate taxa colonizing plants can be considered food for fish and waterfowl (Moyle 1961, Gerking 1962, Fairchild 1983, Mittlebach 1984, Drobney and Fredrickson 1985). Macrophytes with relatively high numbers of invertebrates may be able to sustain larger populations of fish and waterfowl (Cyr and Downing 1988). However, this is assuming: (1) that predators are able to prey on invertebrates equally well on all plant species; (2) that plants beneficial to invertebrates are acceptable as fish habitat; and (3) that invertebrate densities reflect invertebrate biomass and production.

In the St. Marks River, invertebrate densities were highest on the simple structured, native macrophyte *Sagittaria kurziana*. Although the exotics, *Hydrilla* and *Egeria*, were more dissected, they did not harbor as many invertebrates. The removal of these exotics would likely increase native plant biomass, including *Sagittaria kurziana*. Unless the assumptions listed above prove to be seriously inadequate, plant management favoring *Sagittaria kurziana* may be expected to result in increased populations of those fish and waterfowl which feed on aquatic invertebrates.

Table 4. Macrophyte wet weight (WW) to surface area regressions for five submersed plant species. Surface area estimates are listed for 50 g WW material. n=15 for each regression.

Macrophyte	Regression equation	R^2	Area estimate (50 g WW)
Egeria	Area = -13.86 + 93.11 WW	0.78	4642 cm^2
Hydrilla	Area = 42.43 + 33.40 WW	0.76	1712 cm^2
Potamogeton	Area = 28.72 + 57.50 WW	0.96	2904 cm^2
Vallisneria	Area = 147.86 + 32.40 WW	0.92	1768 cm^2
Sagittaria	Area = - 8.39 + 35.86 WW	0.98	1785 cm^2

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The Role of Aquatic Marshland Plants in Reducing Pollutant Loads to a Florida Lake

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Abstract

Aquatic plants have an integral role in the function of lakes. They are principally recognized for providing habitat and food for fish and wildlife, but their role in reducing pollutant loadings in surface waters is now being realized. Marshes, both natural and man-made, provide an efficient means of reducing nutrient and pollutant burdens from reaching lakes. A study of an artificial marsh created on a stream feeding Lake Jackson provided an excellent opportunity to quantify the role of aquatic plants in reducing nutrient loadings. Lake Jackson is a 1215 ha lake located in Leon County, Florida. A 2.5 ha artificial marsh was created in conjunction with a sand filter and impoundment for the purpose of reducing stormwater pollutant loadings to the lake that originated from a highly urbanized area. The marsh was divided into three cells that contain *Typha* sp. in the first, *Scirpus* sp. in the second and *Pontederia* sp. in the third section. Results of isolation chamber experiments revealed that greater rates of nutrient removal occurred in those chambers containing aquatic plants, than either sediments or water alone. Concentrations of nutrients and suspended solids measured at the inflow and outflow of the marsh during one storm event demonstrated that loadings of organic and inorganic solids were reduced by approximately 70 and 77% respectively, and nitrate and phosphorus removal were as high as 64 and 43% respectively. These findings indicated that aquatic plants have the potential to reduce stormwater pollutants reaching Florida lakes, and have the added benefit of serving as habitat for aquatic birds and animals.

Introduction

During the previous decade, nonpoint-source pollution was recognized as a predominant source of contamination affecting surface waters throughout the United States. By the mid 1980s, nonpoint-source pollution was determined to be the primary factor affecting 76% of impaired lake acreage and 65% of assessed river miles designated as impaired (USEPA, 1987; Peterson et al., 1985).

Nonpoint-source pollution may be generally defined as contaminants that originate from one or more diffuse sources. A variety of compounds

can contribute to this pollutional burden such as soil erosion, nutrient, pesticides, metals, pathogenic bacteria and hydrocarbons. Excessive loading of these contaminants can lead to eutrophication, depletion of oxygen, biotic toxicity, bioaccumulation, and in some cases may adversely affect human health. Because of the deleterious effects, nonpoint-source inputs have become a growing concern to lake managers.

Sources of nonpoint pollutants include urban stormwater, agricultural runoff, silvicultural operations, septic tanks, landfills and underground waste disposal facilities. Urban stormwater in particular is a significant contributor and is considered second only to agricultural runoff as the most widespread nonpoint pollution problem throughout the U.S. (USEPA 1987). In Florida, over half of the pollution loads to receiving waters are the result of urban stormwater runoff (Livingston and Cox 1985).

Urbanization alters natural hydrology and causes changes in total runoff, peak flow volume and the chemical quality of stormwater (Leopold 1968), and has created a need for construction of Stormwater Management Systems (SMS). A SMS provides many benefits including: 1) flood control, 2) erosion and sedimentation control, 3) surface drainage, 4) reduction of pollutants in runoff, and 5) aesthetic amenities. To ensure that the hydrology and quality of runoff after urbanization is similar to that occurring before development, SMS are designed to affect the retention and filtration of stormwater, and typically include an impoundment to regulate stormwater hydrology and permit settling of suspended material, and a wetland to remove dissolved nutrients.

After extensive experimentation throughout the U.S., wetlands are now recognized as an integral component in the treatment of point source pollutants (USEPA 1985). Their usefulness in treating nonpoint source pollution, however, has not been fully realized and in fact some concern has been expressed about the ability of wetlands to effectively treat stormwater runoff (Whalen and Cullum 1988). Some studies have provided insight on the effectiveness of wetlands in reducing stormwater pollutant loadings to lakes (Brown 1985, Hickok et al. 1977) and indicated that aquatic plants are instrumental in reducing suspended solids, nutrients and other types of pollutants.

In an effort to quantify wetland systems for contaminant removal, a study was conducted in north Florida to examine the effectiveness of an artificial marsh for stormwater treatment. The purpose of this paper is to briefly present some of the findings of this study. A more detailed analysis will be published elsewhere (LaRock and Johengen 1991).

Study Area

Lake Jackson is a 1215 ha lake located in Leon County, Florida. Designated as a Class III surface waterbody (Florida Administrative Code 17-3), the lake provides recreational amenities to the citizens of Tallahassee and Leon County. Urban stormwater represents the principal type of pollution entering the lake predominantly through the southern and western portions of its watershed.

Comprehensive studies of the water quality of Lake Jackson begun by Harris and Turner (1974), and continued by Turner et al. (1977), Burnett and Donahue (1982) and LaRock (1990) have demonstrated the effect of urbanization on the portion of the lake known as Megginnis Arm A comparison of alkalinity (Fig. 1a) and conductivity (Fig. 1b) observed in Megginnis Arm (Station 5) over a 17 year period indicated that urbanization within the watershed drastically increased these parameters relative to the center of Lake Jackson (Station 9). Stormwater runoff also contributed increased nutrients and sediment loads to the arm, and as water quality declined, the potential of degradation of other parts of the lake became apparent. A SMS was developed in a cooperative effort of the Florida Department of Environmental Regulation (DER) and the Northwest Florida Water Management District (NWFWMD). Funding for the system was provided through the Environmental Protection Agency's Clean Lakes Program and by matching funds and

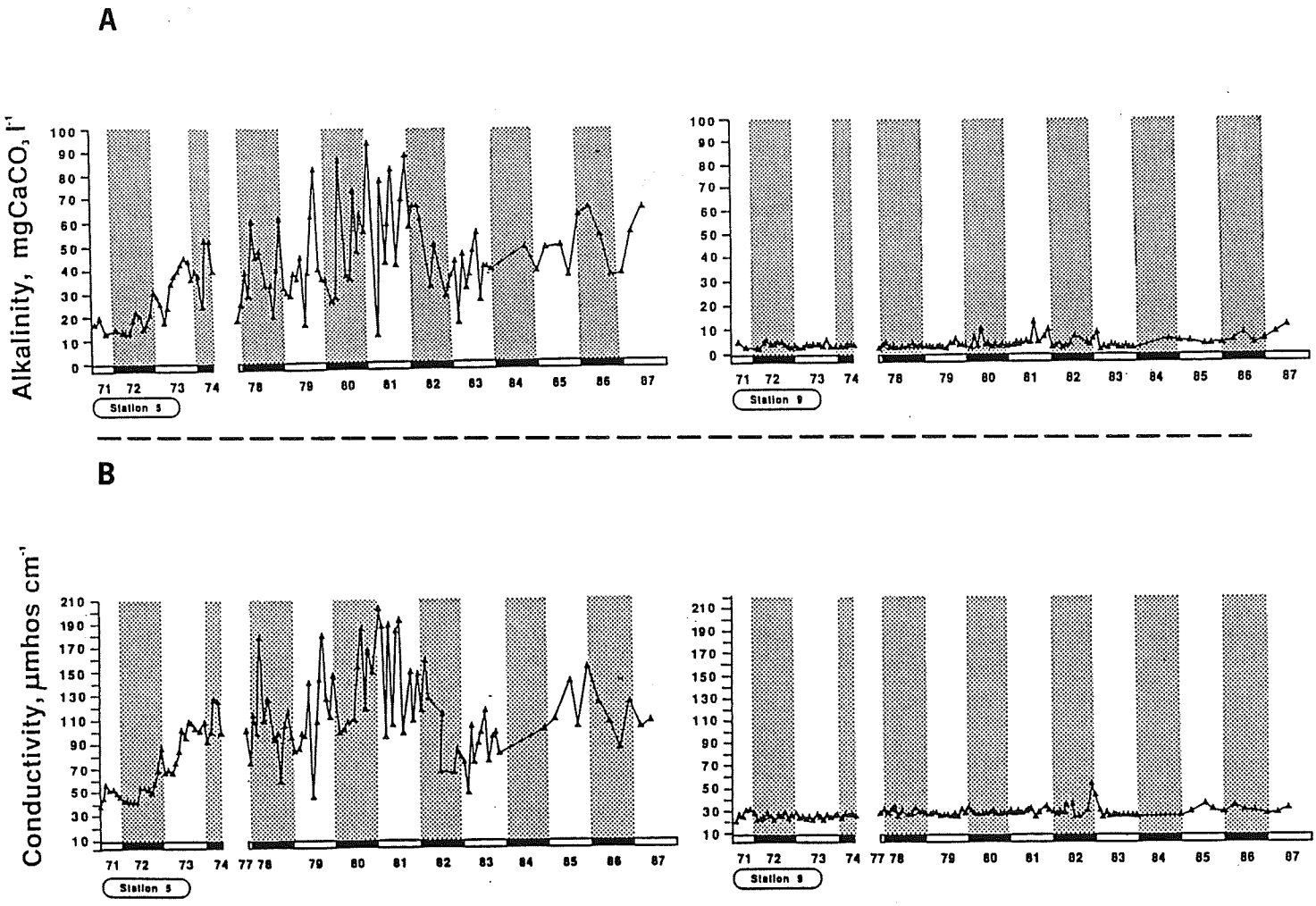

Figure 1. Long-term (a) alkalinity and (b) conductivity measurements in Megginnis Arm (Station 5) and the center of Lake Jackson (Station 9). The early increase in alkalinity and conductivity seen between 1972 to 1974 at Station 5 reflects the effects of interstate 10 construction that was underway one half mile from Megginnis Arm. The sharp decline in alkalinity and conductivity in 1982 is the result of a natural drawdown of the lake that occurred October 25.

services from the NWFWMD.

The SMS is composed of a 163,000 cubic meter impoundment basin, a 1.8 ha intermittent sand filtration filter, and a 2.5 ha artificial marsh (Fig. 2). The wetland component of the system is divided into three cells planted with *Typha* sp. in the first cell, *Scirpus* sp. in the second cell, and *Pontederia* sp. in the third cell. Stormwater runoff flows into the impoundment, through the sand filter, into the marsh, then finally into the lake. During periods of excess runoff, stormwater may be diverted around the filter and directly into the marsh. A detailed description of the various components of this facility, its hydrology, and the efficacy of the components is reviewed by LaRock (1988) and LaRock and Johengen (1991).

Methods

Two different approaches were used to evaluate the efficacy of the marsh system at nutrient removal. Because accurate flow and chemical measurements were made at the inflow and outflow of the system, it was possible to determine mass

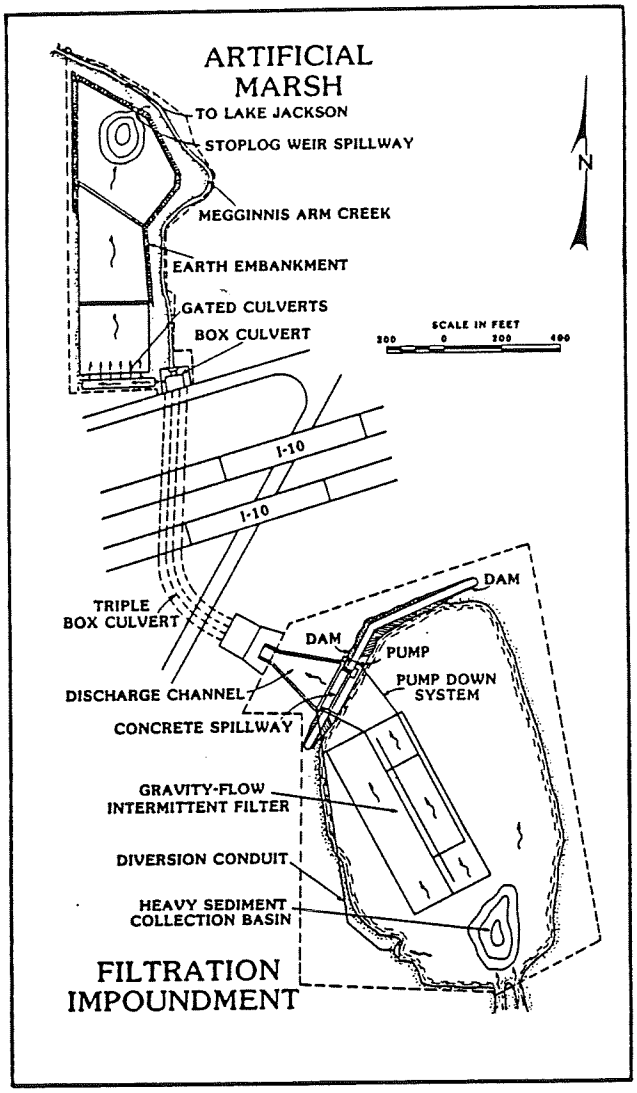

FIGURE 2
The physical plan of the impoundment, filter and artificial marsh to treat stormwater runoff. Inflowing stormwater enters the system at the extreme southern end of the impoundment (bottom of the figure). Heavy particulate material settles out in the 14 ft depression immediately beyond the inflow point, and the water fills the impoundment until the liquid level reaches the elevation of the surface of the filter. When the filter becomes submerged, the stormwater flows through the filter bed, is collected by an underdrain system and flows out through a pipe into a box culvert (the main collection pipe is seen as the line bisecting the filter in the figure). A spillway at the northern end of the impoundment provides overflow capabilities in the event of extreme runoff conditions such as encountered in a 25 year storm. Space limitations necessitated that the marsh be located at the opposite side of Interstate 10, and the box culvert directs the filter effluent and any overflow from the spillway into a distribution chamber that serves to direct the filtered water into the marsh. In instances of extremely high flow, the distribution chamber partitions the water between the marsh and Megginnis Creek which serves as a bypass. Water enters the marsh via a series of gated culverts to assure uniformity of distribution, and then flows through the three cells of the marsh and finally into Lake Jackson. Stoplog weirs are used to adjust the water levels in both the marsh and the diversion chamber. A pump down arrangement (as well as 36 inch gate in the spillway) facilitates emptying the impoundment. When the pumping system is used, the water is sprayed on to the filter surface and is treated as it would be in a storm event.

loadings and hence calculate mass removal efficiency for any given storm event. Alternatively, we used mesocosms or isolation chambers to experimentally determine the daily nutrient removal per unit area of the marsh, and then predict overall efficiency of the entire marsh. The mesocosm approach allows one to make predictions of how a small, isolated portion of a wetland functions and thus yields information which can be used to design future systems. The mass balance approach, on the other hand, is used after construction and indicates how an existing facility performs. The key, of course, is how well do these two approaches agree. In an extensive comparison study, La Rock and Johengen (1991) determined that mesocosms predicted nitrate and phosphate removals to within plus or minus 5% of the removal values actually determined by mass balance.

For the discussion at hand, mass balances will primarily be used to demonstrate removal effectiveness, but the results of a mesocosm experiment will be presented to illustrate differences in removal efficiency for different nutrient species and the various components of the marsh system, (i.e., plants, sediments and water column).

Mass Balance - Water samples were collected at inflow and outflow stations using automated samplers (ISCO Model 1680). Samples were taken every 15 minutes during peak storm flow, with the sampling interval gradually being increased to one, two, four, six and twelve-hour intervals as the flow decreased and eventually ceased. Flow calculations were performed by the U.S. Geological Survey (USGS) using weir equations and changes in pool elevation recorded at five minute intervals. Loading values were calculated by multiplying the unit concentrations found in each sample by the total water volume that flowed through the marsh within that sampling interval, and then adding all the individual loading values over the storm event. Discrete sampling on short-term intervals and good flow data assured an accurate assessment of mass removal.

Mesocosm Experiments - A series of isolation chambers were used to quantify the cycling of nitrogen and phosphorus among rooted plants, water column algae, and sediment components of the marsh. The chambers were 1 meter tall, constructed from styrene. and enclosed a sediment area of 0.203 square meters. Chambers were typically placed in waters depths of 50 centimeters and entrained water volumes of approximately 100 liters. Experimental chambers with open bottoms were placed over either macrophytes or areas of sediment devoid of plants, to evaluate removals by these components. Chambers with sealed bottoms were used to assess removals by the water-column organisms alone. These chambers were filled through holes in their sides and when equilibrated, the holes were resealed.

Nutrient removal experiments were performed by raising the concentrations of NO_3^-, NH_4^+, and PO_4^{-3} within the chambers to approximately 0.5 mg/l. An enrichment concentration of 0.5 mg/l was used in all experiments as this was the maximum concentration of these nutrients found in a storm event. Before each sample was collected, the volume of water enclosed in each isolation chamber was measured for use in mass-flux calculations, and to correct for any changes arising from transpiration, leakage, or rain. Water samples were collected from both inside and outside the chambers on an hourly basis. Chambers were gently stirred by hand prior to sampling, with care being taken not to resuspend sedimentary material. A 500 ml sample was withdrawn each time and stored in the dark at 4 C until analyzed. Nutrient removal rates were calculated as both a flux rate (mg m^{-2} day^{-1}), and a percent change in concentration per day.

Analytical Techniques - Samples were filtered through a 0.45 micron TCM Gelman 450 uM filter prior to analysis for dissolved nutrients. Automated colorimetric procedures were performed on a Technicon Autoanalyzer according to Standard Methods (APHA 1985) for nitrate (cadmium reduction method), ammonium (indophenol method), ortho-phosphate (phosphomolybdenum blue method), and chloride (ferric thiocyanate methods). Total phosphate (standard methods) and total nitrogen (Solarzano and Sharp 1980) were determined after

digestion by persulfate as these techniques afforded rapid analysis of large numbers of samples. Dissolved oxygen was measured using a YSI Model 54 oxygen probe, calibrated against the azide modification of the Winkler method. Chlorophyll a was determined by chloroform-methanol extraction and read on a Turner 111 fluorometer.

Results and Discussion

Artificial Marsh - Much consideration was given to the plant types that were to be introduced in each cell of the marsh. Evaluation criteria included growth habit, vigor, cold and disease tolerance, nutrient uptake rates, drought and inundation tolerance, biomass production and accumulation, winter die-back, propensity for domination, propagation techniques (seeds, cutting, soil mulch, etc.), and availability of propagation material. The use of several plant types in a compartmentalized system afforded the chance to evaluate each species as well as increasing the functional success for which the plants were chosen. Additionally, using compartments allowed for independent inundation or desiccation of each as desired. In the first cell of the marsh, stands of *Cladium jamaicense* (sawgrass) were planted because of its long-term, seasonal growth rate. Plants were obtained from donor sites in adjacent Wakulla County, FL., but the first planting had a low survival rate and 60 percent of the plants were replaced. Plants other than the sawgrass also sprouted during the summer of 1983, because their seeds were apparently carried by soil attached to the roots of the sawgrass.

The second compartment is 1.8 acres in size and operated at water depths of 1.0 to 2.5 feet. This cell was planted with *Scirpus validus* (bulrush) as it will grow dense enough to retard and distribute flow of water through the system. Bulrush are considered to be good filtering agents that will remove both dissolved nutrients and bacteria. Some 40 percent of the plants died and were replaced with another type of bulrush, *Scirpus californicus*. The first plants were obtained from an aquatic nursery in south Florida, and although there was no problem with undesirable weeds, difficulty arose with a moth larvae infestation that killed the plants below water level.

The third compartment is 2.7 acres in size, has an overall depth of 1 to 2.5 feet except at the outflow where there is a deep basin designed for the deposition of organic sediment. The shallow portion (1.63 acres) was inoculated with topsoil from a farm pond that contained a high content of pickerelweed (*Pontederia lanceolate*), *Sagittaria* species and *Bacopa caroliniana*. The pickerelweed grew densely and has the effect of slowing waterflow which in turn promotes siltation and nutrient uptake. This plant does die back in the winter and will release some nutrients to the effluent.

Mass Balance Determinations - Findings from mass balance calculations made between 1985-1987, revealed that the marsh system was eventually effective in removing solids and nutrients, and that efficiency improved temporally since the planting in 1984 (Table 1). Mass balance determinations from five stormwater events in 1985 (the year after the marsh was replanted) suggested that the marsh was not particularly effective in removing either inorganic or organic solids during the first 12-16 months after its construction (Table 1), with the exception of the June 11 storm event. Calculations for three stormwater events in 1986 and 1987, however, suggested that as the marsh became established it also became efficient in solids and nutrient extraction. Of particular interest is the apparent seasonality noted in 1985, where the most efficient removals are noted in June, but reasonable removals are found between February and October. Calcium, magnesium, and ammonia tended to be exported more frequently than most other chemical species, which we attribute to dissolution of the dolomite filter bed (Ca and Mg) and extensive accumulations of ammonia in the impoundment that were flushed with each storm event.

Studies were also undertaken to assess the dynamics of NO_3^-, NH_4^+, and PO_4^{-3} uptake by marsh plant communities using mesocosms, and the results of

Table 1. Percent changes in mass loading in an artificial marsh.*

Loading Parameter	Storm Date 07Jan85	12Feb85	11Jun85	30Oct85	26Nov85	25Oct86	01Dec86	30Mar87
Inorg. Sol. (%)	+209.3	+30.2	-76.4	-59.4	+164.5	-99.91	-77.57	-76.48
Org. Sol. (%)	+2831.2	+55.7	-69.1	-41.0	+287.9	-97.45	-40.13	-53.82
Calcium (%)	+1038.9	+1.7	-5.3	+11.6	+197.0	-89.01	+16.61	-19.38
Magnesium (%)	+806.1	-3.6	-22.1	+8.0	+1063.4	-88.26	+10.24	-0.35
Chloride (%)	+1981.2	+11.4	-23.3	+20.1	+303.1	-89.42	+7.01	-14.64
Total-N (%)	------	+23.5	-36.4	+16.6	------	-88.68	-4.96	-50.43
Ammonia (%)	+1600.0	+1.0	-2.4	+79.7	+905.9	-95.25	+24.47	-37.61
Nitrate (%)	+669.4	-83.8	-63.4	+133.5	-96.1	-98.24	-30.38	-51.39
Nitrite (%)	------	------	-34.5	+77.6	------	-98.00	-26.61	-26.65
Total-P (%)	------	+5.8	-60.2	-13.4	------	-98.39	-25.16	-40.01
Unfilt. Phos. (%)	+666.6	-12.2	-42.8	+18.4	+724.8	-99.72	-4.18	-58.61
Filt. Phos. (%)	+920.0	-9.1	-51.4	------	------	------	+9.4	------
Silica (%)	+783.1	-24.0	-34.8	+7.0	------	-97.78	-10.81	-39.01
Sulfate (%)	+2312.7	+48.3	+0.6	------	------	-92.77	+86.04	-22.09

* Data represent mass retained (-sign) or exported (+sign) between the point of entry and exit of the marsh.

one such experiment are seen in Fig. 3. The results demonstrate the relative effectiveness of the complete plant system over either the sediments and water column alone. Examination of the curves in Fig. 3 reveals that NH_4^+ is eliminated faster than NO_3^-, and that there is much less difference between the sediment and plant systems for PO_4^{-3}, reflecting the sorption characteristics of the sediments for this compound. Of particular importance is the finding that plant systems will out compete planktonic algae (the water component) in nutrient removal, and thus plant systems may represent a significant nutrient sink in lakes and wetlands. As far as kinetics are concerned, approximately 30% of the added PO_4^{-3} is removed in the first eight hours of the experiment. The mean residence time for the ten storms we measured was 18 hours, and thus we could expect overall removals of approximately 60%, a figure that we frequently measured. Work reported elsewhere (LaRock and Johengen, 1991) indicated removals would continue until levels of about 30 ug/l were reached for the PO_4^{-3} and NH_4^+ after 24 to 48 hours, but that NO_3^- would be completely removed.

The differences between NH_4^+ and NO_3^- removals are probably the result of nitrifying bacteria, which grew abundantly in the filter bed and which we isolated from the stormwater leaving the impoundment and entering the marsh. Recall that these bacteria use NH_4^- as an energy source, oxidizing it to NO_3^- in the process. Thus the apparent enhanced NH_4^+ removal may in fact not be uptake of the material, but rather its conversion to NO_3^- resulting in a depressed removal rate for that compound.

One major source of NH_4^- that we did identify was the anoxic bottom water of the receiving impoundment surrounding the filter. Massive phytoplankton blooms developed that served as a source of organic material that continuously accumulated on the bottom sediments. The system was designed so that inflowing stormwater first filled the impoundment raising the water level to the elevation of the filter surface. The water then flowed through the filter and into the marsh, and ultimately into the receiving lake water. Between storms the impoundment never drained and we measured an oxygen gradient that ranged from supersaturation at the surface to zero at the bottom. If one measured the concentration of some given chemical parameter of incoming water during a storm

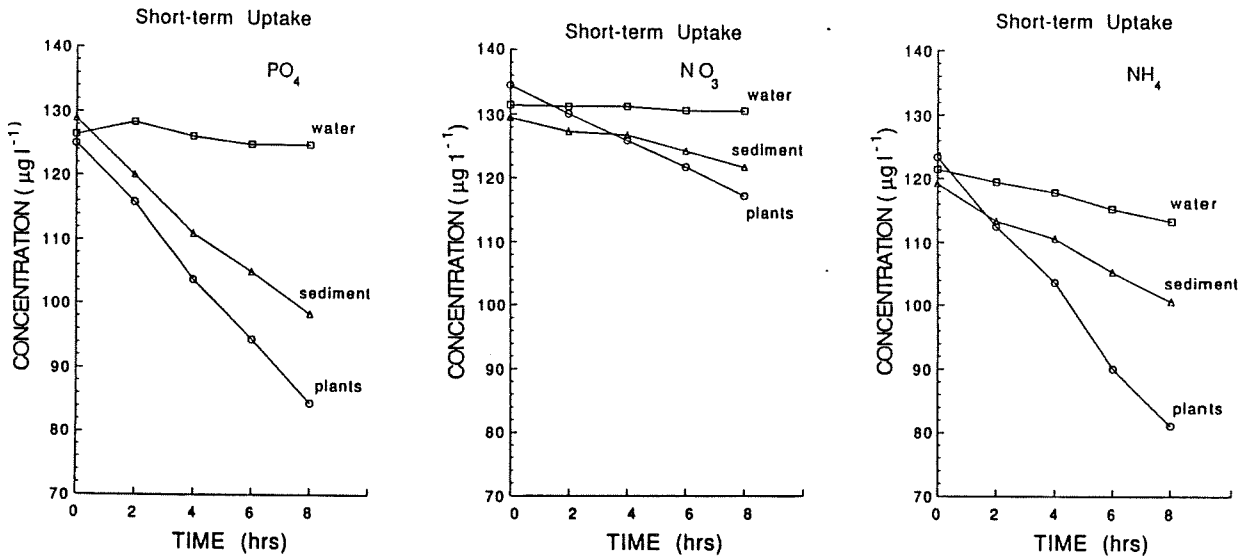

Figure 3. Nutrient removal dynamics in a short-term mesocosm experiment in the artifical marsh. Differences in removal rate of the various nutrient species is discussed in the text.

event, the usual response would be to observe a maximum concentration within the first hours, followed by a continual decline as the storm progressed to its end. This was not the case with NH_4^+ in the filter effluent (marsh influent), where we observed a continual increase in NH_4^+ through the storm event (Fig. 4). The entering stormwater served to flush the NH_4^+ rich bottom water in the impoundment into the marsh, a consideration that should be investigated when using any retention or detention impoundment for storm water control.

A final point is that there is temporal shift in removal efficiency during the year. LaRock and Johengen (1991) found that NO_3^- was removed more effectively than NH_4^+ at the beginning of the year but the situation was reversed by years end. This change could reflect the accumulation and decomposition of plant and algal debris giving rise to NH_4^+ and eventually the development of a large nitrifying bacterial community.

Effectiveness of the Marsh

Overall, the data indicate that removals by the artificial marsh for organic and inorganic solids were 60 and 76% respectively, in 1987 after the marsh had matured (Table 1), while total nitrogen and total phosphorus removals were 50 and 40%, respectively. This research demonstrates that low-energy marsh systems can be a viable tool in renovating stormwater, especially in areas where reasonable land and labor costs allow its large-scale application, and where land use controls can be applied to help lessen the impacts of urbanization. It is doubtful that significant reversal of the degradation process will occur in the short term, but it is hopeful that the nutrient-sink characteristics of the aquatic macrophyte will offset the detrimental effects of continued (though reduced) nutrient influx in the long run.

The early imposition of land use controls and proper zoning, combined with practical low-intensity designs for artificial marsh systems will prevent the degradation of valuable habitats. This

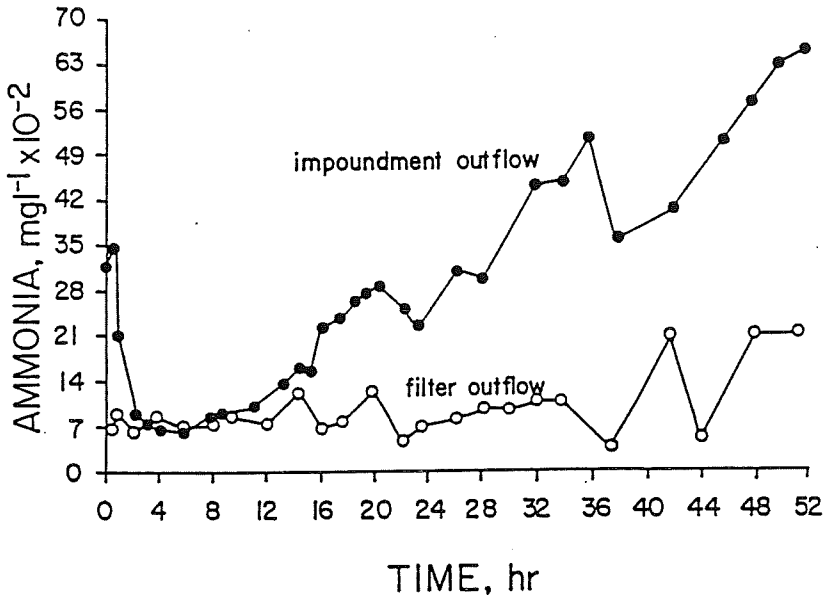

Figure 4. Ammonia release from the impoundment pond feeding the artificial marsh. Inflowing stormwater flushes the basin and in so doing discharges ammonia that had accumulated at the bottom of the pond between storm events. Note that the ammonia levels increased consistently through the storm, and attained levels that were two to three times greater than the peak nitrogen concentrations measured in the stormwater inflow itself. The filter outflow (open circles) is the water that actually passed through the sand filter in contrast to the impoundment outflow (solid circles) which is that water that bypasses the filter during extreme storm events. In essence the pond serves to concentrate nitrogen, and unless frequently emptied can serve as a major nutrient source, an implication that should be considered in the design and management of retention and detention systems.

lesson should be applied in the planning stages as large agricultural tracts are converted to urban use. Stronger restrictions relating to water quality need to be implemented if aquatic resources are to be maintained.

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The Impact of Herbicides on Bulrush Communities - An Evaluation of Water Hyacinth Control Efforts Within Native Plants

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Introduction

This study was undertaken to address concerns of the effects of water hyacinth control performed within valued native plant communities. Bulrushes are highly valued in Florida waters. However, when bulrush communities are weed-infested, weed control must proceed within their boundaries.

The southern or giant bulrush, *Scirpus californicus* (C. Meyer) Steud., is a member of the Cyperaceae or sedge family and is one of the 13 species of Scirpus listed by Godfrey and Wooten (1979) as occurring in Florida. It is known to range throughout the Gulf state to southern California, into South America and also on Easter Island and Hawaii (Small 1972, Godfrey and wooten 1979, Heiser 1979). The genus comprises about 150 species worldwide and is cosmopolitan in distribution with the best representation in North America (Beetle 1950, Riemer 1984).

This species is noted for its long, straight, bladeless stems which may grow to nine feet in length, and in water up to six feet deep (Fassett 1957, Wunderlin 1982, Dresser et al. 1987). The stems may be one inch in diameter at the base and gradually tapering to a sharp point. The cross section of the stem may be cylindrical at the base and more triangular towards its apex. The inflorescence is at the tip of the stem with chestnut brown spikelet clusters. The fruit is a grayish brown nutlet surrounded by feathery bristles. The plant reproduces by seed or vegetatively from rhizomes (Small 1972, Godfrey and Wooten 1979, Dressler et al. 1987).

Environmental requirements for species in the genus Scirpus vary greatly in relation to sediment type, salinity, water depth and water quality (Laing 1941, Rossi and Nuncia 1976, Beal 1977, Stanley and Hoffman 1977, Barko and Smart 1978, Heiser 1979, Barclay and Crawford 1982 and 1983). Giant bulrush is found in shallow ponds, lake margins, canals and ditches, stream or river banks and fresh to brackish marshes. The literature on the ecology of giant bulrush, *S. californicus* is limited. A study by Rossi and Nuncia (1976) in lakes and rivers of Argentina indicated that in winter, giant bulrush put more growth into rhizome elongation rather than in new longitudinal stem growth. They found that rhizomes could withstand long periods of moisture deprivation and, under adequate humidity, rapidly recolonized. They also found that development of new rhizome was strongly correlated with soil aeration. Anaerobic rhizome development was only 4 percent of rhizome growth under aerobic conditions.

A species often found growing in association with *S. californicus* and commonly misidentified as such, is *S. validus*. *Scirpus validus*, or softstem bulrush, prefers sandy to sandy-loam soils and attempts to transplant *S. validus* indicates that it does not survive in silty or silty-loam soils with organic contents greater than 35 percent (Stanley and Hoffman 1977). Barko and Smart (1978) reported that the growth of *S. validus* on silty clay

soils was nearly ten times greater than growth on pure sand soils during a two month period under laboratory conditions. It was reported by Laing (1941) that shoots of *S.validus* died under soil conditions with less than one percent oxygen after a period of only eight days and the buds on the rhizome of *S validus* remained dormant when placed in oxygen free conditions. Bud development sustained active growth when placed in moist, aerobic conditions and obtained maximum growth at 10 percent oxygen, by volume. The shoots produced from these buds were killed when returned to an oxygen-free environment. Another species, *S. maritimus* remained healthy and produced normal shoot development and growth from unextended buds for up to eight weeks of anaerobic conditions (Barclay and Crawford 1982 and 1983).

Scirpus californicus and other bulrush species are widely acclaimed for their value as fishery and wildlife habitat (Beetle 1950, Davis and vander Valk 1978, Heiser 1979, Langeland 1981, Denson and Langford 1982, Johnson and Montalbano 1984). Parts of plants of the genus Scirpus have been found in the stomachs of 31 species of American ducks with most frequent consumers of bulrush seeds being mallard, pintail, green-winged and blue-winged teals, and lesser scaup (Beetle 1950). The open growth form of bulrush communities creates an "edge effect", providing increased substrate for the attachment of food organisms and also serves as important spawning areas for centrarchids. Bulrush stands also function as natural fish attractors in Lake Tohopekaliga as evidenced by high rates of fish capture there (Denson and Langford 1982). In Central and South America *S. californicus*, or *totora*, is used by the Indians for making boats called *maraias*, mats or *esteras* used as beds, rugs, windbreaks, fences, storage bins, roofs and even entire houses are also made from the reeds. The rhizome of *totora* is utilized to make bread and entire plants are commonly consumed by livestock (Heiser 1979). *Scirpus* sp. are also an excellent candidate for marsh reestablishment because of the easy transplanting and rapid expansion rates.

In spite of its many beneficial attributes, Scirpus is also considered to be a "weed" species in certain water bodies. Hold et al. (1977) even list Scirpus as one of the worlds worst weeds. Scirpus species are vigorously stoloniferous and are well adapted to habitat created by flooding of irrigation ditches (Beetle 1950, Wunderlin 1982). It quickly colonizes and clogs canals, slowing stream flow and necessitating clearing operations. Flow has been reduced to one-third of normal rates in severe cases. These plants have been known to interfere with rice production and disrupt harvesting in fish hatcheries and fish rearing ponds (Surber 1947, Mashhor 1987).

Where control operations are justified, the primary methods appear to be mechanical clearing, ditch reconstruction and / or herbicidal control. Control with 2,4-D at a rate of 1 gal/ac (assumed 4 lb a.i./gal) in 50 gallons of water plus six ounces of "wetter" has been reported (Suber 1947, Lowman 1965, Lopinot 1976); however, it was noted that plants treated the previous fall resprouted the following spring indicating that complete control required more than one application. Other herbicides listed as effective include: granular 2,4-D ester at 100 lbs/ac, dichlobenil granules at 100 lbs/ac and a solution of three quarts diquat and one quart Banvel D in 50 gallons of water.

MATERIAL AND METHODS

The evaluations were conducted on East Lake Tohopekaliga, a 12,500 acre lake in northern Osceola County, Florida. The work was coordinated with the Florida Department of Natural Resources, the University of Floridas Center for Aquatic Plants, the Florida Game and Freshwater Fish Commission, the South Florida Water Management District and the U.S. Army Corps of Engineers, Jacksonville District. The site was selected because of the extensive stands of bulrush located throughout the littoral zone, approximately 225 acres (Schardt 1986), and the historically problematic water hyacinth management programs associated with these dense stands of vegetation. Treat-

ment plots were all 3,000 square feet (30 x 100) divided into three subplots with three replications per subplot. Plots were located on the west shore of the lake in approximately three to four feet of water. The average density of live bulrush stems in all the plots prior to treatment was 44 stems per square meter. Table 1 lists the herbicide treatments applied in 1985 and Table 2 lists treatments applied in 1986. All herbicide applications were tank mixed in an equivalent of 100 gal/acre water. To assess treatment injury, permanent sample stations (nine per plot) were established using one quarter meter square (0.25 m2) floating PVC frames.

For the 1985 study, live stem counts were performed biweekly throughout the project in each of the 21 plots. Because the determination for living versus dead stems was a subjective measure for early herbicide symptoms, stem counts for successive sampling dates were averaged into sampling periods in order to reduce variability associated with counting error. Data from the first sample period (14 and 21 days post treatment) and the last sample period (16 and 20 weeks post treatment) are presented.

For the 1986 study, live stem counts were performed at periodic intervals following each of the three application dates.

For both the 1985 and 1986 study, a percentage change in live stem density from initial stem density was calculated for all frames in each of the plots and for each of the sample dates. Means for each

Table 1. Herbicide treatments applied to bulrush in 1985 at East lake Tohopekaliga, Osceola County, Flordia.

Treatment #	Herbicide	Rate (lbs/acre)	Adjuvant	Rate
1	2,4-D	2.0	-	-
2	2,4-D	1.0	-	-
3	2,4-D	1.0	CIDE-KICK[1]	0.25%v/v
4	2,4-D	2.0	CIDE-KICK	0.25%v/v
5	2,4-D	2.0	573Polymer[2]	0.06%v/v
6	2,4-D	1.0	573 Polymer	0.06%v/v
7	diquat	1.0	-	-
8	diquat	2.0	-	-
9	diquat	1.0	CIDE-KICK	0.25%v/v
10	diquat	1.0	573 Polymer	0.06%v/v
11	diquat	2.0	CIDE-KICK	0.25%v/v
12	diquat	2.0	573 Polymer	0.06%v/v
13	2,4-D + diquat	2.0 + 2.0	CIDE-KICK +	0.25%v/v + 1-30
14			I'VOD	
15	-	-	CIDE-KICK	0.25%v/v
16	-	-	573 Polymer	0.06%v/v
17	-	-	I'VOD	1-30
18	diquat	1.0	I'VOD	1-30
19	diquat	2.0	I'VOD	1-30
20	-	-	KAMMO[2]	0.25%v/v
21	-	-	AGRI-DEX[2]	0.25%v/v
	Control (no treatment)			

[1]CIDE-KICK and I'VOD are manufactured by JLB International Chemical Co.
[2] 573 Polymer, KAMMO and AGRI-DEX are manufactured by Helena Chemical Co.

Table 2. Treatments applied to bulrush in 1986 at East Lake Tohopekaliga, Osceola County, Florida.

Treatment #	Treatment	Number of Applications Per Treatment
1	2,4-D @ 3.0 lbs/acre	ONE
2	Control 1	
3	2,4-D @ 3.0 lbs/acre	TWO
4	Mechanically Cut	THREE
5	2,4-D @ 3.0 lbs/acre	THREE
6	Control 2	
7	diquat @ 1.25 lbs/acre	TWO
8	diquat @ 1.25 lbs/acre	THREE
9	Mechanically Cut	TWO
10	Control 3	
11	diquat @ 1.25 lbs/acre	ONE
12	Mechanically Cut	ONE

of the treatments were calculated and comparisons between treatments were made at each sample date.

RESULTS AND DISCUSSION

The 1985 study results presented in Tables 3 & 4, provide the Duncan multiple range groupings for significant differences between means during sampling periods one and four, respectively. Analysis of the data and field observations indicated that the initial impact of the contact herbicide, diquat, had manifested itself by producing brown necrotic stems to the waterline. By 21 days post treatment (sampling period one) all diquat treatments produced at least an 83 percent reduction in live stems when compared to pre-treatment densities. During the same period, the systematic herbicide, 2,4-D treatments, with the exception of plot 13, exhibited a maximum 42 percent reduction. The greatest percent change in bulrush density with 2,4-D was at the 2.0 lbs per acre rate with the addition of the adjuvant CIDE-KICK.

As winter approached, a rapid decline in live stem density for the control plot (number 21) was observed; thus, sampling period four was used to represent the end effects of the treatments conducted in the study. As noted from Table 4, the diquat treatments, with the exception of two treatment, exhibited a tendency towards recovery from the initial effects of the contact herbicide. This was particularly true with the lower rate of diquat. All of the 2,4-D treatments continued to decrease in live stem densities through time. In particular, the presence of adjuvant in 2,4-D treatments tended to enhance the effects of 2,4-D on bulrush. This could be due to interaction with the cuticle and possible increased uptake and/or translocation of 2,4-D. However, no significant difference was noted between the 2,4-D and diquat at the lower rates with no adjuvants. It should be remembered that this data is based on a single mid-summer treatment when the plants had perhaps reached maximum annual biomass and density. Herbicide treatments made early in the growing season and/or multiple treatment, as would be the case in an actual water hyacinth management programs, may produce more definitive and different results. Therefore, the goal of the 1986 study was to address these questions.

It is clear, however, that single treatments with the combination of 2,4-D, diquat, IVOD, and CIDE-KICK

Table 3. Mean percent change in live stem density (sampling period 1) from initial bulrush stem density resulting from herbicide-adjuvant treatments. Means with the same letter are not significantly different (p=0.05).

Treatment #	Herbicide	Rate (lbs/acre)	Adjuvant	Mean % Change in Live Stem Density	Duncan Grouping	
20	-	-	AGRI-DEX	33.3		A
14	-	-	CIDE-KICK	28.0	B	
21	Control	-	-	16.4	B	C
15	-	-	573Polymer	16.0	B	C
19	-	-	KAMMO	11.9	D	C
16	-	-	I'VOD	4.8	D	C
6	2,4-D	1.0	573Polymer	-1.4	D	
1	2,4-D	2.0	-	-17.9		E
2	2,4-D	1.0	-	-24.8		E
3	2,4-D	1.0	CIDE-KICK	-27.8	F	E
5	2,4-D	2.0	573 Polymer	-33.1	F	E
4	2,4-D	2.0	CIDE-KICK	-42.2	F	
9	diquat	1.0	CIDE-KICK	-83.5		G
7	diquat	1.0	-	-84.4		G
10	diquat	1.0	573 Polymer	-85.6		G
18	diquat	2.0	I'VOD	-90.5		
8	diquat	2.0	-	-91.9		G
17	diquat	1.0	I'VOD	-94.0		G
11	diquat	2.0	CIDE-KICK	-94.0		G
12	diquat	2.0	573 Polymer	-94.0		G
13	2,4-D + diquat	2.0 + 2.0	CIDE-KICK + I'VOD	-96.2		G

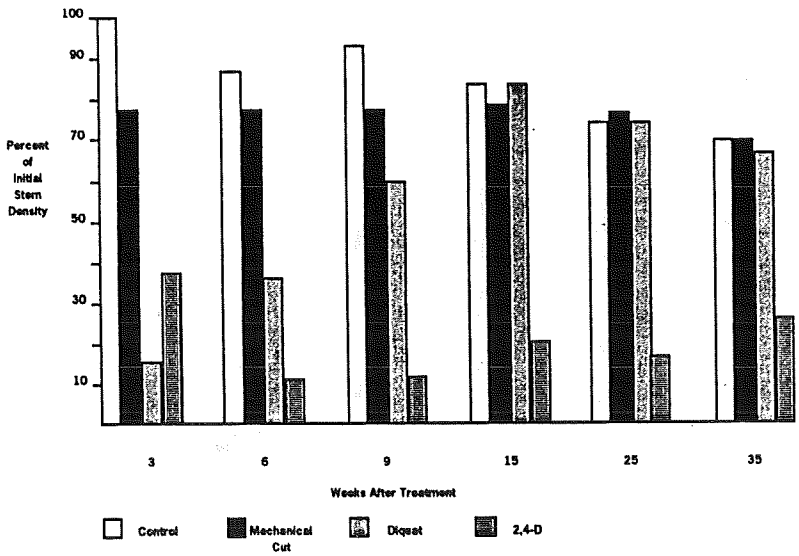
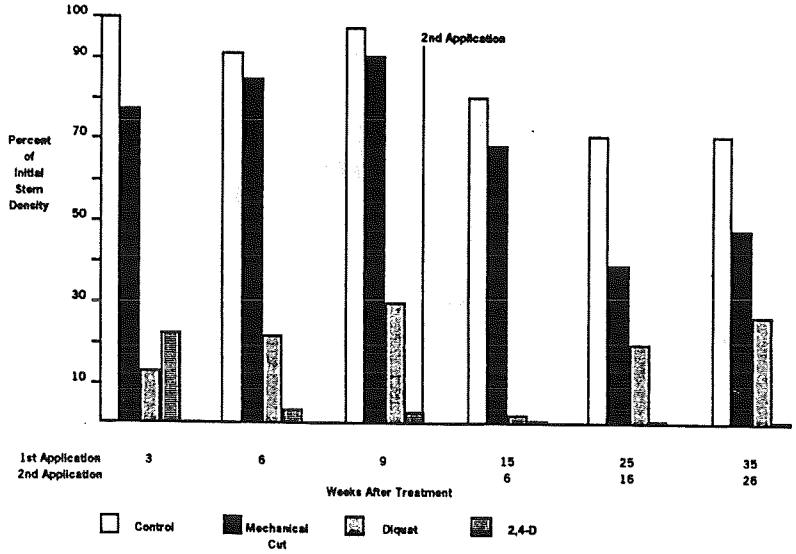


Figure 1. Results of 1986 study, single application to bulrush.

Table 4. Mean percent change in live stem density (sampling period 4) from initial bulrush stem density resulting from herbicide-adjuvant treatments. Means with the same letter are not significantly different (p=0.05).

Treatment #	Herbicide	Rate (lbs/acre)	Adjuvant	Mean % Change in Live Stem Density	Duncan Grouping		
14	-	-	CIDE-KICK	28.8		A	
20	-	-	AGRI-DEX	23.4	B		
19	-	-	KAMMO	2.4	B	C	
21	Control	-		-17.0		C	
15	-	-	573 Polymer	-18.2		C	
16	-	-	I'VOD	-22.2	D		
10	diquat	1.0	573 Polymer	-46.5	D	E	
2	2,4-D	1.0	-	-57.8	F	E	
7	diquat	1.0	-	-58.0	F	E	
18	diquat	2.0	I'VOD	-58.8	F	E	
1	2,4-D	2.0	-	-65.9	F	E	G
17	diquat	1.0	I'VOD	-68.8	F	E	G
3	2,4-D	1.0	CIDE-KICK	-69.9	F	E	G
4	2,4-D	2.0	CIDE-KICK	-70.7	F	E	G
6	2,4-D	1.0	573 Polymer	-74.5	F	H	G
11	diquat	2.0	CIDE-KICK	-76.1	F	H	G
8	diquat	2.0	-	-79.9	F	H	G
5	2,4-D	2.0	573 Polymer	-86.6		H	G
9	diquat	1.0	CIDE-KICK	-97.7		H	
12	diquat	2.0	573 Polymer	-98.5		H	
13	2,4-D + diquat	2.0 + 2.0	CIDE-KICK	-98.6		H	

Figure 2. Results of 1986 study, double application to bulrush.



had severe impact on bulrush when compared to lower rates of the herbicides alone.

It was interesting to note that none of the adjuvants were greatly phytotoxic by themselves when compared over the entire period of the study. However, in the absence of the herbicides, AGRI-DEX and CIDE-KICK significantly increased the density of stems when compared to the control plot (Tables 3 & 4). The mechanism of this effect is unknown.

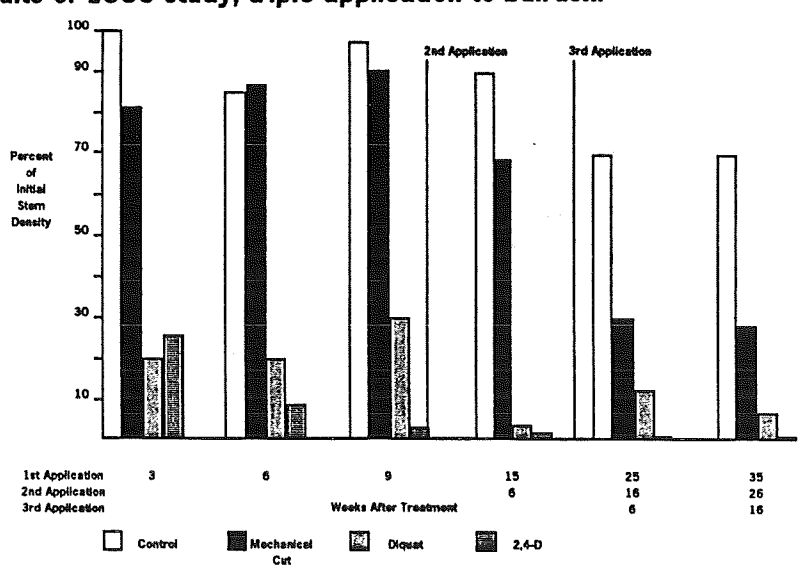
Results of the 1986 study are presented in Figures 1-3. Results from the 1985 study indicated that early growing season treatments with multiple applications throughout the growing season, would be more indicative of an actual water hyacinth management program. Therefore, plots were established which were treated once, twice and three times throughout the growing season using diquat, 2,4-D and mechanical cutting of bulrush stems.

Percent change in live stem density for control (no treatment) plots shows a gradual decrease over time, indicating that bulrush reaches a maximum annual biomass and density early in the spring. Bulrushes in the mechanically cut plots were clipped several inches above the water line to insure that the floating frames remained stationary. Although cut stems continued to grow 35 weeks after clipping, nearly all the living bulrush stem were new sprouts, indicating that the cut stems did not grow and rotted soon after cutting.

The effects of multiple applications of mid-label rates of both diquat and 2,4-D can be seen in Figures 2 and 3. It appears that 2,4-D effectively translocates into rhizomatous tissue preventing resprouting from treated stem. Multiple applications effectively controlled bulrush and no new growth was observed in the plots throughout the study period. Diquat killed emergent stems, but appeared to have little to no effect on the ability of the bulrush rhizome to send up new sprouts. The effect of diquat on bulrush stems was similar to the effect of mechanical cutting when percents of live stem density of new sprouts are compared.

Based on the results of the 1985 and 1986 studies, it can be concluded that the use of the lower rates of 2,4-D or diquat with no additional adjuvants appeared to have low adverse impact on bulrush. The combination of 2,4-D and diquat is commonly used when water hyacinth and water lettuce are found growing together. However, it should be avoided in bulrush communities. Throughout the summer months, bulrush treated with 2,4-D gradually decreased in overall stem density, whereas diquat treated bulrush stems died rapidly, but tended to recover over the growing season. Field and lab studies (data not presented here) indicate that the uncontrolled growth of water hyacinth has an adverse impact upon the bulrush community

Figure 3. Results of 1986 study, triple application to bulrush.



which is equal to the effects of some of the more severe herbicide-adjuvant combinations. This water hyacinth effect could be due to a combination of environmental factors. The mechanism is believed to include effects of the deposition of organic matter by water hyacinth. This has been estimated at up to 5.2 tons per acre per year (Joyce 1985). Negative effects also include shading of the hydrosoil and wing blown water hyacinth mats which physically tear bulrush stems from the rhizomes. When water hyacinth are allowed to grow uncontrolled, the combined effect of the water hyacinth and their subsequent herbicidal control can locally eliminate bulrush communities. Multiple treatments (1 every 3 months) using the mid-label rate of 2,4-D effectively eradicated the bulrush in the treatment plots after two applications. Diquat used at mid-label rates killed emergent vegetation, but appeared to have little to no effect on the ability of the rhizome to send up new sprouts, even after three applications. When comparing the new sprouts of mechanically cut bulrush to the regrowth of diquat treated bulrush, stem densities appeared to be very similar.

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