REPLACEMENT OF GIZZARD SHAD (*Dorosoma cepedianum*) BY BLUE TILAPIA (*Tilapia aurea*) AS A POTENTIAL BIOMANIPULATION AGENT IN FLORIDA EUTROPHIC LAKES.

By

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REPLACEMENT OF GIZZARD SHAD (Dorosoma cepedianum) BY BLUE TILAPIA (Tilapia aurea) AS A POTENTIAL BIOMANIPULATION AGENT IN FLORIDA EUTROPHIC LAKES.

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The effects of replacement of gizzard shad (Dorosoma cepedianum) as a dominant filter feeding fish by blue tilapia (Tilapia aurea) in six eutrophic Florida lakes were analyzed using the lakes' historical limnological characteristics and sedimentary records. The digestive physiology of gizzard shad and blue tilapia (with emphasis on fecal material) was studied in order to evaluate the relative impact in aquatic systems of the feces produced by different fish species.

Five of the six lakes studied are classified as hypereutrophic systems. Lake Gibson is classified as eutrophic. All of the study sites are urban lakes subject to anthropogenic pressures in the form of industrial, residential and commercial development in the watershed,

those of the lakes studied, biomanipulation techniques can mitigate cultural eutrophication to improve water clarity, if gizzard shad are replaced by blue tilapia.

some agricultural activities, and recreational uses such as boating, fishing, and skiing.

Cluster analysis performed grouping all six lakes revealed that, with the exception of Lake Parker, all lakes could be aggregated by common limnological features, and that the relationships were independent of year, season and the combination of year and season.

Mean primary productivity for the studied lakes was 334.22 $mgC.m^{-3}.h$ ± 106.35. Primary productivity in Lake Parker (mean= 66.41 $mgC.m^{-3}.h$) was five times less than the mean for the other five lakes.

Sedimentary records reveal that the lakes in this study have been eutrophic for at least the last century with a significant accumulation of organic material in the superficial mud. Superficial sediment content was always ≥40% of the organic matter for any lake. Lake Parker was the only lake investigated that displayed a progressive decline in water column total phosphorus in the last fifteen years, perhaps signaling some reversal of cultural enrichment.

There was a significant difference in the amount of feces hand-stripped from the fishes studied. Blue tilapia had a greater amount of fecal material than did gizzard shad. The bulk of the fecal material collected from both fish species consisted of small green and blue-green algae, which were the dominant algal taxa in the environment at the time the fishes were collected.

There were also significant differences in the composition of the fecal material produced by each fish species. Gizzard shad fecal material consisted of 37% more organic matter and 50% more protein than did blue tilapia fecal material. Caloric content, measured as gross heat, in gizzard shad feces was 60% greater than in blue tilapia fecal material.

Initial chlorophyll *a* values from the feces measured for blue tilapia were 10 to 15 times greater than those measured for gizzard shad feces. Two bioassays were conducted over a seven day period for each procedure. Final chlorophyll *a* values for gizzard shad feces presented a ten-fold increase from the initial chlorophyll *a* concentration, regardless of the presence of blue-green or green algae as the inoculum. Blue tilapia, on the other hand, exhibited a three- to fivefold increase in samples where a green algal species was used as the inoculum (there was zero growth in 66% of the samples); and there was no apparent growth in samples inoculated with a blue-green algae.

The results indicate that blue tilapia fecal material completely suppressed blue-green algal chlorophyll *a* production and appears to have suppressed green algal chlorophyll *a* production in more than 60% of the samples. Gizzard shad feces increased chlorophyll *a* values in blue-green and green algal groups (tenfold increase). This study suggests that in Florida systems with conditions similar to

CHAPTER 1
INTRODUCTION

Management of planktonic food webs to improve lake trophic state is still an experimental procedure, and many interactions are unknown or poorly understood (Crowder et al. 1988). There is now enough evidence that top-down interactions (fish introduction or removal) have a significant effect on planktonic communities (algal biomass reduction), particularly in low nutrient lakes (Cooke et al. 1993).

Chemical treatment, unless it involves an inactivation of nutrient release (i.e., from the bottom), has proved unsatisfactory for algal control. Mechanical treatments such as algal harvesting, artificial circulation, and bottom sealing have been shown not to be satisfactory due to either their ephemeral effects or their high cost. Biological control, even in its earliest stage, seems to be a promising approach for long term algal control.

Traditionally, limnologists have considered the interactions in a lake ecosystem as an unidirectional flow of components consisting of a nutrient-phytoplankton-zooplankton-fish pathway. Manipulation of food webs is an area fertile with possibilities for understanding ecosystem

function and for developing techniques for lake management that are not dependent on chemical and mechanical means.

All research done in the field of biomanipulation supports the conclusion that the presence of large-bodied zooplankton, Daphnia spp., is required for a strong control over phytoplankton populations. Unfortunately, not all geographic areas meet this requirement. Tropical and subtropical areas, namely Florida and other areas of the southeastern U.S., do not have large species of zooplankton. Crisman and Beaver (1990) described only the presence of small Cladocera (e.g., Eubosmina spp., Ceriodaphnia spp.) in Florida lakes which are not good candidates for top-down biomanipulation purposes.

Lacking the presence of the most studied and accepted candidate for the top-down type of biomanipulation, tropical and subtropical researchers have had to seek another organism which could perform a corresponding role. Filter-feeding fish, which are very common in those areas, seem to be the most suitable candidates. Filter-feeders do not visually detect individual prey items, but engulf a volume of water containing the food organisms and retain the planktonic prey and particles by passing this volume over entrapment structures (Lazzaro, 1987).

The most common filter-feeding fishes in Florida eutrophic lakes are gizzard shad (Dorosoma cepedianum), which can produce a total grazing pressure on algal

populations even greater than the large bodied zooplankton in temperate systems (cf. Drenner et al., 1982a). Another very common filter-feeder fish inhabiting Florida eutrophic lakes is blue tilapia (Tilapia aurea) which was introduced into the United States in 1961. Both of these fishes feed on small, particulate material. For example, gizzard shad 5 cm in length can filter algae >19μ in size; those 15 cm in length can filter algae >40μ; and those 25 cm in length can filter algae >63μ. Blue tilapia can consume algae >25μ in size (Opuszinsky and Shireman, unpubl.).

The filtering rates of these species can be very high. According to Drenner et al. (1982a), the gizzard shad population in Lake Barkley, Texas (85 ha), can effectively filter a volume equivalent to the entire lake every 2.3 days.

Undoubtedly, filter-feeding fishes can alter the phytoplankton community of aquatic systems. Nevertheless, contrary to expectations, an increase in planktonic algae biomass and primary production has been observed (Drenner et al., 1984, 1986; Janusko, 1974,1978; Opuszynski, 1978). Some of the reasons for the failure of filter-feeding fish to control algal blooms and in some cases even worsen water quality are [1] elimination of larger zooplankton species; [2] more rapid cycling of plant nutrients (Opuszynski and Shireman, unpubl); and [3] differential release of fecal material by individual species of fish.

Digestion in animals has been an almost totally neglected subject. Fish studies have been limited to gut content and selectivity (Halver, 1989). Very little is known of the digestive capabilities of fishes.

Fish, like all other animals, require energy to sustain life, and they are among the most efficient animals in converting food to body tissue (Halver, 1989). However, a rough generalization can be made that about one-third of the energy in the food offered to fish will be lost as combustible waste (Halver, 1989). This will consist of uneaten food, feces, urine, and gill excretions. From the lost combustible wastes, feces (25% of the total) constitute the majority of this part of the energy balance.

Fecal material produced from different fishes and released into the environment can produce a differential impact and consequently a differential response. Blue tilapia feces, being encased in mucilage, are more coherent and will be broken down slowly, whereas shad feces are released practically as aqueous substance permitting an almost instantaneous availability of nutrients for the phytoplankton community.

Because little is known of their feeding ecology, especially quantitative feeding under natural conditions, attempts to predict and efficiently manage these filter-feeding fish as a tool to alleviate eutrophication processes still is in its experimental stage. Some researchers (i.e.

Opuszynsky and Shireman, unpubl.) have developed a new approach to improve the use of filter-feeding fish to counteract eutrophication. They constructed an apparatus consisting of a cage where fish are kept that was equipped with funnels under the cage to collect fish feces for estimating food consumption via a quantification of the production of feces. According to these researchers, this cage combines two indispensable features for an effective use of filter-feeding fish to control planktonic algal growth and to reduce eutrophication: [1] it enables the coexistence of filter-feeding fish and zooplankton by eliminating the fish from the water column thus improving phytoplankton consumption; and [2] fish feces can easily be collected and removed from the bottom of the cage, eliminating a major source of nutrient for the phytoplankton community.

There is a high degree of trophic overlap between young blue tilapia and larval gizzard shad in Lake George, Florida (Zale, 1984). Beaver and Crisman (1989) reported phytoplankton and zooplankton community alterations in eutrophic Florida lakes when blue tilapia has an established population. In eutrophic central Florida lakes, blue tilapia is quickly replacing gizzard shad as a major filter-feeding fish (Beaver and Crisman, 1989).

The current study addresses gaps on current knowledge regarding fish as biomanipulation agents for phytoplankton.

I worked with the digestive physiology of blue tilapia and gizzard shad with an emphasis on their fecal material, as well as with the limnological characteristics and paleolimnological interpretations from six central Florida lakes. The research addressed the following hypotheses: [1] There is differential nutrient bioavailability from blue tilapia and gizzard shad feces; [2] different type of feces will produce a different effect on the lake ecosystem; [3] fundamental differences in the physiology and biochemistry of blue tilapia and gizzard shad feces will impact the sediment composition in lakes; and [4] interlake differences should exist in key limnological parameters between lakes of differing relative abundance of blue tilapia and gizzard shad. The goals of this study are to [1] learn something about the fecal material composition of filter-feeding fish to help understand their role in freshwater systems; [2] evaluate management techniques that couple physicochemical water information with biotic community components and the lake sedimentary record; and [3] gather new information on the effect of different fish species' fecal material on lake ecosystems.

CHAPTER 2
LITERATURE REVIEW

Introduction

This chapter reviews the literature on biomanipulation and shad and tilapia ecological energetics by examining planktivorous fish from the community standpoint with special emphasis on fecal material composition and fate in the system and the significance of different fish species as trophic linkages in freshwater food webs.

Biomanipulation

The two main strategies to control eutrophication of fresh waters are 1) reduction of external and internal loading of nutrients (Bjork, 1985) and 2) control of internal ecological processes. With respect to toxic substances, organic wastes and acid precipitation, the first strategy alone will provide acceptable solutions to the problems on a long-term scale (Benndorf, 1988). However, a combination of strategies 1 and 2 could lead to an improvement in water quality and to a lower cost/benefit ratio in the management of the water resource.

Until recently, eutrophication problems were tackled primarily by reducing external nutrient loading (Hosper and

Meijer, 1986; Van Liere, 1986). In recent years, many studies have shown that food web manipulation (biomanipulation) can help restore eutrophied lakes (Andersson et al., 1978; Henrikson et al., 1980; Reinertsen and Olsen, 1984; Carpenter et al., 1985).

Biomanipulation, as it was originally defined by Shapiro et al., (1975), refers to management of aquatic communities by controlling natural populations of organisms with the end goal being water quality improvement. This definition was reviewed later by Shapiro (1990), and he pointed out the considerably broader connotation that the term has taken on recently relative to that espoused by him and many others in the field. In the original idea of Shapiro (1990), biomanipulation is a series of manipulations of the biota of lakes and their habitats to facilitate certain interactions and results which we, as lake users, consider beneficial, namely reduction of algal biomass and, in particular, of blue-green species.

According to Gophen (1990), in a broad sense, biomanipulation is equatable to top-down forces, trophic cascade interactions or food-web manipulation. All these terms refer to manipulation of secondary or tertiary aquatic producers and its impact on lacustrine community structure. Whole-lake food-web manipulation by fish-stock management, i.e., reduction of planktivorous fish, through enhancement of piscivorous fish, may accelerate the rate of the

restoration process (e.g. Shapiro and Wright, 1984; Edmondson and Abella, 1988).

For studying the effect of food-web manipulations on lake restoration and to examine the fundamental mechanisms underlying ecosystem regulation, application of whole-lake manipulation-experiments can have major advantages (van Donk et al., 1990). These experiments simulate or actually encompass the conditions that would be expected to occur naturally in lakes (Carpenter and Kitchell, 1988). Problems of enclosure-size and omitted members of communities are not relevant to whole-lake manipulations (Frost et al., 1988). It is, however, difficult to perform these manipulations on a large scale and to interpret their results (Hulbert, 1984). Interpretations may be greatly eased by results of small-scale manipulations on similar systems (Frost et al., 1988).

Nevertheless, to obtain a good overview of the community's processes, investigations must be conducted simultaneously as small-scale in situ or laboratory experiments that can be replicated under controlled conditions. In this approach, whole-lake manipulations can be considered as generating as well as testing hypotheses (O'Neill et al., 1986).

Examination of trophic-level interactions has long been an integral part of limnology (Hrbacek et al., 1961; Nauwerck, 1963; Brooks and Dodson, 1965). However, Shapiro

et al. (1975) and Shapiro (1978) recognized that eutrophication problems are biological manifestations of nutrient availability, and were the first to suggest that manipulation of trophic interactions (biomanipulation) could be used as a lake management tool to alleviate the biological consequences of eutrophication without the need for costly controls on nutrient loading (Crisman and Beaver, 1990).

Since biomanipulation research began in temperate regions, the emphasis has focused basically on alterations of the zooplankton community (enhancement of large cladocerans especially of the genus Daphnia), because, through these manipulations, phytoplankton has been reduced (blue-green algae) and water transparency has increased despite difficulties in maintaining such a condition for a long time. Only recently has research been conducted using fishes as a major element in the biomanipulation process, but still only under temperate conditions.

In subtropical and tropical regions, conditions are quite different. One particular element must be stressed; there is a complete absence of large zooplankton in tropical regions. This and numerous other physicochemical and biological differences between these climatic regions impedes the applicability of biomanipulation research from the temperate regions to warmer climates (Crisman and Beaver, 1990). Thus, the natural choice for biomanipulation

schemes of subtropical and tropical regions is the use of planktivorous fish. Despite the potential for utilization of filter-feeding fish in biomanipulation techniques to maintain algal control, especially in shallow eutrophic lakes and reservoirs characterizing subtropical and tropical regions, little information exists regarding the direct impact on the general planktonic community.

Florida lakes possess characteristics which fundamentally differentiate them from most temperate systems. The relative shallowness of Florida lakes, combined with a moderate climatic regime, dictates that stable thermal stratification usually only occurs in deeper (>5-6 m) lakes of the region (Beaver et al., 1981). Such lakes are warm monomictic and circulate throughout the winter months, while shallower basins are more strongly affected by local meteorological events and are subjected to higher internal nutrient loading via sediment resuspension (Pollman, 1982).

Subtropical lakes of Florida are inherently different from temperate lakes in a number of respects that could affect the success of whole lake biomanipulation. The size structure of subtropical zooplankton communities is skewed toward smaller individuals, and large-bodied cladocerans are absent (Bays and Crisman, 1983). Despite the lack of large-bodied crustaceans, intense grazing activities by gizzard shad may strongly determine plankton structural and functional characteristics (Bays and Crisman, 1983). These

subtropical systems maintain primary production all year while production in temperate systems is greatly depressed or absent during winter. Thus, if non-vegetative seasonal estimates of production are included in determining annual production, it is likely that subtropical systems would realize greater yearly production.

From the few papers about the use and application of biomanipulation techniques in the subtropical zone, Crisman and Beaver (1990) noted that the increase in cladoceran abundance in Florida lakes following elimination of fish predation agrees with observations in temperate lakes (Lynch, 1979; Carpenter et al., 1987; Van Donk et al., 1989), but unlike the latter lakes, community species composition was not altered nor was there a marked increase in crustacean mean body size (Shapiro and Wright, 1984; Benndorf et al., 1988). Large-bodied daphnids, the focus of all temperate studies, are absent in Florida regardless of trophic state or predation intensity (Crisman and Beaver, 1990). The results of Crisman and Beaver (1988) from research conducted in Lake Apopka, Florida, show a fundamental disagreement with the biomanipulation of Round Lake, Minnesota (Shapiro and Wright, 1984), and suggest that zooplankton size, structure and standing crop have only minimal influence on phytoplankton biomass in Florida lakes.

Regarding fish composition, Crisman and Beaver (1990) noted that unlike the eutrophic temperate lakes

characterized by size-selective planktivorous fishes, in subtropical systems this composition shifts to pump-filter feeding fishes, reflecting faunal dominance by gizzard shad (Dorosoma cepedianum). Crisman and Beaver (1990) noted that in both temperate and subtropical systems, removal of planktivorous fish results in higher macrozooplankton populations.

Unlike temperate systems, however, algal biomass was not reduced in the presence of enhanced macrozooplankton abundance, but actually increased. Crisman and Beaver (1990) suggested that small-bodied macrozooplankton, even if freed from fish predation, are of questionable value as biomanipulation tools in eutrophic subtropical lakes. These authors also stated that if biomanipulation were to be successful in the subtropics, emphasis should be shifted from zooplankton to the role played by planktivorous fish (pump-filter feeding).

Shad Ecological Energetics

The gizzard shad (Dorosoma cepedianum) occurs commonly in many lakes and streams in North America (Scott and Crossman, 1973), where it is the principal phytophagous fish (Crisman and Kennedy, 1982). This species is predominately found in eutrophic lakes, being reportedly a key link in the food chain between primary producers and the top carnivores

which are important to commercial and sport fishermen (Lewis, 1953; Jester and Jensen, 1972).

Despite its role as a forage fish, it has been suggested that in warm-water, shallow lakes with soft mud bottoms, high turbidity, and relatively few predators, the gizzard shad may become a nuisance from an ecological and economic standpoints (Miller, 1960). Kutkuhn (1958) and Cramer and Marzolf (1970) reported that gizzard shad longer than 35 mm are primarily herbivorous, with some selection for zooplankton when sighted, while young gizzard shad (< 30 mm) feed primarily on zooplankton (Bodola, 1966). Organic detritus at times can also be an important food for the gizzard shad (Dalquest and Peters, 1966; Baker and Schmitz, 1971).

Apparently, owing to the high reproductive capacity and rapid growth of gizzard shad in shallow lakes and reservoirs, predators cannot effectively crop young-of-the-year fish and hence provide a control on population numbers (Miller, 1960; Bodola, 1966). It has also been suggested that gizzard shad may inhibit growth of more desirable fish and/or limit their numbers through interspecific competition (Berry, 1958; Miller, 1960).

The gizzard shad is a dominant native grazer in Florida eutrophic lakes. Crisman and Kennedy (1982), based on mesocosm experiments, showed that gizzard shad did not impact chlorophyll *a* values, productivity or phytoplankton

densities, and that shad caused a significant increase in both the concentration of orthophosphate and its ratio to total phosphorus under natural stocking densities. The fish was also responsible for a significant decrease in copepod density during that study.

Crisman and Beaver (1988) and Threlked and Drenner (1987) noted that the overall effect of shad grazing was to stimulate the phytoplankton community and to decrease Secchi disk transparency. Shad do not effectively graze more evasive zooplankton such as Diaptomus (Drenner et al., 1978), particularly during the summer months, although other evasive herbivores may be simultaneously depressed (Drenner et al., 1982a). Many common algal taxa remain viable after gut passage through the digestive tract of shad, especially blue-greens (Velasquez, 1939; Smith, 1963; Crisman and Kennedy, 1982).

It has been assumed that the enhancement of phytoplankton populations by shad grazing is an indirect effect caused by suppression of herbivorous zooplankton (Threlkeld, 1987). Shad may also enhance phytoplankton populations directly by providing a larger quantity of highly assimilable nutrients from their digestive products (Crisman and Kennedy, 1982).

Gizzard shad and other clupeids are very sensitive fish that can be easily stressed and killed. In systems where gizzard shad is considered a nuisance, as in all Florida

eutrophic lakes, the death of large numbers of shad, besides any aesthetic consequences, will represent an enormous return of phosphorus and nitrogen into the water column that is readily available to stimulate the growth of already overly productive algae, resulting in oxygen depletion that could lead to another fish kill.

The results of the Crisman and Kennedy (1982) investigation suggest that the presence of gizzard shad can promote lake eutrophication both through elevation of orthophosphate concentrations and differential digestion of diatoms and green algae, thus increasing the competitive advantage of blue-green algae. Finally, they emphasize that this fish species does not appear to be a suitable candidate for use as a biocontrol agent for phytoplankton in eutrophic subtropical lakes.

Considerable literature is available concerning the life history of Dorosoma cepedianum (Drenner, 1977; Lazzaro, 1987). Only a few investigators, however, have attempted to describe and quantify energy relationships in this species (Smith, 1971; Garland, 1972; Pierce, 1977; Crisman and Kennedy, 1982).

Tilapia Ecological Energetics

Blue tilapia (Tilapia aurea), a species native to Africa and the Middle East, was introduced into the United States in 1961 in a Hillsborough County phosphate pit (Ware,

1973) as a means of controlling nuisance aquatic macrophytes (Courtenay and Robins, 1973; McBay, 1961). After its original introduction, the fish spread rapidly throughout the southeast, particularly Florida, but have not controlled excessive macrophyte growth (Ware et al., 1975).

Qualitative and quantitative measurements of the gut contents of blue tilapia *in situ* indicate that this species is an opportunistic omnivore, utilizing zooplankton (Spataru and Zorn, 1978; Mallin, 1986), phytoplankton (Hendricks and Noble, 1980; Mallin, 1986), and detritus (Hendricks and Noble, 1980; Mallin, 1986). Gophen et al. (1983) found in laboratory tests that tilapia greater than 7.6 cm utilized a series of rapid suctions to draw prey into their buccal cavity. This mechanism is undirected, and thus, fish in this size range function as filter feeders. Tilapia smaller than 7.6 cm also function as filter-feeders; however, they also feed as size selective predators on individual zooplankton specimens.

Drenner et al. (1984a) reported that grazing activities by blue tilapia depressed some large algae (*Uroglenopsis* and *Ceratium*), while the smallest phytoplankton taxa were enhanced (*Rhodomonas*, *Chrysochromulina*, *Chlamydomonas* and *Cyclotella*). This enhancement was ascribed both to nutrient regeneration during gut passage and to fish feces, as well as the accompanying compositional shifts in the herbivorous zooplankton community. Little information is available on

nutrient release and degree of algal digestion by blue tilapia (Crisman and Beaver, 1988), but Popma (1982) noted that some algal cells may remain viable following passage through the digestive tract of tilapia.

The zooplankton community is also modified by tilapia grazing activity, and Drenner et al.(1984a) showed in pond studies that the population of Keratella was suppressed, while copepodids and adult Diaptomus were enhanced. Gophen et al. (1983) reported that blue tilapia are selective feeders on Bosmina and Ceriodaphnia, taxa that have poor evasive capabilities when compared with more successful zooplankters such as Mesocyclops (Crisman and Beaver, 1988). Dickman and Nanne (1987) noted that in Central American fish ponds very high levels of tilapia (2.5 adults.m^{-2}) suppressed zooplankton populations and increased the importance of the bluegreen alga Microcystis aeruginosa.

The spawning behavior of Tilapia aurea is typical of many cichlids with all incubating duties being performed by the female (McBay, 1961). Reproductive behavior progresses from schooling, territorial establishment by the males, prespawning courtship, spawning, and parental care. McBay (1961) suggested that the mature female will spawn at a constant water temperature of 23°C. The number of hatchlings per spawn of T.aurea was comparatively smaller than in most fishes native to the United States, but apparently compensated for by strong parental care behavior.

Foote (1977) considered temperature, predation and salinity to be the primary limiting factors in the distribution of blue tilapia. The sensitivity of blue tilapia to low temperature is apparently the most important factor affecting the potential range of the species in North America. Shafland (1978) tested lower lethal acclimation temperatures for blue tilapia, spotted tilapia and the Mozambique mouth brooder, and found that all tilapia tested died between 6 and 12°C. Of these three tilapia species, blue tilapia was the most tolerant of cold water. According to McBay (1961), T. aurea will not tolerate temperatures as low as 9°C. Based on these results and January isotherms, Shafland concluded that the blue tilapia has the potential of extending its range to include the entire state of Florida (Williams et al. 1985).

Salinity has a slight but significant effect on the cold tolerance of blue tilapia, suggesting that the species may be expected to extend its range farthest north along the coast and that populations in estuarine systems may be able to withstand exceptionally cold weather better than inland populations (Zale, 1984). However, the fish is capable of finding thermal refugia during cold weather; the presence and location of these must be considered when assessing habitat suitability based on thermal criteria (Zale, 1984).

There is concern that the presence of blue tilapia reduces largemouth bass populations through competition for

nesting sites or predation on bass eggs (Noble et al., 1975). Zale (1984) also found a high degree of trophic overlap between young tilapia and larval shad in Lake George, and perhaps the enhanced abundances of individuals documented following introductions of blue tilapia have resulted from exploitative competition for zooplankton during early life stages. He suggested this to be a more realistic explanation than the "competition for algae and detritus among adults" theory usually invoked.

These results are confirmed by Beaver and Crisman (1990) who reported phytoplankton and zooplankton community alterations in subtropical Florida lakes when blue tilapia replaces gizzard shad. Chlorophytes and rotifers were proportionally more abundant in tilapia-dominated lakes (Crisman and Beaver, 1988), while cladocerans composed a greater percentage of the zooplankton population in shad-dominated lakes. Limited zooplankton distribution data suggest that total zooplankton biomass is comparably depressed in systems dominated by tilapia. Empirical evidence suggests that little measurable improvement in water quality through differential grazing will be realized if blue tilapia displace shad as the dominant rough fish in subtropical Florida lakes (Beaver and Crisman, 1990).

A summary from previous investigations in Florida lakes is compiled in Table 2-1. The data reported are condensed from many different sources which were already cited.

Table 2-1. Summary of previous investigations with blue tilapia and gizzard shad in Florida lakes.

BLUE TILAPIA	GIZZARD SHAD
↑ Chlorophytes	↑ Cyanophytes
? Chrysophytes	↑ Chrysophytes
↑ Rotifers	? Rotifers
↓ Cladocerans	↓ Cladocerans
↓ Zooplankton biomass	↑ Primary Productivity

Previous studies (Courtenay and Robins, 1973; Ware et al., 1975; Crisman and Kennedy, 1982; Bays and Crisman, 1983; Zale, 1984; Crisman et al., 1986; Crisman and Beaver, 1988) delineated the differential impact that each fish has on distinct groups of organisms in Florida lakes. It is not clear if any change can be deduced for chrysophytes in systems dominated by blue tilapia, whereas some increase in chlorophytes is reported. Gizzard shad increased cyanophytes and chrysophytes. Regarding zooplankton composition, blue tilapia is reported to increase rotifers while suppressing cladocerans. The gizzard shad impact on rotifers has not been determined, although it is known that cladoceran populations decrease. Finally, both species seem to suppress total zooplankton biomass while gizzard shad was shown to stimulate phytoplankton primary productivity.

CHAPTER 3
RELATIVE IMPORTANCE OF BLUE TILAPIA AND GIZZARD SHAD TO LAKE SEDIMENT AND WATER COLUMN NUTRIENT CONCENTRATION-- ANALYSIS OF FISH FECES

Introduction

Examination of trophic-level interactions has long been an integral part of limnology (Caird, 1945; Hrbácek et al., 1961; Nauwerck, 1963; Brooks and Dodson, 1965). Elements such as herbivory, predation, and nutrient recycling by animals have always been considered to affect the biomass and species composition of prey populations in terrestrial as well as aquatic communities. Several investigators (e.g., Hairston et al., 1960; Wiegert and Owen, 1971; Patten, 1973; Porter, 1977; and Paine, 1980) have examined several aspects of food webs and the factors which control the biomass and productivity of various trophic levels. Applicability of these ideas to aid management practices and promote aquatic ecosystem recovery are now under investigation (Cooke et al. 1993).

Shapiro et al. (1975) and Shapiro (1978) recognized that eutrophication problems are biological manifestations of nutrient availability and were the first to suggest that manipulation of trophic interactions (biomanipulation) could be used as a lake management tool to alleviate the

biological consequences of eutrophication without the need for often costly controls of nutrient loading (Crisman and Beaver, 1990). Biomanipulation, as stated by Shapiro and Wright (1984), is based on the prediction that increased piscivore abundance will result in decreased planktivore abundance, increased zooplankton abundance, and increased zooplankton grazing pressure leading to a reduction in phytoplankton abundance and improved water clarity.

This "biological" approach could make it possible to increase herbivore density in aquatic communities, thereby lowering algal biomass to levels less than expected for a given nutrient concentration. In addition, limiting the abundance or even the occurrence of certain fish could curtail the flux of nutrients from epilimnetic or littoral sediment to the pelagic zone. Improvement of the water quality of lakes with algal blooms by implementing a combination of these biological techniques could reduce or eliminate the need for the use of common chemical (i.e. cooper sulfate) and mechanical methods to deal with eutrophication (Cooke et al., 1993).

Caird (1945), with his experiment of adding largemouth bass to a 15-ha Connecticut lake, was one of the first to publish observations about the effect of increased biomass of piscivorous fish on the phytoplankton community. More recently, investigators such as Hrbácek et al. (1961), Brooks and Dodson (1965), and Hulbert et al. (1972) have

demonstrated that planktivorous fish (mainly filter-feeder fish) can severely reduce or even eliminate large-bodied zooplankter Daphnia spp.

Planktivorous fishes use two distinct means to feed on plankton: particulate feeding and filter feeding. Particulate feeders attack single individual planktonic prey items which they visually select from the water column (Werner, 1977; Vinyard, 1980; Lazzaro, 1987). Filter feeders do not visually detect individual prey but swallow a volume of water containing the food organisms and entrap the planktonic forms in structures such as gill rakers and other filtering structures (see Lazzaro, 1987), using rhythmic suctioning actions to capture the prey, while either swimming slowly or remaining quite stationary (Drenner, 1977; Gophen et al., 1983).

Gizzard shad (Dorosoma cepedianum), a filter feeder, is a dominant native grazer in Florida eutrophic lakes. Crisman and Kennedy (1982) used mesocosm experiments to demonstrate that gizzard shad had no impact on chlorophyll *a* values, lake productivity, or phytoplankton density. Gizzard shad was responsible for a significant increase in both the concentration of orthophosphate and its ratio to total phosphorus under natural stocking conditions and for the decrease in copepod density (Crisman and Kennedy, 1982).

Later experiments, however, demonstrated that the overall effect of shad grazing was to stimulate the

phytoplankton community and to decrease Secchi disk transparency (Crisman and Beaver, 1988; Threlked and Drenner, 1987). Shad do not effectively graze on more evasive zooplankton such as Diaptomus (Drenner et al., 1978), and many common algal taxa remain viable after gut passage through the digestive tract of shad, especially blue-green algae (Velasquez, 1939; Smith, 1963; Crisman and Kennedy, 1982).

Blue tilapia, Tilapia (=Sarotherodon =Oreochromis) aurea, a fish native to West Africa and Palestine (Trewavas, 1965), was introduced into the United States in 1957 by researchers at Auburn University who were investigating its potential as a food and sport fish (Swingle, 1960). In 1961, the Florida Game and Freshwater Fish Commission acquired juvenile fish from Auburn University to study the potential use of blue tilapia as a sport fish (Crittenden, 1962) and as an agent for weed control (McBay, 1961; Courtenay and Robins, 1973). After its introduction, the fish successfully invaded natural habitats throughout the Southeast, particularly Florida. Blue tilapia has not, however, been a successful candidate for use as a sport fish or as a biological control for excessive macrophyte growth (Ware et al., 1975).

Tilapia can be classified as an opportunistic omnivore, consuming zooplankton (Spataru and Zorn, 1978; Mallin, 1986), phytoplankton (Hendricks and Noble, 1980; Mallin,

1986), and detritus (Hendricks and Noble, 1980; Mallin, 1986). Gophen et al. (1983) reported that tilapia greater than 7.6 cm act as filter feeders, and tilapia smaller than 7.6 cm, in addition to filter feeding, also feed as size selective predators on individual zooplankton species (particulate feeder).

The grazing activities of blue tilapia depressed some large sized algal groups (Uroglenopsis and Ceratium), while the smallest phytoplankton taxa were enhanced (Rhodomonas, Chrysochromulina, Chlamydomonas, and Cyclotella) (Drenner et al., 1984a). Tilapia grazing did suppress the zooplankter Keratella, while copepodid and adult Diaptomus populations were enhanced (Drenner et al., 1984a). Tilapia also selectively fed on Bosmina and Ceriodaphnia (Gophen et al., 1983), taxa reportedly having poor evasive capabilities.

Little information is available on nutrient release and degree of algal digestion by blue tilapia. Popma (1982) noted that some algal cells may remain viable following passage through the digestive tract of tilapia. Dickman and Nanne (1987), found that high concentrations of adult tilapia (2.5 adults m^{-2}) raised in some Central America ponds suppressed zooplankton populations and increased the blue-green alga Microcystis aeruginosa.

Methods

Fish Collection

Blue tilapia were collected by electroshocking in Lake Alice, Florida. A boom-style electrofishing boat was used (7.5 GPP, with a 7 KW generator, Smith-Root, Inc. Vancouver, Washington). This model was the most efficient electrofishing unit for the type of lake sampled. The collected fish were immediately placed on ice to minimize post-capture digestive processes.

Blue tilapia were measured for total length (mm) and weight (g). All fish collected were separated into two size groups (TL), less than 380 mm and greater than 380 mm, which will subsequently be referred to as size groups T-1 and T-2, respectively. Thirty-six fish were collected. Thirteen of these were in group T-1; twenty-three were in group T-2 (fish length ranged from a minimum of 284 mm to a maximum of 450 mm). Fecal material, here defined as the collection of all digested material that could be identified by its brownish-dark coloration in the fish intestines, was hand-stripped from all the fishes and pooled into two different jars (T-1 and T-2). The sample jars were immediately placed on ice in a cooler for transport to the laboratory. All sample material was kept at 4°C within four hours after collection.

Samples were subdivided into four different groups; groups 1 and 2 pulled from the T-1 jar and groups 3 and 4

from the T-2 jar. Three aliquots for analysis (replicates) were taken from each of the four groups.

Gizzard shad were collected using an experimental monofilament gillnet. All fish sampled were collected as part of a population control project being conducted by the Florida Game and Freshwater Fish Commission, Eustis (FL). The sampling net had 6 panels ranging from 6.3 to 12.7 cm stretch mesh in 1.3 cm increments. Gillnetting was done in Lake Denham, Florida, in increments of two hours.

Gizzard shad were measured for total length (mm) and weight (g). All fish collected were separated into two size groups (TL), <320-mm and >320-mm, which will subsequently be referred to as size groups S-1 and S-2, respectively. A total of 218 fish were collected in five different gillnet settings. Minimum fish length was 203 mm, and the maximum length was 440 mm. Fecal material was hand-stripped in the field from all the captured fishes and poured into two jars (S-1 and S-2). The sample jars were immediately placed on ice in a cooler for transport to the laboratory. All sample material was kept at 4°C within four hours after collection.

Samples were subdivided into four different groups; groups 1 and 2 pulled from the T-1 jar and groups 3 and 4 from the T-2 jar. Three aliquots for analysis (replicates) were taken from each one of the four groups.

Feces Analysis

The digestive tract was removed through an incision made from the anus anterior to the base of the pelvic fin, then dorsally to the posterior side of the pectoral girdle to the dorsal side of the coelom. Fecal material was hand-stripped from the beginning of the intestine to the base of the anus and placed into a 250 ml jar. The feces were then pooled according to size group (T-1, T-2, S-1 and S-2).

Laboratory analyses consisted of determination of percent dry matter, percent organic matter, bulk density, heat content upon combustion, protein analysis, total phosphorus and total nitrogen content and two algal bioassays. Dry weight/water content of feces was measured by weighing samples before and after drying at 105°C for 16 hours; organic matter was measured using a cool muffle furnace before ashing the samples at 600°C for at least 3 hours and then weighing. Nitrogen was measured as total Kjeldahl nitrogen (TKN) using a Technicon-II semi-automated manifold following a modification of the methods described by Bremner and Mulvaney (1982) (selenium was not used as a catalyst). The digestate was also used for total phosphorus (TP) determinations. Liberated orthophosphate was determined with the ascorbic acid method (APHA, 1985).

All samples for calorimetry analyses were freeze dried. Volatilization began at -30°C. Total energy was determined by oxygen bomb calorimetry (Parr Model 1261 Isoperibol

Calorimeter). Pre-weighed (pre-burn weight, approximately one gram of dried material) fecal samples were placed in a microbomb. The bomb was placed in 2 liters of water at 30-31°C. The sample was ignited and burned in the presence of oxygen. After ignition, increases in water temperature were registered to calculate energy within the sample, expressed as gross heat ($cal.g^{-1}$).

An algal bioassay was done using thirty-six 300 ml flasks containing FWH+Si (a standard medium for maintenance culture of freshwater blue-green algae). Eighteen flasks were prepared without KNO_3 (medium -N); and eighteen flasks prepared without K_2HPO_4 (medium -P). These two groups of eighteen flasks were each subdivided into three different subgroups of six; one subgroup containing 3 ml of shad feces; another containing 3 ml of tilapia feces; and the third subgroup without feces, to function as a control. Each group of six flasks was subdivided further into two subgroups: three flasks inoculated with a strain of blue-green algae (Microcystis aeruginosa), and three flasks inoculated with a strain of green algae (Selenastrum capricornutum). These sets of flasks were incubated for seven days in a room under constant light and temperature conditions.

Samples from all experimental flasks were analyzed for chlorophyll *a* using extractive and fluorimetric methods (APHA, 1989). Cell counting of algae from samples collected

at the beginning and end of the experiment from each flask was performed, using a 2 ml chamber on an inverted Leitz microscope.

Results and Discussion

Bulk density calculated for tilapia feces had a mean value of 0.2576 $g.cm^{-3}$, with a minimum value of 0.2326 $g.cm^{-3}$, and maximum value of 0.2815 $g.cm^{-3}$ (Table 3-1). For shad feces, the mean value was 0.1401 $g.cm^{-3}$, with a minimum value of 0.1258 $g.cm^{-3}$ and maximum value of 0.1493 $g.cm^{-3}$ (Table 3-1).

Percent dry matter for tilapia feces had a mean value of 13.0 %, with a minimum value of 12.6 %, and a maximum value of 13.6 % (Table 3-1). For shad, the mean value was 8.0%, with a minimum value of 7.5 % and a maximum value of 8.6 % (Table 3-1). Percent organic matter calculated from tilapia feces had a mean value of 49.6%, with a minimum value of 48.6 %, and maximum of 51.6 % (Table 3-1). Shad had a mean value of 68.0 %, with a minimum of 67.0 %, and a maximum of 68.9 % (Table 3-1).

Percent protein was calculated for tilapia and had a mean value of 21.6 %, minimum of 21.0 %, and maximum of 23.2 % (Table 3-1). For shad, the mean value was 32.4 %, with a minimum of 30.25 %, and maximum of 33.75 % (Table 3-1).

Table 3-1. Physico and chemical characteristics of shad and tilapia excretions expressed as ml and g.

PARAMETER	SHAD	TILAPIA	SHAD	TILAPIA	SHAD	TILAPIA
	Mean (ml) (g)	Mean (ml) (g)	St. Deviat. (ml) (g)	St. Deviat. (ml) (g)	Range(*) (ml) (g)	Range(*) (ml) (g)
Bulk Density ($g.cm^{-3}$)	1.0/ 0.1401	1.0/ 0.2576	0.72/ 0.010	0.08/ 0.022	(0.90-1.06) (0.12-0.15)	(0.9-1.09) (0.23-0.28)
% Dry Matter	N/A /8.0366	N/A /12.988	N/A /0.5748	N/A /0.4856	(7.49-8.64)	(12.6-13.66)
% Org. Matter	N/A / 67.84	N/A / 49.66	N/A /0.7954	N/A /1.3929	(67.06-68.90)	(48.6-51.63)
% Protein	N/A /32.427	N/A /21.655	N/A /1.3467	N/A /1.0568	(30.25-33.75)	(21.1-23.19)
Calories ($cal.g^{-1}$)	7516.35 ml 4041.05 g	4531.18 ml 2604.13 g	253.49 136.28	278.53 160.08	(7147-7852) (3842-4222)	(3805-5091) (2187-2926)
TN ($\mu g.l^{-1}$)	N/A / 52.98	N/A / 34.65	N/A /0.9307	N/A /1.7055	(48.38-54.00)	(33.4-37.14)
TP ($\mu g.l^{-1}$)	N/A / 6.51	N/A / 6.63	N/A /0.3636	N/A /0.4708	(5.98 - 7.04)	(6.05- 7.13)

(*) Numbers expressed in the 1st line are in ml.
Numbers expressed in the 2nd line are in g.
When only one line presented, it is expressed in g.

Caloric content, expressed as gross heat, for tilapia feces had a mean value of 2604.13 $cal.g^{-1}$, with a minimum value of 2186.88 $cal.g^{-1}$, and maximum of 2926.08 $cal.g^{-1}$ (Table 3-1). Shad had a mean value of 4041.05 $cal.g^{-1}$, and minimum and maximum values of 3842.40 $cal.g^{-1}$ and 4212.81 $cal.g^{-1}$, respectively (Table 3-1). Pierce (1977), from data obtained from 27 samples of shad fecal material, calculated its mean caloric content as 1953 $cal.g^{-1}$.

Total nitrogen for tilapia feces displayed a mean value of 34.6 $\mu g.l^{-1}$, with a minimum of 33.42 $\mu g.l^{-1}$, and a maximum of 37.14 $\mu g.l^{-1}$ (Table 3-1). For shad, the mean was 52.98 $\mu g.l^{-1}$, with the minimum and maximum values being 48.38 $\mu g.l^{-1}$, and 54.0 $\mu g.l^{-1}$, respectively (Table 3-1). Mean total phosphorus (TP) for tilapia was 6.63 $\mu g.l^{-1}$, with a minimum of 6.05 $\mu g.l^{-1}$, and a maximum of 7.13 $\mu g.l^{-1}$ (Table 3-1). Shad had a mean of 6:51 $\mu g.l^{-1}$, with a minimum of 5.98 $\mu g.l^{-1}$, and a maximum of 7.04 $\mu g.l^{-1}$ (Table 3-1).

Absolute values for total nitrogen, percent of nitrogen, crude protein, total phosphorus and percent phosphorus composition for gizzard shad and blue tilapia fecal material are provided in Tables 3-2 and 3-3.

There were significant differences in the amount of feces hand-stripped from both fishes. Tilapia had the greatest amount of feces, and it was not possible to isolate any particular factor responsible for such an occurrence. It may be related to the length of the intestinal tract (which

Table 3-2. Total nitrogen, total phosphorus and protein content for gizzard shad feces.

		µg N	% N	Crude Protein	µg P	% P
Group 1	A	48.38	4.84	30.25	5.98	.60
	B	48.95	4.89	30.56	6.84	.68
Group 2	A	54.00	5.40	33.75	6.41	.64
	B	53.65	5.36	33.50	6.19	.62
Group 3	A	51.43	5.14	32.12	7.04	.70
	B	53.36	5.34	33.37	6.79	.68
Group 4	A	52.44	5.24	32.75	6.55	.65
	B	53.04	5.30	33.12	6.26	.63

Table 3-3. Total nitrogen, total phosphorus and protein content for blue tilapia feces.

		µg N	%N	Crude Protein	µg P	%P
Group 1	A	33.68	3.37	21.06	7.13	.71
	B	33.42	3.34	20.87	6.86	.69
Group 2	A	37.14	3.71	23.19	6.47	.65
	B	34.38	3.44	21.50	6.05	.60
Group 3	A	22.61	2.26	14.12	5.50	.55
	B	22.44	2.24	14.00	5.30	.53
Group 4	A	27.40	2.74	17.12	12.66	1.27
	B	28.48	2.85	17.81	11.54	1.15

in tilapia is longer) or the time of the year (i.e., spawning time). In gizzard shad, the length of the intestinal tract increases with increasing standard length. Schmitz and Baker (1969) found that the intestine-to-body-length ratio for gizzard shad was about 2.8:1, while in threadfin shad the ratio averaged 1.8:1. Although the length of the intestine in a fish may be partially determined by its diet, Schmitz and Baker (1969) believe that the length and form of the intestine are probably the result of a complex of intrinsic and extrinsic factors that can not be explained only by diet.

Tilapia produces five times more feces than bighead carp, whereas the growth rate of both species was comparable (Opuszynski and Shireman, unpubl.). In this study, I found that the average feces produced by the shad was 2.08 ± 0.34g fresh feces per 100g fish body weight and 4.5 ± 0.21g fresh feces per 100g fish body weight for the tilapia. The fact that blue tilapia feces' bulk density is twice as high as shad feces (Table 3-1), indicates that tilapia produces four times more feces as dry matter than shad. Therefore, blue tilapia seems to be more suitable than bighead carp (Opuszynski and Shireman, unpubl.) and shad for use in biological schemes for water quality improvement.

There are obvious macroscopic differences in the feces produced by these two fish taxa. Tilapia feces are released

as mucilaginous pellets, while shad feces are released into the environment as a flocculent material.

The bulk of both fish feces consisted of small green and blue-green algae species, dominating the phytoplankton in the assemblage at the time the fishes were collected. Many of the algal cells did not show any signs of digestion, or they were slightly digested but easily identifiable under a microscope. It is known that many algal taxa may be viable after passage through fish digestive tracts (Fish, 1951, 1955; Lowe, 1959; Smith, 1963). Fish (1951, 1955) reported that, of the algae eaten by tilapia, blue-green and green algae generally pass through the fish undigested and that diatoms form the main usable food (Cailteux, 1988). In this study, I found green algae of the genus Closterium, the blue-green Microcystis, and the diatoms Melosira, Fraqilaria, Tabellaria, and Cyclotella in tilapia feces. Shad feces contained the green algal genus Ankistrodesmus, the diatom Melosira, the dinoflagellate Peridinium, and the blue-green Microcystis.

Mean values for chlorophyll *a* (final - initial measurements) in shad feces were [1] for the inoculum made in the medium deficient of nitrogen(-N) and the blue-green algae 541.57 $mg.m^{-3}$ ± 36.85; and 859.18 $mg.m^{-3}$ ± 17.93 with green algae; [2] for the inoculum made in the medium deficient of phosphorus (-P) and the blue-green algae 432.21 $mg.m^{-3}$ ± 22.02; and 1173.76 $mg.m^{-3}$ ± 13.96 with green algae.

Mean values in tilapia feces were [1] for the inoculum made in the medium deficient of nitrogen (-N) and the blue-green algae -364.05 $mg.m^{-3}$ ± 38.06; and 329.33 $mg.m^{-3}$ ± 31.38 with green algae; [2] for the inoculum made in the medium deficient of phosphorus (-P) and the blue-green algae 683.14 $mg.m^{-3}$ ± 38.62; and -335.58 $mg.m^{-3}$ ± 17.98 for the inoculum with green algae.

The experiment was harvested on the seventh day following daily evaluation of fluorescence. As a major trend, it can be said that fluorescence values increased dramatically (in the shad samples) during the first four days, after which they leveled off and did not change very much. In the medium containing blue-green algae, the combination shad feces -N as well as shad feces -P displayed a steady increase in values, which became more noticeable after the fourth day (Figure 3-1)). For tilapia feces, the medium deficient of nitrogen (-N) had the same value from the beginning and declined after the fourth day; whereas the medium deficient of phosphorus (-P), remained steady and began to increase slightly after the fourth day (Figure 3-1).

In the medium containing green algae, the mixture of shad feces -N increased progressively daily and reached its highest values after the fourth day, which was a ten-fold increase. The combination -P displayed the same trend, however, with higher values increasing thirteen times after

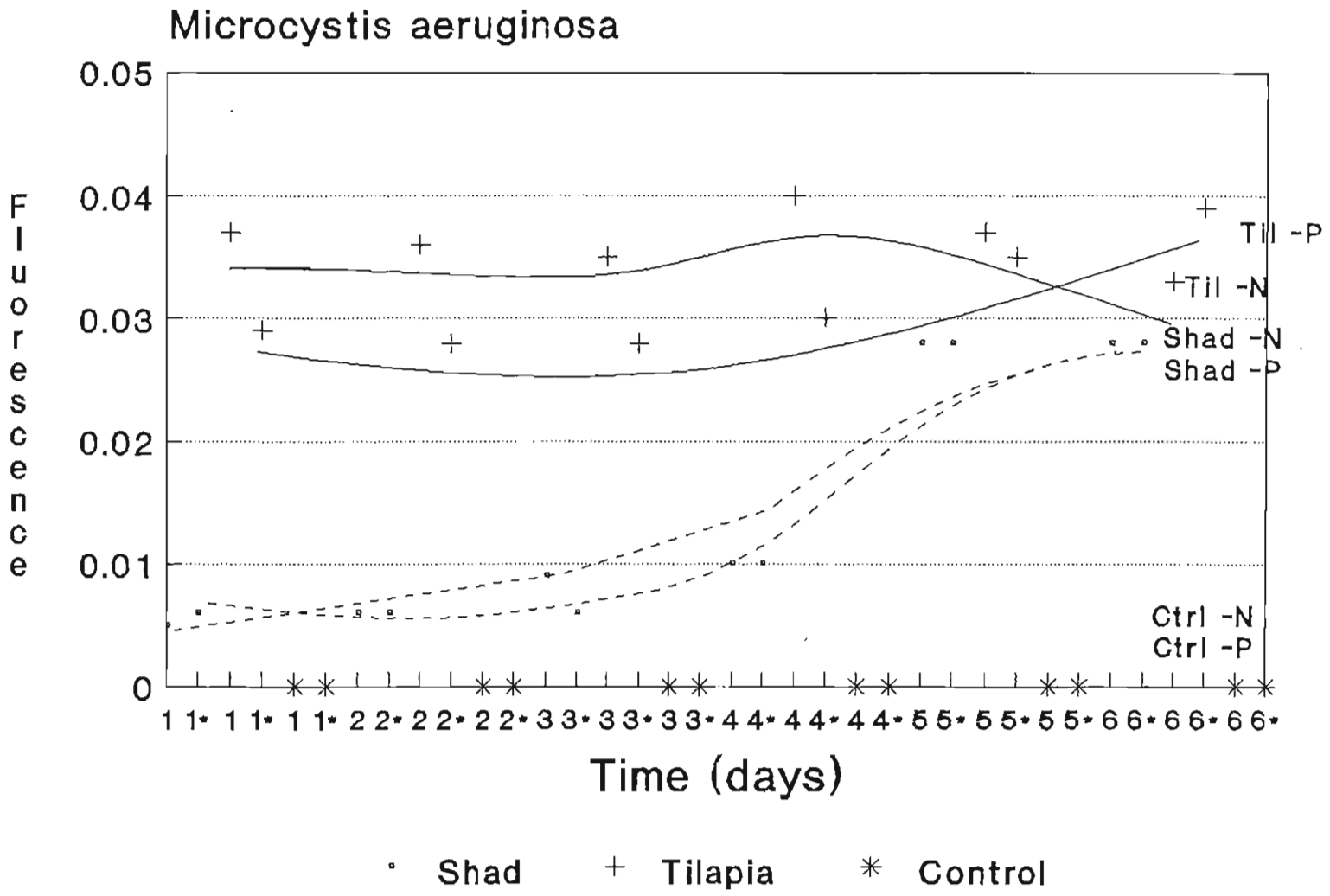

Figure 3-1. Fluorimetric activity from shad and tilapia feces in medium with Microcystis aeruginosa.

the fourth day (Figure 3-2). For tilapia feces, the medium deficient of nitrogen (-N), as well as the medium deficient of phosphorus (-P), showed a daily decrease in fluorescent values and on the fourth day of the experiment they stabilized in a value 50% lower than the initial number. (Figure 3-2). Shad feces displayed a five fold increase in fluorometric values compared with the measured in the medium with tilapia feces.

Phytoplankton biomass in the fecal material was calculated from chlorophyll *a* concentrations assuming 1 mg chlorophyll *a* = 67 mg dry weight (APHA, 1982). Blue tilapia fecal material suppressed phytoplankton biomass (mg.dry weight) in 75% of the samples of blue green and green algal prepared in -N and -P media. Mean suppression in these samples was approximately 50% of the initial values (Table 3-4). Gizzard shad fecal material, on the other hand, increased phytoplankton biomass in all of the samples of blue green and green algae. The mean concentration increase was greater than tenfold of the initial values (Table 3-4).

Conclusions

Adult fishes that employ filter-feeding as their primary feeding method are typically visual particulate-feeding planktivores when juveniles. They switch from a diet

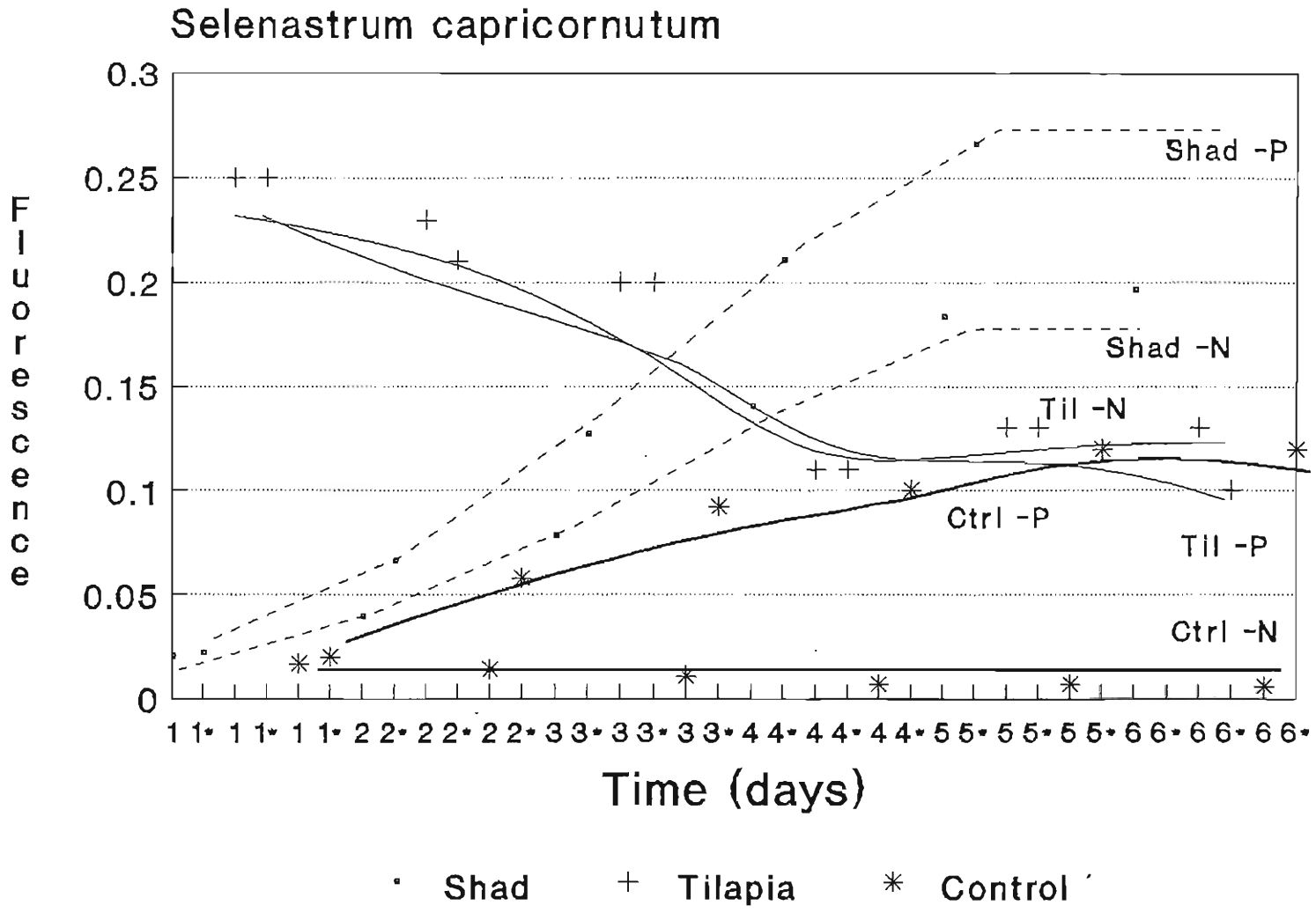

Figure 3-2. Fluorimetric activity from shad and tilapia feces in medium with Selenastrum capricornutum.

Table 3-4. Phytoplankton biomass (mg.dry weight^{-1}) in fish excretions calculated from chlorophyll *a* values.

FISH + ALGA MEDIUM	INITIAL VALUE mg/dry weight	FINAL VALUE mg/dry weight	FIN-INIT (X) mg/dry weight	(X)-CTRL. mg/dry weight
Til (-N)+ Bluegreen	68054	43663	-24391	-23939
Til (-N)+ Green	62232	84297	22065	23269
Til (-P)+ Bluegreen	79421	33651	-45770	-51642
Til (-P)+ Green	68756	46273	-22483	-43562
Shad (-N)+ Bluegreen	2710	38995	36285	36737
Shad (-N)+ Green	4818	62383	57565	58769
Shad (-P)+ Bluegreen	3061	32019	28958	23086
Shad (-P)+ Green	2810	81454	78644	57565
Ctrl (-N)+ Bluegreen	903	451	-452	
Ctrl (-N)+ Green	1882	678	-1204	
Ctrl (-P)+ Bluegreen	502	6374	5872	
Ctrl (-P)+ Green	2308	23387	21079	

composed of large zooplankton as juveniles to increasing dependence on phytoplankton and/or smaller and non-evasive zooplankton as adults (Kutkuhn, 1957; Cramer and Marzolf, 1970; Janssen, 1978; Durbin, 1979; Drenner et al., 1982b; Lazzaro, 1987; and Yowell and Vinyard, 1993). Although the feeding habits of filter-feeding fish have been extensively studied, information about the quantitative feeding under natural conditions is scarce.

The theoretical basis of fish bioenergetics has received considerable attention in the fisheries literature. Sophistication in describing basic equations and the addition of computer analysis have enabled investigators to partition more accurately the energy requirements of a population. The goal of this study was to provide additional information on the feces produced by gizzard shad and blue tilapia including fecal composition, and nutrient and caloric content. Experiments were conducted to compare the potential impact of the fecal material of each respective fish species on the primary production in aquatic systems.

Feces composed the greatest constituent of the energy balance estimated from an experiment with bighead carp employing energy budget equations to calculate food consumption and food assimilation, i.e., C=P+R+F+U (1) and A=C-F+U (2), where C=consumption, A=assimilation, P=production, R=respiration, F=feces, and U=urea (Opuszynski and Shireman, unpubl.).

I found that bulk density for tilapia fecal material was twice the value for shad feces. Averaged shad feces analysis had a composition of 37% more organic matter, 50% more protein and 50% less dry matter than tilapia feces which makes shad feces more organic, more protein-rich and more easily dissolved in the environment. Caloric content of shad feces was 60% higher than tilapia feces.

In general, assimilation efficiencies, which are calculated using the balance of total calories of ingestion and egestion material in fish, have been assumed to approximate 80 percent of ingested energy (Winberg, 1956; Ricker, 1968; Mann, 1969; Moriarty and Moriarty, 1973). Although little is known about the energetics of omnivorous fish such as shad and tilapia, an assimilation efficiency of 42 percent is reported for shad by Pierce (1977), and of 70-80 percent for tilapia (Moriarty and Moriarty, 1973).

In this study, I found caloric content in shad fecal material to have a mean of 4041.05 $cal.g^{-1}$ (Table 3-5), whereas tilapia had a mean of 2604.13 $cal.g^{-1}$ (Table 3-6). Considering both fishes relative to a natural diet, tilapia is egesting material that has 60% less caloric content than shad. This is, consequently, in accordance with what is stated in the literature about the assimilation efficiency for both, where tilapia has roughly two times greater assimilation efficiency than shad, being expected to egest material with a lower caloric content.

Table 3-5. Caloric content of gizzard shad feces.

SHAD	CALORIC CONTENT (Cal/g)
Group 1 (A)	4115.56
(B)	4134.18
(C)	4150.35
Group 2 (A)	4066.21
(B)	4221.81
(C)	4196.37
Group 3 (A)	4026.14
(B)	3935.90
Group 4 (A)	3894.76
(B)	3867.86
(C)	3842.40

Table 3-6. Caloric content of blue tilapia feces.

TILAPIA	CALORIC CONTENT (Cal/g)
Group 1 (A)	2532.20
(B)	2490.92
Group 2 (A)	2722.98
(B)	2648.89
(C)	2926.08
Group 3 (A)	2533.76
(B)	2560.45
(C)	2417.74
Group 4 (A)	2312.26
(B)	2186.88

The diet of the gizzard shad is one of the most controversial aspects in the ecology of this species. It is generally agreed that very young shad (<35 mm) feed almost exclusively on microcrustaceans (Kutkuhn, 1958; Cramer and Marzolf, 1970), and fish larger than 35 mm have been reported to consume a wide variety of food materials. Tilapia are primarily vegetative feeders, feeding on algae or aquatic plants, though occasionally zooplankters and insect larvae picked up with bottom debris are eaten (Lowe, 1959). However, despite the similarity in feeding habits, differing fecal composition and assimilation rates suggest that tilapia and shad play significantly different roles in the nutrient dynamics of aquatic systems.

Chlorophyll *a* values for both shad and tilapia fecal material were very different between the beginning and end of the bioassays. In both bioassays for blue-green and green algal groups, tilapia initial values were 10 to 15 times higher than for shad (tilapia mean value of 915 $mg.m^{-3}$ and shad mean value of 67 $mg.m^{-3}$); whereas at the end of bioassays, the situation was completely inverted, with shad displaying a tenfold increase in chlorophyll *a* (shad mean value of 685 $mg.m^{-3}$) regardless of the algal group tested. (Figures 3-3 to 3-6) Tilapia chlorophyll *a* values at the end of bioassay varied with the algal group tested. In the Microcystis experiment, all values were negative indicating suppressed chlorophyll *a* values (mean of -691 $mg.m^{-3}$)

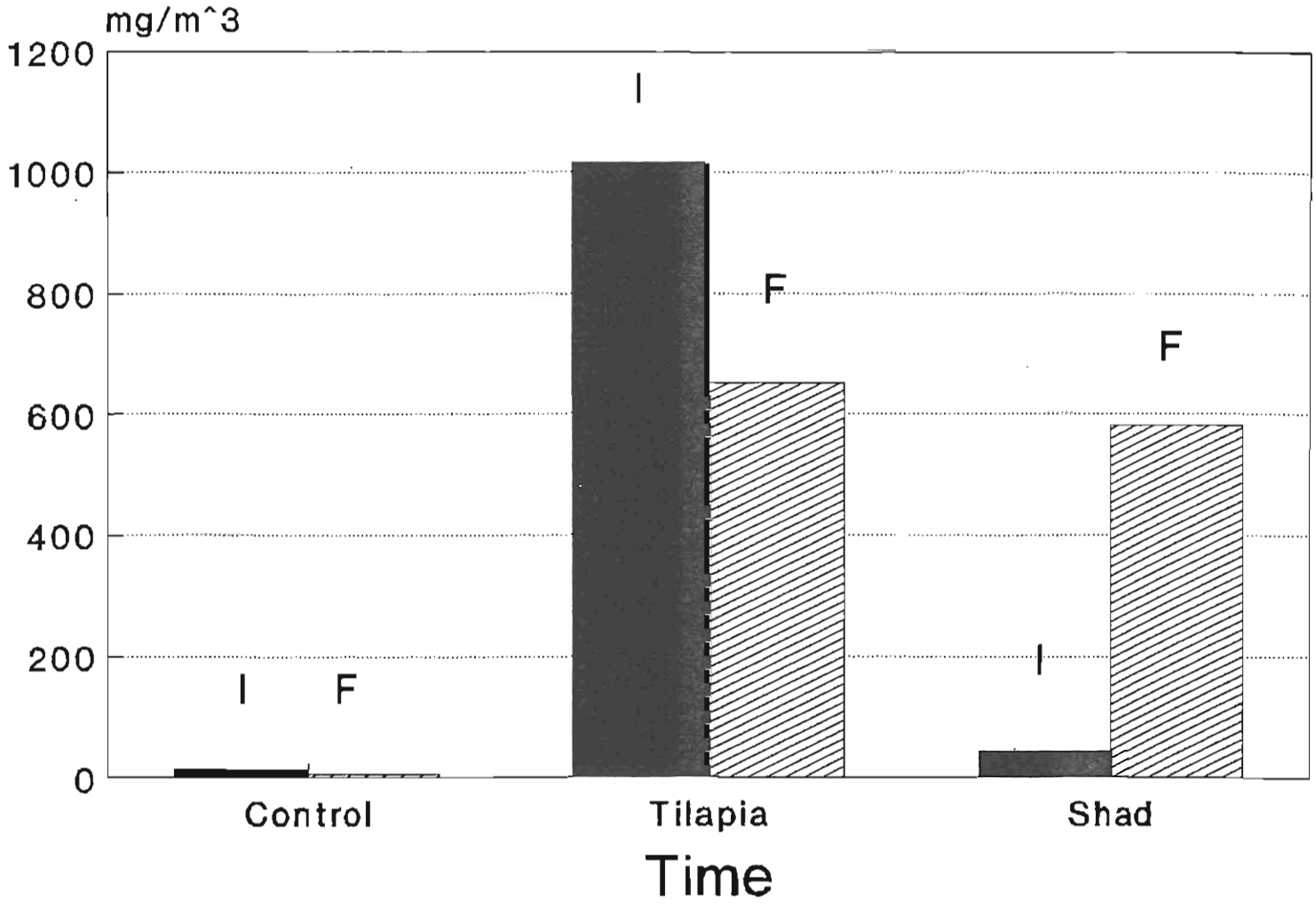

Figure 3-3. Bioassay initial (I) and final (F) values of chl. a ($mg.m^{-3}$) with M.aeruginosa in medium -N.

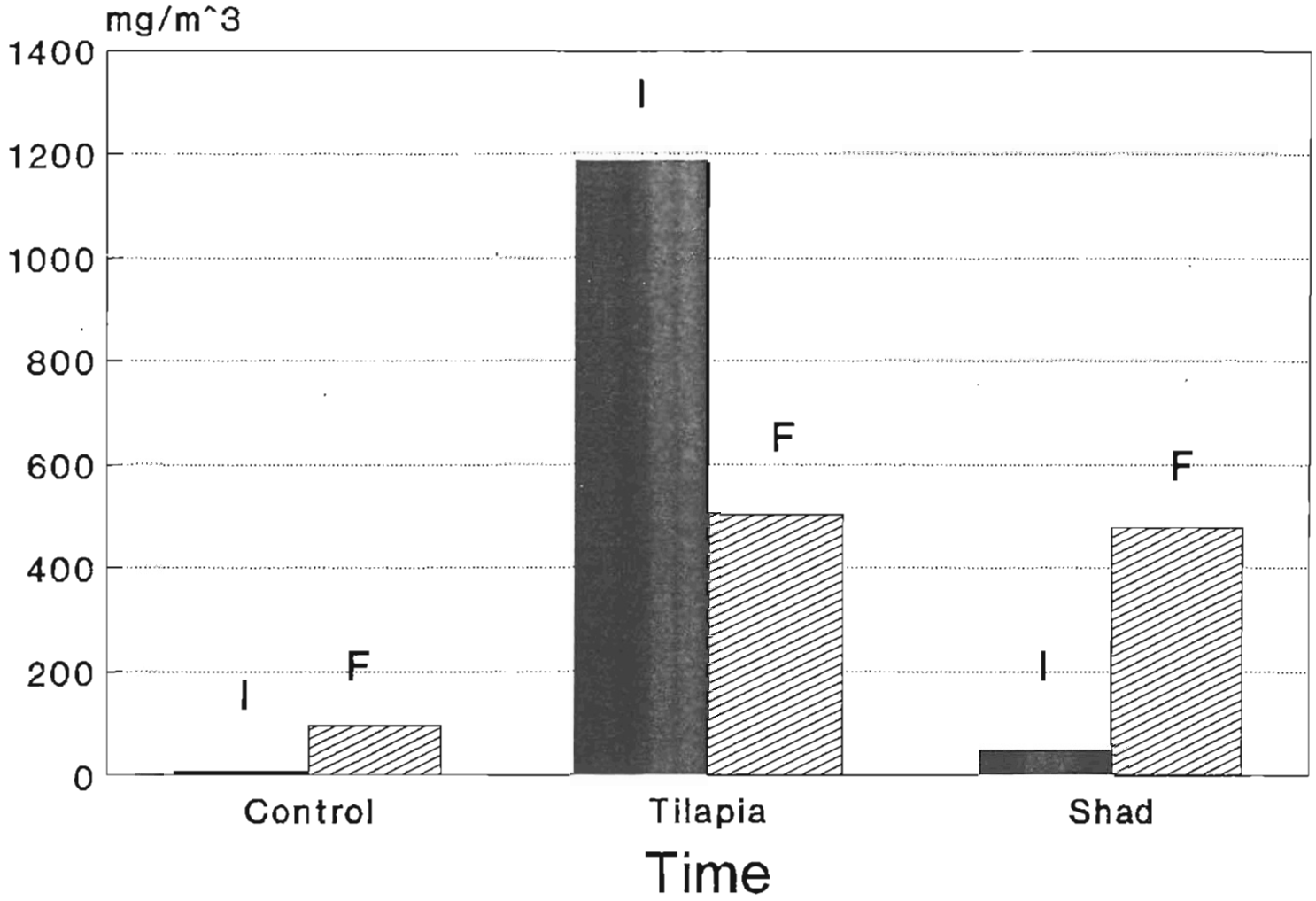

Figure 3-4. Bioassay initial (I) and final (F) values of chl. a ($mg.m^{-3}$) with M.aeruginosa in medium -P.

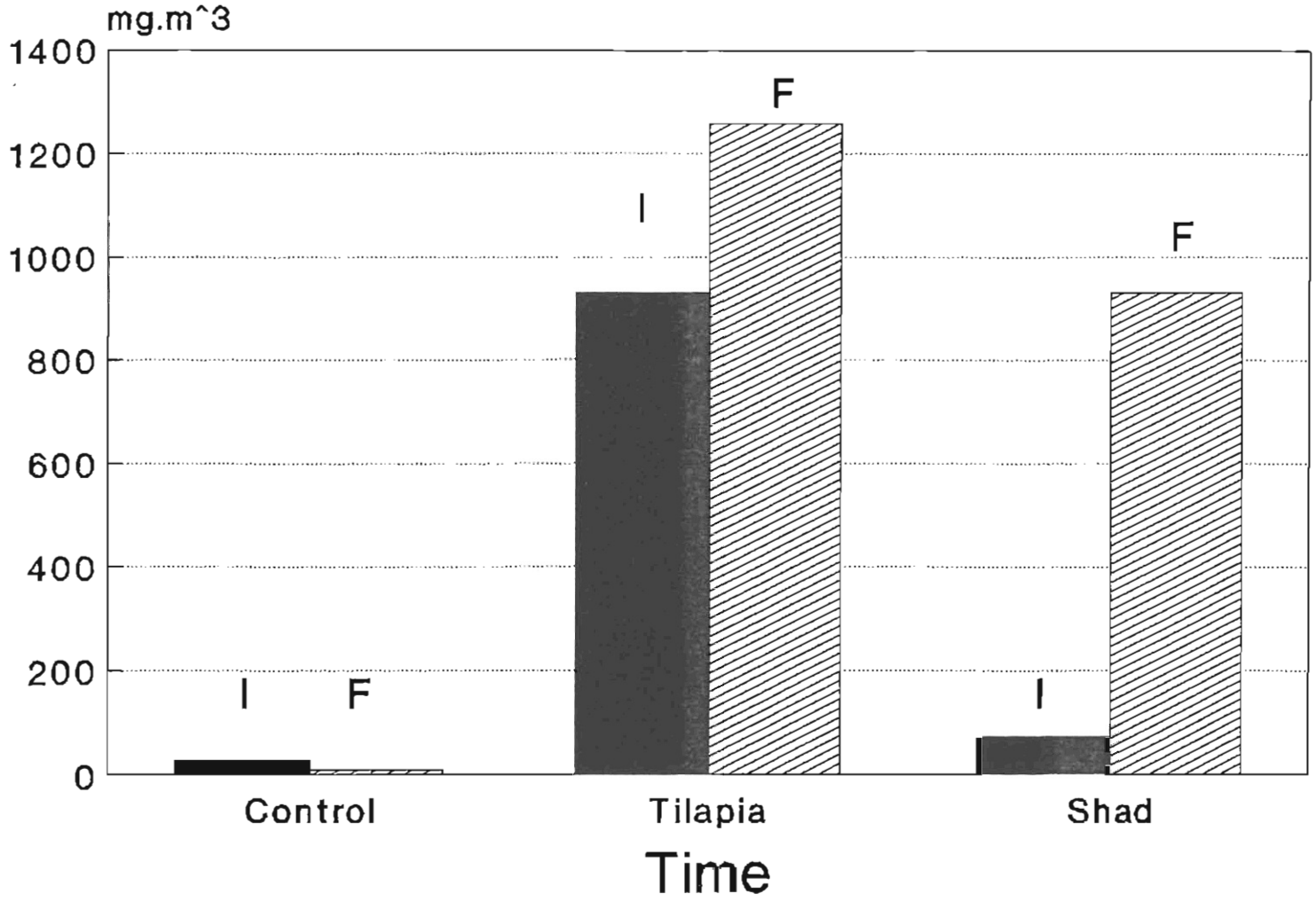

Figure 3-5. Bioassay initial (I) and final (F) values of chl. a ($mg.m^{-3}$) with S.capricornutum in medium -N.

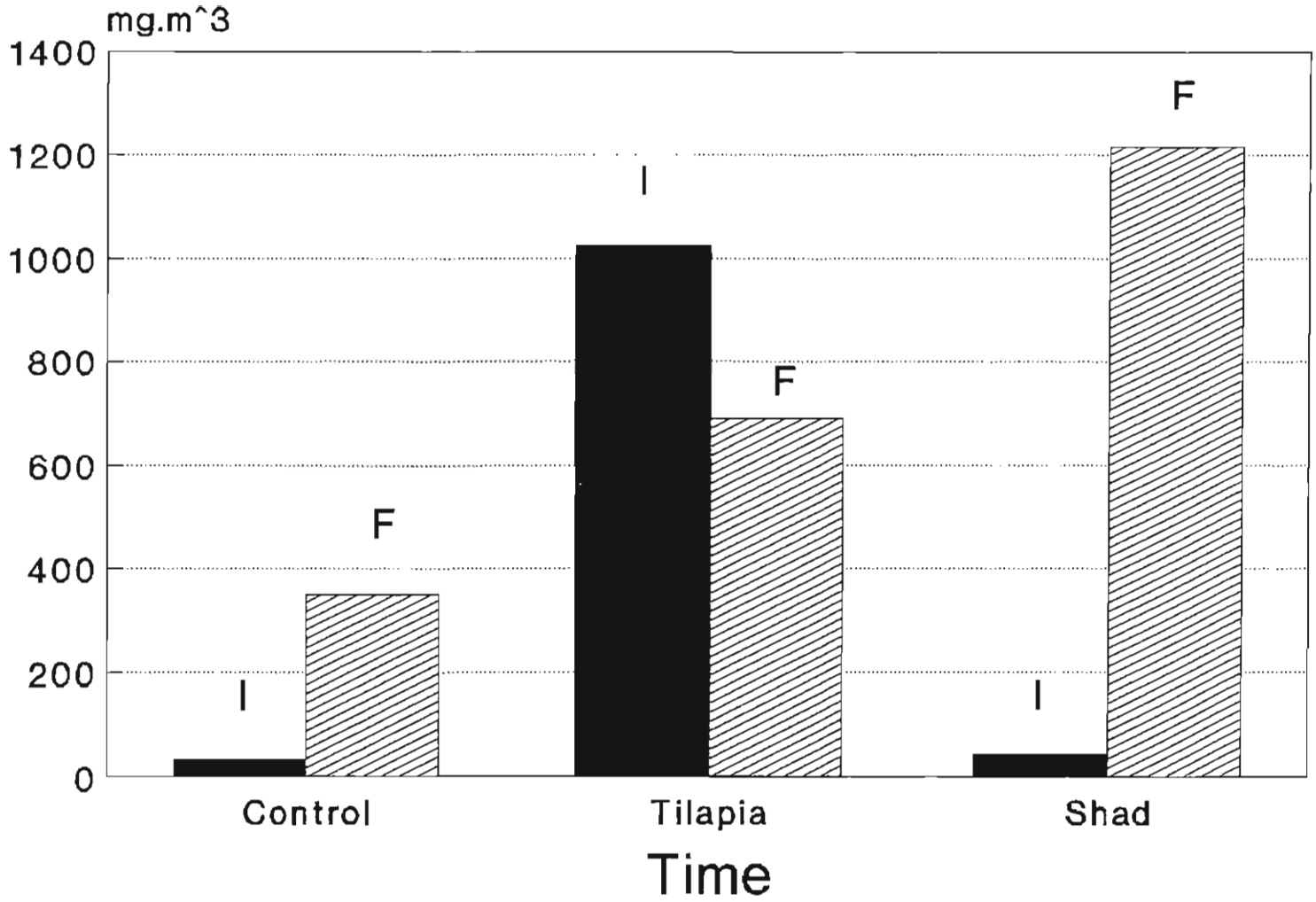

Figure 3-6. Bioassay initial (I) and final (F) values of chl. a ($mg.m^{-3}$) with S.capricornutum in medium -P.

(Figures 3-3 and 3-4), while in the Selenastrum tests, about 66% chlorophyll *a* had negative values indicating suppression, with the remaining values being positive with a mean of 600 $mg.m^{-3}$ (Figures 3-5 and 3-6).

McDonald (1985) found that blue tilapia enhanced the growth of Ankistrodesmus cells. Ware et al.(1975) concluded that blue tilapia had displayed little potential for algal control in Florida waters, and in fact, had spread rapidly and become a nuisance. Phytoplankton biomass in the fish fecal material calculated from chlorophyll *a* values at the beginning and end of the current bioassay showed that blue tilapia suppressed phytoplankton biomass in 75% of the samples of blue green and green algae to a level exceeding 50% of the initial value. Gizzard shad increased phytoplankton biomass concentration in all samples of blue green and green algae more than tenfold from the initial values.

It has been reported (Crisman and Kennedy, 1982) that gizzard shad are not suitable for use as a biocontrol agent for phytoplankton because they have no impact on chlorophyll *a* values, productivity or phytoplankton densities, and they can promote lake eutrophication through elevation of orthophosphate concentrations and differential digestion of some algae groups, especially blue-greens and greens.

On the other hand, results of extensive work with tilapia are still inconclusive regarding the possible

importance of this species in biomanipulation schemes. It is known that adult blue tilapia employ filter-feeding as their primary feeding method, while they feed as visually-oriented, particulate-feeding zooplanktivores as juveniles (Yowell and Vinyard, 1993). It is assumed that minimization of feeding costs while maximizing net energy return has been the basis for the filter-feeding strategy. The remaining question is what causes these fishes to switch their feeding behavior.

Facultative planktivores that use both feeding modes interchangeably (Leon and O'Connel, 1969; Crowder and Binkowski, 1983; and Helfman, 1990) usually filter-feed on small, abundant zooplankton and particulate-feed on large, less abundant forms. Yowell and Vinyard (1993), suggest that this is a stratagem which maximizes net energy return in response to a changing prey situation. The reduction in zooplankton density may then allow an increase in non-grazed phytoplankton (Drenner et al., 1987; Vinyard et al., 1988).

The results of this study indicate that blue tilapia fecal material, either by its composition and/or the absence of some essential nutrient, completely suppressed Microcystis aeruginosa chlorophyll *a* production, and appears to have suppressed Selenastrum capricornutum chlorophyll *a* production in > 60% of the samples. Gizzard shad feces increased chlorophyll *a* production values in both blue-greens and greens (tenfold increase). This study suggests

that, in Florida systems with conditions similar to those of the lakes studied, biomanipulation techniques can mitigate cultural eutrophication if gizzard shad are replaced by blue tilapia.

CHAPTER 4
SEDIMENTS AND HISTORICAL ECOLOGY OF TWO CENTRAL FLORIDA URBAN LAKES

Introduction

In the absence of historical data, past lake conditions can be reconstructed based on interpretations of the sedimentary record. Assuming that lake sediments accumulate in an orderly fashion, paleolimnology may provide information about how the lake ecosystem has changed over time. Reconstructing the reason for past changes in the lake may be used to predict the lake's response to future management strategies (Smeltzer and Swain, 1985; Brenner et al., 1993).

Lake sediments originate from numerous sources. The main sources are both the biochemical substances produced by organisms, or resulting from their degeneration, and morphological pieces of specific organisms. The sediment constitution is influenced primarily by the geomorphology of the lake basin and the drainage basin (Wetzel, 1983). Paleolimnology has as a principal objective a formulation of general principles about the way lakes change with time (Livingstone, 1981; Cohen and Nielson, 1986; Johnson et al., 1991).

Florida has about 7800 lakes varying in size from 0.4 ha to over 180,000 ha. (Canfield and Hoyer, 1988). Lakes in Florida serve many agricultural, domestic, industrial and recreational purposes (Canfield and Hoyer, 1992). Outside the glaciated regions in North America, Florida lakes constitute the largest group of natural lakes (Hutchinson, 1957) on the continent and the most important group of solution basins (Hutchinson, 1957; Crisman, 1992)

Limnological characteristics and productivity from these lakes vary widely and range from oligotrophic to hypereutrophic (Canfield and Hoyer, 1988; Brenner et al. 1990; Beaver et al. 1981). Although Florida's aquatic systems are large in number and have a significant economic impact, water quality data have been collected for few lakes (>10%) (Brenner et al., 1990), and routine data acquisition began fairly recently, in the 1960s and 1970s (Huber et al., 1982).

Considering Florida's aquatic system dimensions and economic importance, little information has been reported on the sediments of Florida lakes, i.e. Flannery et al. (1982); Brenner and Binford (1988); Brenner et al. (1990); Stoermer et al. (1992); Gottgens (1992); Brenner et al. (1993). Here I provide information on the relationship between accumulation rates of sediment variables such as organic matter, water content, total nitrogen and total phosphorus

and estimated trophic status based on sedimentation rates ($g.cm^{-2}.yr$) for the last century.

Sediment cores collected from Lake Bonny and Lake Gibson (Polk County, Florida) were dated isotopically with ^{210}Pb and ^{137}Cs to estimate material accumulation rates in the sediment profile using radioactive decay of fallout ("unsupported") ^{210}Pb. These techniques have been widely used to detect changes in sediment accumulation rates due to urban development in a watershed (Smeltzer and Swain, 1985), clear cutting of vegetation (Oldfield et al., 1980) and major climatic events (Robins et al., 1978).

In this study, these techniques were applied to identify lake sediment profile physicochemical characteristics and evaluate the potential impact of gizzard shad feces when it is the dominant filter feeder fish on lake sediment composition and net accumulation rates of organic matter, total nitrogen and total phosphorus for Lakes Bonny and Gibson. Whenever possible, this information will be correlated with the results from sediment cores for Lake Parker and Lake Hollingsworth reported by Schelske et al. (1992). For this study, both lakes were considered to be dominated by blue tilapia population as the main filter feeder fish.

According to the classification of Forsberg and Ryding (1980), Lake Bonny is hypereutrophic and Lake Gibson is an eutrophic lake. Mean values for all physicochemical and

biological parameters are provided in Chapter 5 of this study. Both are small urban lakes subject to anthropogenic pressures in the form of industrial, residential and commercial development in the area and recreational uses such as boating, fishing, and skiing. In addition, there are agricultural activities at Lake Gibson watershed.

Study Sites

Lake Bonny is a small (144 ha of surface area), shallow (z_{max}= 2.5 m), urban lake bordered by the city of Lakeland, (Polk County, Florida). It has no known discharges or withdrawals (Polk County Water Resources Division, 1990).

The population of Lakeland has increased from less than 500 inhabitants in 1880 to nearly 75,000 people in 1993. As a result, all of the lake watershed was developed for commercial and residential uses. Lake Bonny has consistently had poor water quality (Polk County Water Resources Division, 1990) with an average 1992 TSI of 73.5. Urban runoff from commercial and residential development in the watershed and low lake levels resulting from seepage have been a major problem. Extremely poor water quality was recorded in 1986 at a time of extreme low lake levels (Polk County Water Resources Division, 1990). Based on current data, the TSI was 98; total nitrogen 9.0 $mg.l^{-1}$, total phosphorus 0.7 $mg.l^{-1}$, Secchi 0.2 m and chlorophyll *a* was 220 $mg.m^{-3}$.

Using the classification system of Forsberg and Ryding (1980), Lake Bonny is classified as hypereutrophic. Average total phosphorus concentration is 59 $\mu g.l^{-1}$ and average total nitrogen concentration is 1858 $\mu g.l^{-1}$. Total chlorophyll a concentrations average 40 $\mu g.l^{-1}$, the water clarity as measured by a Secchi disc averages 0.6 m, and average pH is 7.8. Table 5-10 exhibits water quality data for Lake Bonny.

Before 1985, the plant community of Lake Bonny was comprised mainly of Hydrilla verticillata. Due to the macrophyte chemical control program employed by the City of Lakeland, it has been replaced by Typha sp. as the dominant macrophyte. Typha sp. is responsible for a percent lake area coverage (PAC) of 10% (cf. Canfield and Hoyer, 1992).

The fish population in this lake has changed dramatically. The species that survived the drought of 1983 are currently repopulating the lake. The most abundant open-water species collected in experimental gillnets are gizzard shad and Florida gar with 17.0 and 11.3 fish/net/24 hr, respectively (Canfield and Hoyer, 1992).

Lake Gibson is a small (192 ha of surface area), shallow lake (z_{max}= 6.1 m) located on the outskirts of the City of Lakeland in a rapidly urbanizing area where citrus groves and pastures are giving way to large-scale commercial and moderate-density residential development. A domestic wastewater treatment plant from one local elementary school

discharges to a wetland north of Lake Gibson, which overflows into the lake. There are no known withdrawals (Polk County Water Resources Division, 1990).

Lake Gibson is classified as eutrophic according to Forsberg and Ryding's (1980) classification system. The 1992 Florida TSI for Lake Gibson is 55 (Polk County Water Resources Division, 1990). The water is slightly tannic (color is 43 cpu) which lowers the Secchi disc value and may bias the TSI value. The average total phosphorus concentration is 94 $\mu g.l^{-1}$, and the average total nitrogen concentration is 1570 $\mu g.l^{-1}$. Total chlorophyll *a* concentration averages 23 $\mu g.l^{-1}$ and the water clarity as measured by use of a Secchi disc averages 1.3 m. Lake's annual primary productivity (mean = 241.18 $mgC.m^{-3}.h$ ± 229.1) displaying noticeable seasonal variability and the presence of blooms of cyanobacteria every few years suggests an eutrophic status for Lake Gibson. Water quality data for Lake Gibson are shown in Table 5-8.

The dominant macrophytes of the lake are water primrose, alligatorweed, and pennywort, which are responsible for 5% lake area coverage (Canfield and Hoyer, 1992).

No recent fish population evaluation was obtainable for this lake. From the 1979 evaluation, ten species of fish were reported, with channel catfish being dominant.

Methods

An 89 cm mud-water interface core was collected from the middle of Lake Bonny using a piston corer with a 12 cm diameter, 1.83 m long cellulose acetate butyrate (CAB) core barrel. Externally, the core was characterized by a light color sediment for the first 10 cm and a typical black mud color below in the rest of the core.

A 60 cm core was taken from the central portion of Lake Gibson using the same piston corer described above. The presence of sand mixed with mud was noticed beginning at the 50 cm depth. Sand content increased deeper in the core. Due to the soupiness of the sediments, in both cores a spoon was used to collect the samples down to the 10-cm layer. From that point on, collection was possible using a spatula. The sediment from Lake Gibson was more consolidated on the bottom than the sediment from Lake Bonny.

Concentrations of ^{210}Pb and ^{137}Cs were measured by direct γ-assay using a P-type, intrinsic-germanium detector (Princeton Gamma Tech). The counting system used for spectral analysis is located in the University of Florida's Department of Environmental Engineering Sciences Low Background Counting Room. The electronics for the system include a preamplifier (RG11B/C, Princeton Gamma Tech,Inc.), amplifier (TC 242, Tennelec), bias supply (5 kV, TC 950, Tennelec), power supply (TC 909, Tennelec), and transformer (Sola).

Samples for isotope analysis were dried at 95°C for 24 hours, pulverized by mortar and pestle, weighed, and placed in small plastic vials. Core sections were combined (up to 4 cm) to obtain an adequate sample weight (> 1 g). Vials were sealed with plastic cement and left for 14 days to equilibrate radon (^{222}Rn) with radium (^{226}Ra). Counting times were never less than 23 hours. Standards were counted to track efficiency (counts per γ) and to calculate a ^{226}Ra conversion factor (PCi per count per sec.). Blanks were counted to determine background radiation.

Bulk density and water content of sediments were measured by weighing samples before and after drying at 95°C. Percent organic matter was evaluated by loss on ignition (LOI) at 550°C for one hour (Håkanson and Jansson, 1983). Nitrogen was measured as total Kjeldahl nitrogen (TKN) using a Technicon-II semi-automated manifold after digestion following Bremner and Mulvaney (1982) but modified to exclude selenium as a catalyst. The digestate was also used for total phosphorus (TP) determinations. Liberated orthophosphate was determined with the ascorbic acid method (APHA, 1985).

Results and Discussion

The Lake Bonny Record

The 89 cm Lake Bonny sediment core had a mean bulk density of 0.094 $g.cm^{-3}$ ± 0.042, mean dry matter of 9.16% ±

3.93, mean organic matter of 56.00% ± 12.69, mean total nitrogen of 22.32 $mg.g^{-1}$ ± 2.68 and mean total phosphorus of 4.10 $mg.g^{-1}$ ± 1.53 (Table 4-1).

Lake Bonny had high organic content in surface deposits, but this declined somewhat with depth and age down to 19 cm of the core which coincides with the 1980s, and then increased again to the bottom where it reached highest values (dated at the beginning of this century, 1909) (Figure 4-1). Total nitrogen and total phosphorus plotted against time were constant throughout the whole core (Figure 4-2). Bulk density increased in the top 40 cm of the core which coincides with a ^{210}Pb determination of sediment age of approximately 1950, i.e., about the time when much agriculture in the watershed was being converted to urban development (USGS Topographic Maps, Lakeland Quadrangle, 1944, 1975).

Total P reconstruction near the base of the core suggests that the lake has always been mesotrophic to eutrophic. However, the lake has experienced periods of much higher nutrient enrichment, and values for total P in excess of 5.0 $mg.g^{-1}$ were computed for several sections that post-date the ^{137}Cs peak of 1960s and 1970s. No change in total P concentration has been noticed since then, suggesting that the lake trophic state has remained the same for the last thirty years.

Table 4-1. Sediment core physical and chemical characteristics from two central Florida urban lakes.

PARAMETER		LAKE GIBSON			LAKE BONNY	
	Mean	St. Dev.	Range	Mean	St. Dev.	Range
Bulk Density ($g.cm^{-3}$)	0.279	0.319	0.094 1.644	0.094	0.042	0.021 0.218
Dry Matter (%)	21.61	13.93	9.47 76.49	9.16	3.93	2.24 19.81
Organic Matter (%)	38.17	14.85	1.73 59.56	56.0	12.69	32.72 76.71
Nitrogen ($mg.g^{-1}$)	9.33	2.99	1.46 13.46	22.32	2.68	18.07 28.00
Phosphorus ($mg.g^{-1}$)	1.86	1.49	0 4.57	4.10	1.53	1.51 6.38

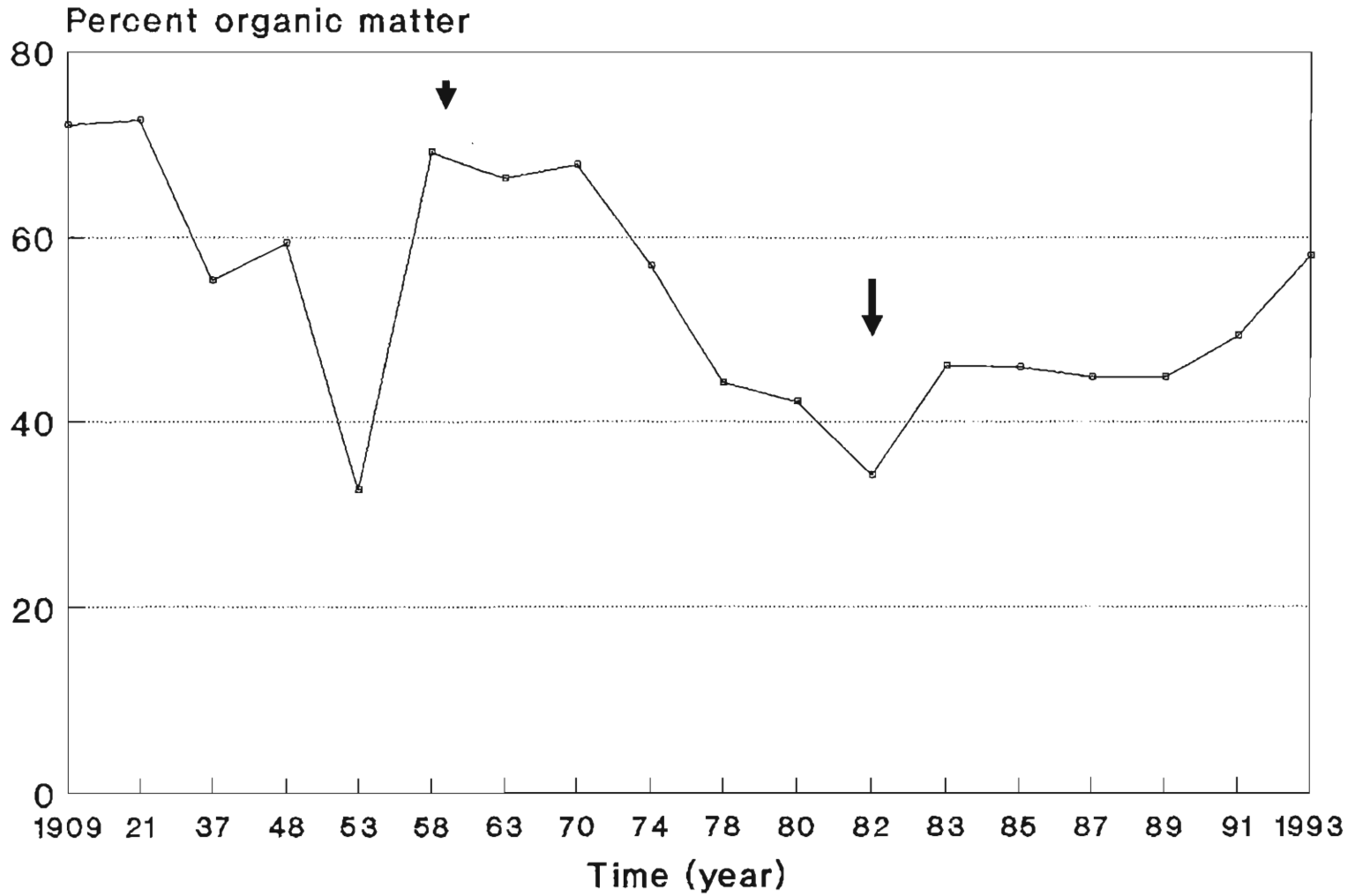

Figure 4-1. Lake Bonny percent organic matter sediment core content versus time (year).

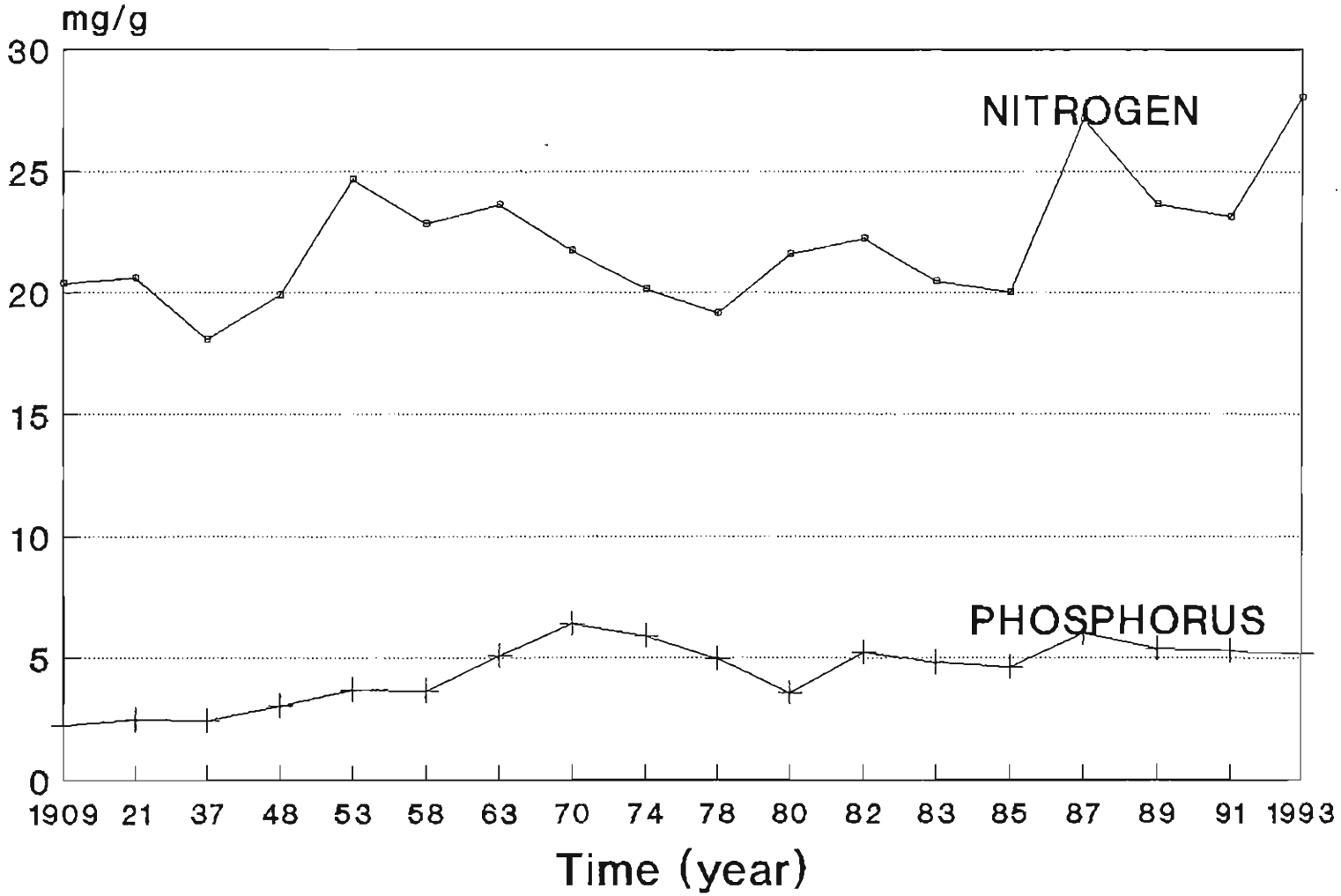

Figure 4-2. Lake Bonny TN ($mg.g^{-1}$) and TP ($mg.g^{-1}$) sediment core content versus time (year).

Sedimentation rates expressed in Figure 4-3 for Lake Bonny are in agreement with the lake history. The first peak of ± 0.1 $g.cm^{-2}.yr$ coincides with the 1955 development boom in the area, and the second peak of 0.13 $g.cm^{-2}.yr$ in 1983 coincides with the time when the lake was almost completely dry.

Detectable ^{137}Cs, which is an human-produced radionuclide that was injected into the atmosphere as a consequence of nuclear weapons testing in the early 1950s, matched well with the determined ^{210}Pb chronology (Table 4-2). This agreement is remarkable, considering the shallowness of the lake and the flocculence of the bottom substrate which makes these upper sediments vulnerable to physical disturbance. Eutrophic systems such as Lake Bonny, however, accumulate sediments rapidly, so these physical disturbances likely affect short time-intervals only.

The Lake Gibson Record

The sediment core from Lake Gibson had a mean bulk density of 0.279 $g.cm^{-3}$ ± 0.319, mean dry matter of 21.61% ± 13.93, mean organic matter of 38.17% ± 14.85, mean total nitrogen of 9.33 $mg.g^{-1}$ ± 2.99, and mean total phosphorus of 1.86 $mg.g^{-1}$ ± 1.49. (Table 4-1).

The organic content of Lake Gibson sediments was somewhat low in surface deposits (30% organic matter), increased with depth to about the midpoint of the core, matching with 1955, and declined to its lowest values of

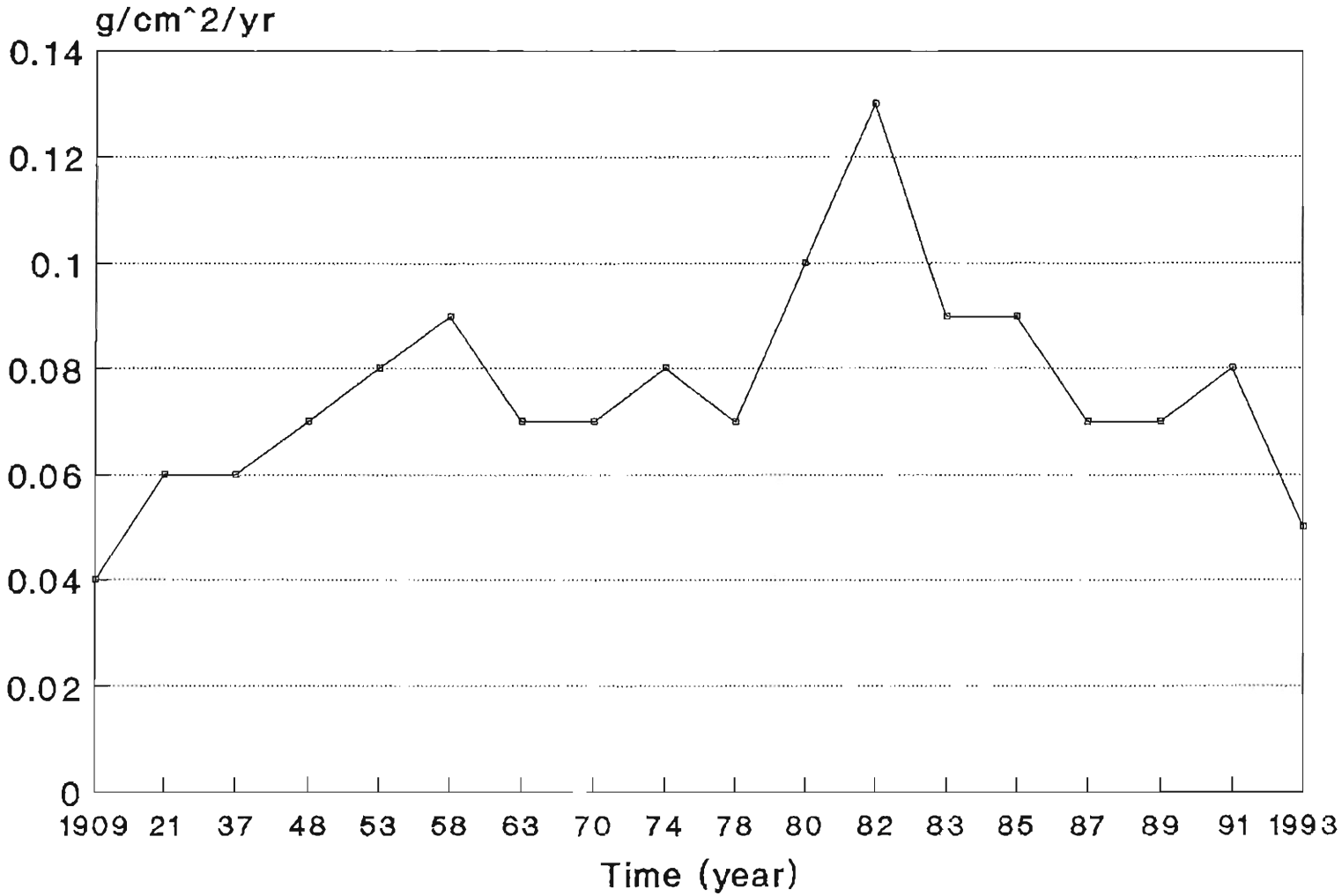

Figure 4-3. Lake Bonny sedimentation rate ($g.cm^{-2}.yr$) versus time (year).

Table 4-2. Lake Bonny sediment analysis.

Core Section (cm)	Bulk Density $g.cm^{-3}$	^{210}Pb $pCi.g^{-1}$	^{137}Cs $pCi.g^{-1}$	Deposition Period year	Deposition Rate $g.cm^{-2}.yr$	TN $mg.g^{-1}$	TP $mg.g^{-1}$
0 - 4	0.029	18.72	1.5	1993	0.05	28.00	5.16
4 - 7	0.050	12.30	3.0	1991	0.08	23.11	5.29
7 - 9	0.061	12.50	4.13	1989	0.07	23.62	5.35
9 - 11	0.066	11.22	4.01	1987	0.07	27.15	6.03
11 - 13	0.074	8.46	3.65	1985	0.09	20.02	4.62
13 - 15	0.078	8.14	3.20	1983	0.09	20.45	4.81
15 - 17	0.079	5.27	4.02	1982	0.13	22.26	5.21
17 - 19	0.090	6.99	4.56	1980	0.10	21.58	3.53
19 - 22	0.092	8.92	4.49	1978	0.07	19.15	4.93
22 - 26	0.092	6.51	3.28	1974	0.08	20.12	5.86
26 - 30	0.090	7.00	2.67	1970	0.07	21.72	6.38
30 - 34	0.089	6.00	1.07	1963	0.07	23.63	5.07
34 - 38	0.091	3.80	0.85	1958	0.09	22.85	3.64
38 - 42	0.102	3.43	0.92	1953	0.08	24.67	3.68
42 - 46	0.164	3.48	0.87	1948	0.07	19.9	3.03
46 - 50	0.174	3.17	0.78	1937	0.06	18.07	2.42
50 - 54	0.142	1.91	0.91	1921	0.06	20.59	2.49
54 - 58	0.137	1.94	0.59	1909	0.04	20.36	2.22

1.4% organic matter at the bottom of the core (approximately 1892) (Figure 4-4). Total nitrogen and total phosphorus displayed high variability. Total nitrogen peaked about 1942, declined after that and peaked again in 1990. Total phosphorus started with low values (1.0 $mg.g^{-1}$) about 1892, increased steadily to peak after 1966, and, from that point on, kept increasing to a 1990 value of 5.0 $mg.g^{-1}$ (Figure 4-5).

Total P reconstruction at the base of the core suggests that this lake, on the basis of phosphorus concentrations, has been oligotrophic in the past. Nevertheless, values for total P in excess of 3.0 $mg.g^{-1}$ and greater were recorded for all levels corresponding to the last 25 years (since 1966), reflecting changes in watershed use.

Bulk density increased below the 24 cm core, which matched with a ^{210}Pb determined sediment age of approximately 1950. Detectable ^{137}Cs coincided with the determined ^{210}Pb chronology (Table 4-3). The sedimentation rate for Lake Gibson corresponded with the lake's history. A significative peak of 0.12 $g.cm^{-2}.yr$ coincided with 1955, the beginning of the development boom for the area (Figure 4-6).

The organic matter content of dry bulk sediment in Florida lacustrine surface mud averaged 39.7% from 97 surveyed lakes (Brenner and Binford, 1988). Lake Gibson had a percent organic matter mean value of 38.2%, while Lake

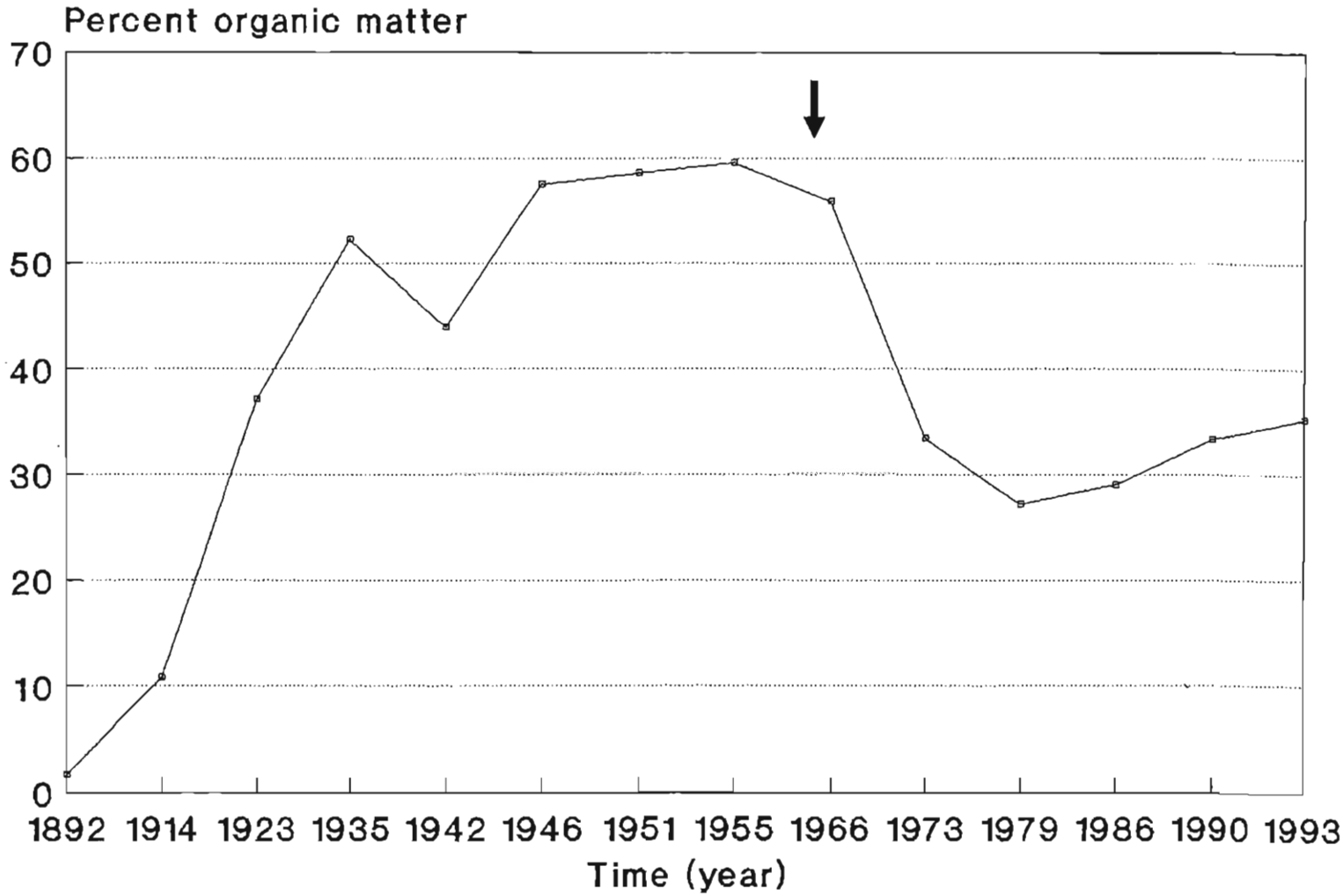

Figure 4-4. Lake Gibson percent organic matter sediment core content versus time (year).

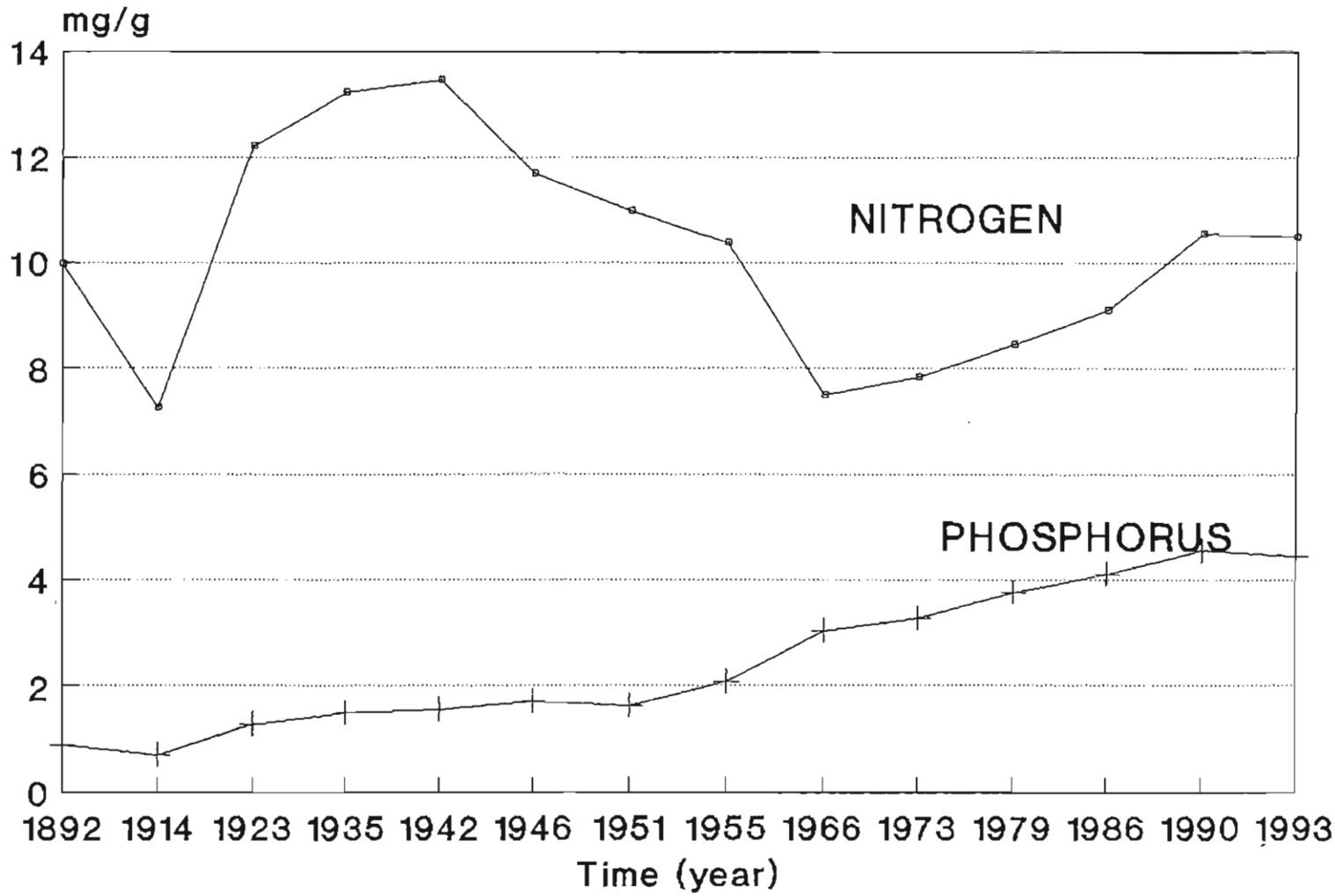

Figure 4-5. Lake Gibson TN ($mg.g^{-1}$) and TP ($mg.g^{-1}$) sediment core content versus time (year).

Table 4-3. Lake Gibson sediment analysis

Core Section (cm)	Bulk Density $g.cm^{-3}$	^{210}Pb $pCi.g^{-1}$	^{137}Cs $pCi.g^{-1}$	Deposition Period year	Deposition Rate $g.cm^{-2}.yr$	TN $mg.g^{-1}$	TP $mg.g^{-1}$
0 - 2	0.113	11.86	4.41	1993	0.07	10.50	4.45
2 - 4	0.167	9.48	5.02	1990	0.08	10.56	4.57
4 - 6	0.221	9.31	4.18	1986	0.08	9.11	4.10
6 - 8	0.233	6.76	3.63	1979	0.08	8.46	3.77
8 - 10	0.271	5.72	2.96	1973	0.08	7.83	3.28
10 - 12	0.284	6.07	2.96	1966	0.06	7.49	3.03
12 - 14	0.229	2.33	1.32	1955	0.12	10.38	2.07
14 - 16	0.178	2.63	1.28	1951	0.09	11.00	1.62
16 - 18	0.168	2.61	0.84	1946	0.08	11.70	1.70
18 - 20	0.160	3.34	0.75	1942	0.05	13.46	1.55
20 - 24	0.161	2.29	0.71	1935	0.06	13.21	1.50
24 - 28	0.185	1.11	0.75	1923	0.09	12.22	1.28
28 - 32	0.208	1.42	0.57	1914	0.05	7.26	0.70
32 - 36	0.183	2.00	0.30	1892	0.02	9.98	0.89

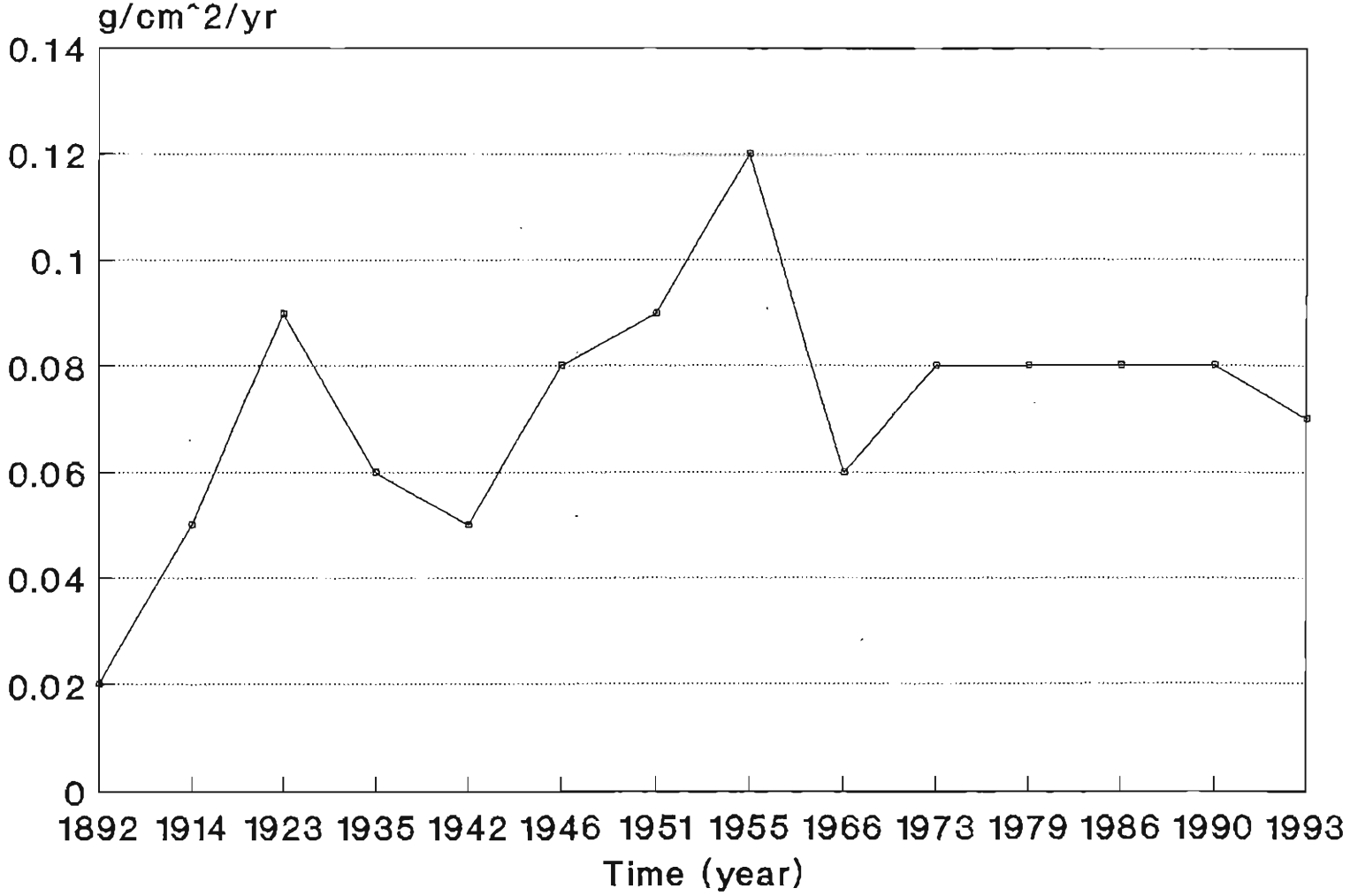

Figure 4-6. Lake Gibson sedimentation rate ($g.cm^{-2}.yr$) versus time (year).

Bonny had 56.0%. Total nitrogen for the same array of lakes displayed high variability, ranging from 0.6 to 42.4 $mg.g^{-1}$ (Brenner and Binford, 1988). Nitrogen concentration for Lake Gibson was 9.33 $mg.g^{-1}$ (Table 4-4), and for Lake Bonny was 22.32 $mg.g^{-1}$ (Table 4-5). Total P concentrations recorded by Brenner and Binford (1988) varied from 0.07 to 8.09 $mg.g^{-1}$. Values for Lake Gibson were 1.8 $mg.g^{-1}$ (Table 4-6) and for Lake Bonny was 4.1 $mg.g^{-1}$ (Table 4-7).

Considering that these lakes overlie phosphatic limestone deposits and are located in urban settings, each one of those isolated factors or a combination of both could be responsible for the high total P values. The total P value registered for Lake Bonny of 4.1 $mg.g^{-1}$ is higher than what was found in 93 of the 97 systems studied for Brenner and Binford (1988).

The Lake Parker Record

The Lake Parker sediment core displayed usually elevated deposition rates for bulk density, organic matter and phosphorus (Brenner et al., 1993). Bulk sediment accumulation rates increased from 11 $mg.cm^{-2}.yr^{-1}$ in 1922 to a maximum of 131 $mg.cm^{-2}.yr^{-1}$ in the late 1950s (Brenner et al., 1993). Phosphorus net accumulation rates increased more than ten times during the last century, reaching the highest value of 600 $\mu g.cm^{-2}.yr^{-1}$ in the late 1970s (Brenner et al., 1993). Appendices D and E show physical and chemical properties of Lake Parker sediment core.

Table 4-4. Total nitrogen ($mg.g^{-1}$) for Lake Gibson sediment core.

Interval (cm)	TN (mg/g)
0-2	10.4995
2-4	10.5649
4-6	9.1080
6-8	8.4608
8-10	7.8340
10-12	7.4951
12-14	10.3830
14-16	11.0008
16-18	11.6984
18-20	13.4574
20-24	13.2149
24-28	12.2212
28-32	7.2597
32-36	9.9815
36-40	10.4658
40-44	10.6539
44-48	7.2012
48-52	3.3171
52-56	1.4618
56-60	10.4134
Mean	9.33

Table 4-5. Total nitrogen ($mg.g^{-1}$) for Lake Bonny sediment core.

Interval (cm)	TN (mg/g)
0-2	28.0057
2-4	26.5107
4-7	23.1092
7-9	23.6192
9-11	27.1484
11-13	20.0185
13-15	20.4545
15-17	22.2599
17-19	21.5829
19-22	19.1497
22-26	20.1250
26-30	21.7168
30-34	23.6266
34-38	22.8466
38-42	24.6686
42-46	19.9064
46-50	18.0703
50-54	20.5948
54-58	20.3690
58-64	24.4444
64-72	19.5555
72-84	23.2875
Mean	22.32

Table 4-6. Total phosphorus ($mg.g^{-1}$) for Lake Gibson sediment core.

Interval (cm)	TP (mg/g)
0-2	4.4495
2-4	4.5700
4-6	4.1031
6-8	3.7696
8-10	3.2824
10-12	3.0312
12-14	2.0766
14-16	1.6263
16-18	1.6984
18-20	1.5514
20-24	1.5047
24-28	1.2811
28-32	0.7041
32-36	0.8948
36-40	0.9506
40-44	0.9809
44-48	0.6574
48-52	0.1171
52-56	0.0097
56-60	0.0000
Mean	1.8

Table 4-7. Total phosphorus ($mg.g^{-1}$) for Lake Bonny sediment core.

Interval (cm)	TP (mg/g)
0-2	5.1567
2-4	5.2729
4-7	5.2941
7-9	5.3534
9-11	6.0308
11-13	4.6168
13-15	4.8145
15-17	5.2071
17-19	3.5261
19-22	4.9261
22-26	5.8654
26-30	6.3821
30-34	5.0733
34-38	3.6455
38-42	3.6842
42-46	3.0309
46-50	2.4239
50-54	2.4907
54-58	2.2233
58-64	2.0648
64-72	1.5072
72-84	1.5842
Mean	4.1

The Lake Hollingsworth Record

Inferences from paleolimnological core studies indicate that Lake Hollingsworth has had high nutrient and chlorophyll *a* concentrations since the 19th century; however, the highest levels occurred between the 1950s and the 1970s, when much of the agriculture in the watershed was converted to urban development (Schelske et al., 1992). Lake sediments' topmost 10 cm is poorly consolidated (densities of <0.032 g.dry.cm^{-3}.wet) (Schelske et al., 1992), although density generally increases with depth and is highest in sand-rich deposits.

Organic matter in the core generally ranges between 40 and 60% of dry weight but decreases in sandy deposits (Schelske et al., 1992). Total phosphorus concentrations have remained relatively constant during the last ten years, ranging from 59 to 71 μg.L^{-1} (Schelske et al., 1992). Appendix F show physical and chemical properties of Lake Hollingsworth sediment core.

Conclusions

The data presented provide support for the conclusion of Flannery et al.(1982) that Florida lakes of higher trophic state have greater proportions of organic matter in their surface sediments e.g. Lake Bonny (TSI of 73.46) with 57% organic matter and Lake Gibson (TSI of 55.18) with 35% organic matter. Organic matter in the Lake Parker

superficial layers was >70.0%, and in Lake Hollingsworth was >51.0% (Schelske et al., 1992). Lake Bonny organic matter is positively correlated with depth and shows no correlation with any other parameter. Bulk density was positively correlated with depth and negatively correlated with total nitrogen and total phosphorus at the 5% level confidence (SAS, 1989) (Table 4-8).

Organic matter in Lake Gibson was negatively correlated to bulk density and showed a positive correlation to total nitrogen. Bulk density was positively correlated with depth, and depth was negatively correlated to total phosphorus at the 5% level confidence (SAS, 1989) (Table 4-9).

Using the reconstruction of historical limnological conditions, paleolimnology may help address issues of lake management (Smeltzer and Swain, 1985) and restoration. Sedimentary records may reflect past trophic status of a lake and, thereby, assist in setting a goal for restoration. Models may then be employed to identify primary nutrient sources and to foresee whether mitigation actions can reduce loading sufficiently to improve water quality or if other management techniques such as biomanipulation may be needed.

When reduction of nutrient loading to a lake is feasible, a cost/benefit evaluation should be completed to ascertain which restoration techniques are financially practical. Then, paleolimnological records can help in the decision making process.

Table 4-8. Correlation matrix for sediment variables for Lake Bonny.

SEDIMENT VARIABLES	CORRELATION	P VALUE
Depth/ Bulk Density	0.7848	0.0000
Depth/ Dry Matter	0.8255	0.0000
Depth/ Organic Matter	0.5482	0.0123
Depth/ Total Phosphorus	-0.8135	0.0000
Bulk Density/T Nitrogen	-0.5724	0.0084
Bulk Density/T Phosphorus	-0.6747	0.0011

Table 4-9. Correlation matrix for sediment variable for Lake Gibson.

SEDIMENT VARIABLES	CORRELATION	P VALUE
Depth/ Bulk Density	0.5862	0.0066
Depth/ Total Phosphorus	-0.9054	0.0000
Bulk Density/ Org. Matter	-0.7335	0.0002
Org. Matter/T Nitrogen	0.6123	0.0041

Historical water-column total P data collected for the last four years at Lake Bonny showed little variation, with the exception of Spring 1991, when the values were double the mean (mean = 0.112 $mg.l^{-1}$). An increase in chlorophyll a and total bacteria were also reported for the same period. This information is in disagreement with Maceina and Soballe (1990), who suggested that resuspension during wind events is probably responsible for the highly variable total P and chlorophyll *a* values measured in many shallow Florida lakes.

Lake Gibson historical records for total P showed no significant variation for the last four years, with a mean of 0.210 $mg.l^{-1}$. Chlorophyll *a*, TSI and total bacteria also displayed no notable variations. Anthropogenic alterations in the watershed, especially increased urbanization, may be responsible for the presence of only one genus of blue-green algae in the lake, as well as for the occasional algal bloom.

The Lake Parker historical record for net phosphorus accumulation rates greatly exceed dangerous P loadings to a basin of <5 m mean depth (13 $\mu g.cm^{-2}.yr^{-1}$) (Brenner et al., 1993). Total phosphorus measured in the water column during this study had a mean of 0.22 $mg.L^{-1}$ (0.07 to 0.602 $mg.L^{-1}$).

Lake Hollingsworth historical total phosphorus concentration show an overall increase upwards in the sediment core (Schelske et al., 1992). Total P values

changed from 2.03 $mg.g^{-1}$ at the base of the core to 6.60 $mg.g^{-1}$ at the top (Schelske et al., 1992). Total phosphorus in the water column measured during this study had a mean of 0.27 $mg.L^{-1}$ (0.081 to 0.61 $mg.L^{-1}$)

Due to the geology in the area, Lake Bonny has been an eutrophic lake for much of its recent history, even with conditions of minimal human occupation in the watershed (Figure 4-7). This should be the foundation for any management plan for this lake. Any plans for water quality improvement in the lake must consider that the lake is edaphically phosphorus-rich, and even a great effort to reduce phosphorus concentrations in the water may provide little or unnoticeable improvement in water clarity.

Lake Gibson, meanwhile, had a different paleolimnological record that displayed alterations during the last 25 years (Figure 4-8) in which the lake changed from mesotrophic to eutrophic, reflecting the switch in past largely agricultural watershed uses to large scale commercial and moderate density residential development. Any attempt to reduce nutrient concentration in this lake must consider control of point source external loading of nutrients as well as use of biomanipulation techniques as a means to achieve water quality improvement.

Figure 4-9 show the historic variation of organic matter content, nitrogen and phosphorus in the Lake Hollingsworth sediment core cf. Schelske et al. 1992. It is

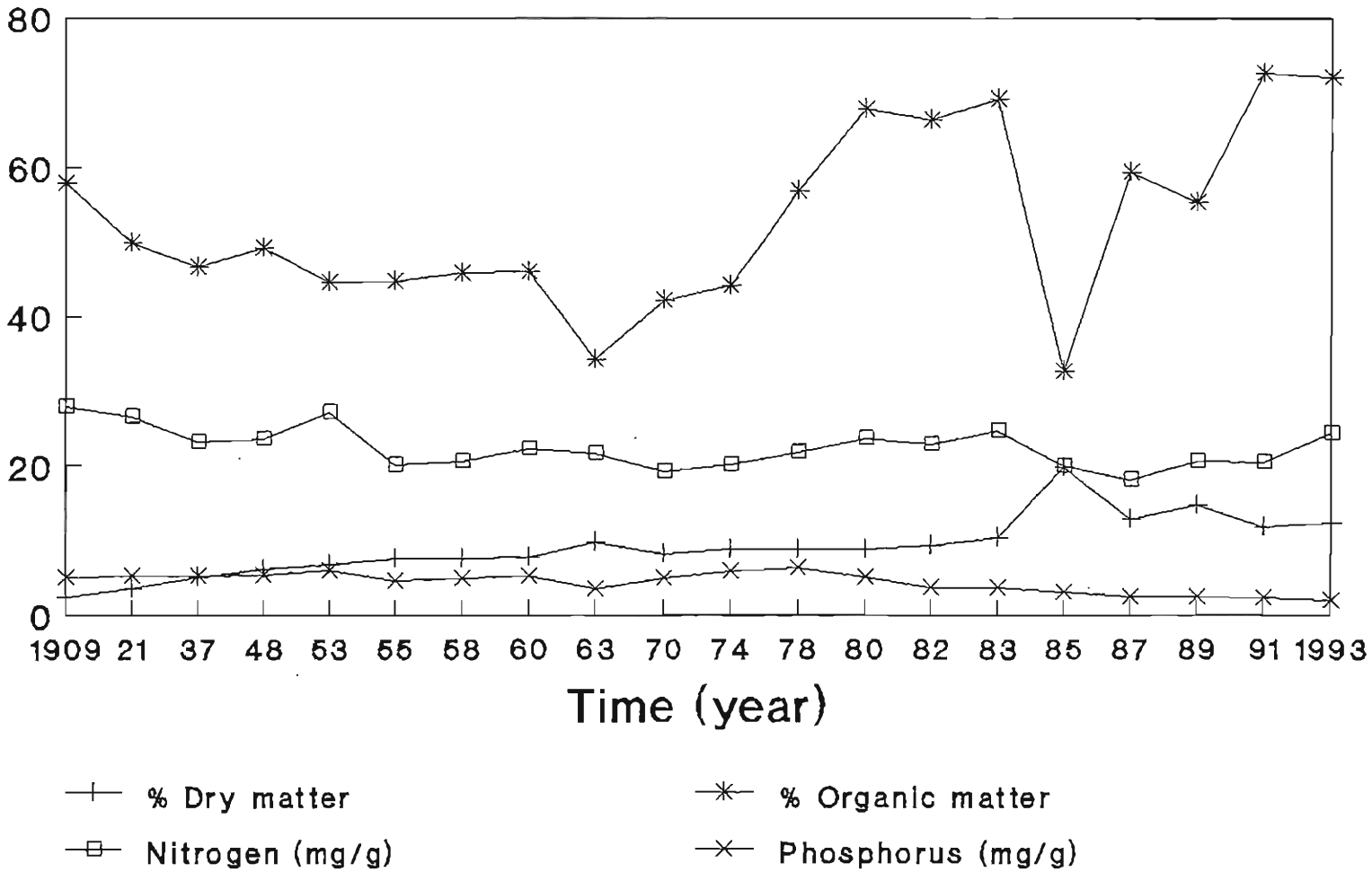

Figure 4-7. Changes in some physico-chemical characteristics in Lake Bonny through time.

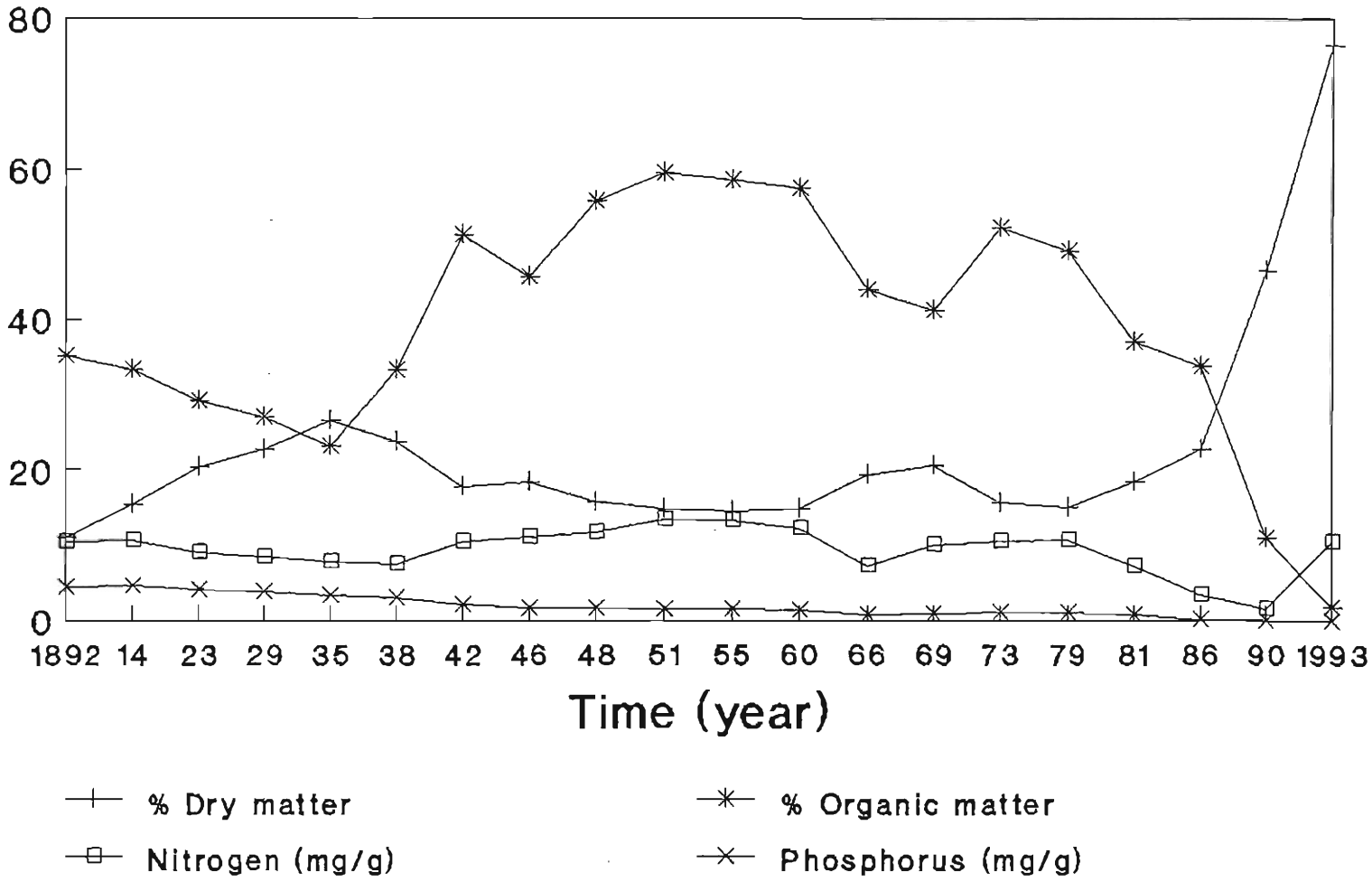

Figure 4-8. Changes in some physico-chemical characteristics in Lake Gibson through time.

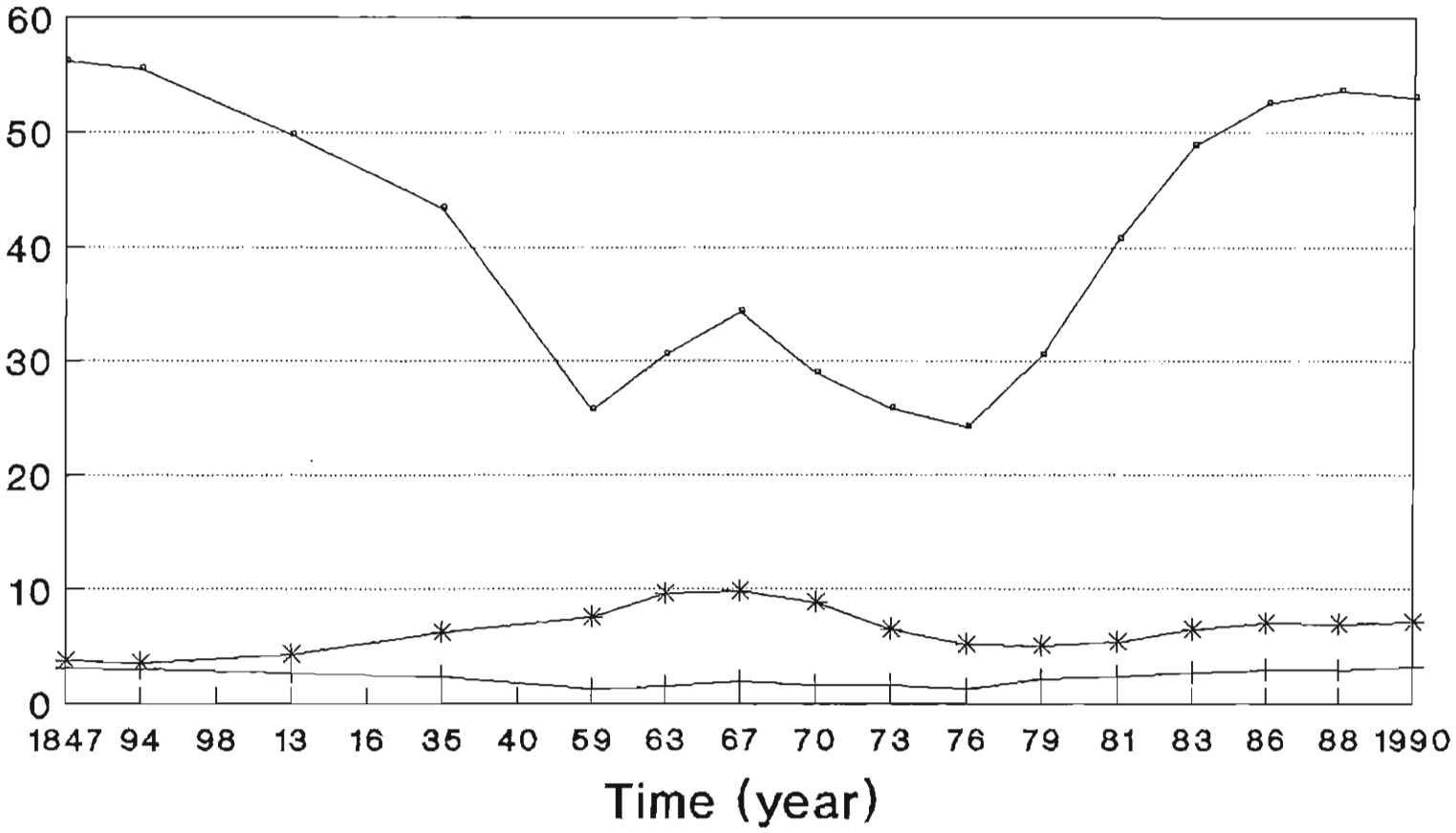

Figure 4-9. Changes in some physico-chemical characteristics in Lake Hollingsworth through time (after Schelske et al. 1992).

noted that then was an increase in phosphorus concentration after the 1950s, matching with the period of more intense occupation in the watershed for the whole area. Water quality measurements collected since 1966 demonstrate that the lake has been hypereutrophic for at least 25 years (Schelske et al., 1992). Although the lake is naturally productive, anthropogenic impacts have accelerated the rate of eutrophication. Eutrophication problems, especially from cultural enrichment, should be resolved in a number of ways, ranging from eutrophication prevention and lake rehabilitation, to learning to live with the problem.

From the sediment core analysis of Lakes Parker and Hollingsworth, lakes considered as having a dominant blue tilapia population in this study, and Lakes Bonny and Gibson, considered as having a dominant gizzard shad population, it is not clear that there are any significant differences in sediment organic matter content, nitrogen, phosphorus or any other analyzed parameter. All four lakes seem to have high organic matter content in the topmost layers, and all of them had a record in the sediments of the main environmental changes for the area e.g. the 1950's switch from totally agricultural practices in the area to watershed conversion to urban development, as well as the efforts been made to recover these systems since the mid 1980's.

Sediment cores from Lakes Bonny and Gibson collected during this study showed no noticeable changes relative to the improvement of any water quality parameter. However, Lakes Parker and Hollingsworth had what perhaps could be called a modification. According to Schelske et al. (1992), total phosphorus reconstructions in Lake Parker's topmost 15 cm (after 1960s) of sediment suggested a progressive decline in the water-column that may signal some reversal of cultural eutrophication. The Lake Hollingsworth core revealed that, for at least in the last ten years, total phosphorus inferences have remained relatively constant (Schelske et al., 1992).

Taking in consideration that blue tilapia came into the systems after 1961, it is reasonable to assume, first of all, a lag time for the establishment of the fish population in the new environment, and, secondly, time for the manifestation of any modification that those fishes would bring to the systems. Since more than thirty years have passed since fish introduction, it is possible to hypothesize that this could be responsible for changes recorded in Lakes Parker and Hollingsworth water quality, even though this might still provide a weak indication of water quality improvement. However, considering that all lakes included in this study are subject basically to the same kind of impacts, as well as the lake recovery program carried out by the City of Lakeland, the explanation for

this slight noticeable improvement and/or stabilization process in the water measured parameters in those lakes may be due to the either an altered biological community and/or the modifications resulting from the introduced species.

CHAPTER 5
LIMNOLOGICAL ASSESSMENT

Introduction

Florida can be divided into three major physiographic zones: the Northern, Central and Southern zones (Puri and Vernon, 1964; White, 1970). All lakes reported in this study are located in Polk County, Florida. Polk County lies within the Central Highlands physiographic province. The vast majority of the county lies within the Polk and Lake Uplands. The Polk Uplands is an area of continuous high ground located between the Gulf Coastal Lowlands and the western edge of the Lake Wales Ridge (White, 1970). Along the eastern ridge, deposits of the Fort Preston Formation occur, but most of the region is underlain by deposits of the phosphatic Hawthorn and Bone Valley Formations (Puri and Vernon, 1964). In the eastern half of the Polk Uplands, most lakes are in association with sand ridges. Polk County has 550 lakes covering 37.9×10^3 ha.

A majority of phosphatic sand deposits and clays of the Miocene Bone Valley and Hawthorne Formations are located in the Polk Uplands and Lake Uplands physiographic regions (Puri and Vernon, 1964). Florida has led the nation in

phosphate production for over 90 years; the bulk of this production is from Polk County (Boyle and Hendry, 1985). The trophic status for lakes of this region is mostly mesotrophic or eutrophic, and waters are moderately organically stained (Canfield, 1981).

Water quality on the Polk Upland is highly variable. Based on data from Florida Game and Fresh Water Fish Commission, Canfield (1981), Polk County Water Resources Division and this study, mean pH ranged from 5.4 to 9.4 and total alkalinity concentrations averaged between 1 and 76 $mg.l^{-1}$ as $CaCO_3$. Total hardness concentration averaged between 14 and 195 $mg.l^{-1}$ as $CaCO_3$, and mean specific conductance ranged from 82 to 281 $\mu mhos.cm^{-1}$. Calcium was the dominant cation and bicarbonate the dominant anion in Lake Parker, whereas in most of the other lakes in the area, sodium and chloride were the dominant anions. Total nitrogen concentration was between 306 and 1566 $mg.m^{-3}$, mean total phosphorus concentration from 2.4 to 144 $mg.m^{-3}$, and Secchi disk depth 0.2 to 2.7 m. The wide range of water chemistry may be directly related to different regional geology (Canfield, 1981; Crisman, 1993).

The following agencies have conducted studies or developed information on the lakes in Polk County: United States Geological Survey (USGS); Environmental Protection Agency (EPA); Polk County Water Resources Division (PCWR); City of Lakeland (CL); Florida Game and Freshwater Fish

Commission (FGFWFC); Southwest Florida Water Management District (SWFWMD); Central Florida Regional Planning Council (CF208).

Methods

I selected six lakes in Polk County, Florida: Lakes Parker, Hollingsworth, Hancock, Gibson, Bonny and Hunter. For this study all lakes were sampled quarterly during the years of 1992 and 1993 following approved U.S. EPA methodology. Additional data for the years of 1989 to 1991 were obtained from the above listed agencies.

Three water quality sampling stations were established for Lakes Parker, Hollingsworth and Hancock, and two stations were established for Lakes Gibson, Bonny and Hunter. Water was collected from just bellow the surface (0.5 m) in acid-cleaned Nalgene bottles. Samples were placed on ice and returned to the laboratory for analysis. Water temperature (°C), dissolved oxygen (mg/L), pH and conductance (μS/cm @ 25°C) were measured by using a Hydrolab Station. Secchi depth (m) was measured at each station where water was collected.

At the laboratory, total alkalinity (mg/L as $CaCO_3$) was determined by titration with 0.02 N H_2SO_4 (APHA, 1985). Total phosphorus concentrations (mg/L) were determined using the methods of Murphy and Riley (1962) after persulfate oxidation (Menzel and Corwin, 1965). Total nitrogen

concentrations (mg/L) were determined by a modified Kjeldahl technique (Nelson and Sommers, 1975). Total suspended solids (mg/L), organic suspended solids (mg/L) and inorganic suspended solids (mg/L) were determined according to standard methods (APHA, 1985). Water samples were analyzed for color (Pt-Co units) using the platinum-cobalt method and matched Nessler tubes (APHA, 1985). (The results of these laboratory analyses were provided by the aforementioned agencies).

Total chlorophyll *a* concentrations (μg/L) were determined by filtering a measured portion of lake water through a Gelman type A-E glass filter. Chlorophyll *a* was determined by using the method of Yentsch and Menzel (1963) and the equations of Parson and Strickland (1963). Primary productivity ($mgC.m^{-3}.h$) was measured using the light and dark bottle method according to Wetzel and Likens (1991). Phytoplankton samples for qualitative analysis were collected quarterly at one station in each lake during the 1992/93 survey. Samples were collected at mid-Secchi depth using a Van Dorn water bottle, and then dispensed into dark, polypropylene bottles containing Lugol's solution (APHA, 1985). At the laboratory, samples were refrigerated in the dark until examined. Zooplankton were sampled quarterly at one station in each lake during the 1992/93 survey. Samples were collected with an 8.0 L Kemmerer bottle and filtered with a 28 μm net, and were preserved with 10 percent sugar-

formalin (Haney and Hall, 1973) containing rose bengal stain. Oblique tows were started as close to the bottom as possible, and the sampler was raised to the surface at a slow, constant rate; this procedure was repeated four times, filtering up to 32 L of water. Fish populations were estimated from data of Canfield and Hoyer (1992) and Florida Game and Freshwater Fish Commission (FGFWFC).

Description of Study Area

Lake Parker

Lake Parker is located in Polk County, Florida (Appendix A). The basin lies in the Polk Uplands physiographic region and is situated in phosphatic deposits from the Hawthorn and Bone Valley Formations. Due to its location just east of the Lakeland Ridge, the lake receives inputs of groundwater which have been in contact with limestone (Stewart, 1966). Lake Parker is a large urban lake with a surface area of 924 ha, shoreline length of 19.8 km and mean depth of 2.2 m (Table 5-1).

Table 5-1. Morphometry of Lake Parker(1)

Surface Area	924 ha
Maximum Depth	3.0 m
Mean Depth	2.2 m
Development of Shoreline	70 %
Drainage Basin Area	6.18×10^3 ha
Shoreline Length	19.8 km
Macrophyte Cover	5.0 %

(1) Data from PCWR (1992).

The drainage area north of the lake receives water from Lake Gibson and a sinkhole basin on the west side. Overflow from Lake Mirror also enters Lake Parker. The lake also receives groundwater inputs. FMC Corporation discharges effluent to the lake by a drainage ditch and the City of Lakeland Power Plant withdraws and discharges water to the lake. Lake Parker has an outflow to Saddle Creek through a small canal and a water level control structure at the outlet.

Land-use in the basin is dominated by a large commercial and residential area. The entire northern section of the Lake Parker watershed has been impacted by phosphate mining. Dominant macrophytic plant species (defined as greater than or equal to 5% macrophyte cover cf. PCWR, 1992) include cattail, hydrilla, and water naiad. Macrophyte chemical control is employed for water hyacinths, water lettuce, and hydrilla.

Using the criteria of Forsberg and Ryding (1980), Lake Parker was hypereutrophic during this study. Canfield (1981) also classified Lake Parker as hypereutrophic. It is suggested that the trophic status of this lake has remained relatively stable for at least the last decade. Table 5-2 shows water quality data for Lake Parker.

Phytoplankton primary productivity was measured in October, December, April and July at surface, 50cm and 100 cm depths each time, using the light and dark bottle method. The results are expressed as gross photosynthesis ($mgC.m^{-3}.h$) cf. Wetzel and Likens (1991) (Table 5-3). Values for net photosynthesis are presented in table 5-4, community respiration in table 5-5, and table 5-6 shows the average annual productivity values for all study lakes. Lake Parker does not display great seasonality in primary productivity. The highest values were always found at the 50 cm depth, with a mean for that depth of 154.87 $mgC.m^{-3}.h \pm 42.51$ throughout the year. A minimum of 103.86 $mgC.^{-3}.h$ for April and a maximum of 203.12 $mgC.m^{-3}.h$ for December were recorded at the 50 cm depth. The lowest value recorded for the lake was 58.93 $mgC.m^{-3}.h$ at surface during July, while the maximum recorded for the lake was 203.12 $mgC.m^{-3}.h$ at 50 cm during December.

The phytoplankton community in Lake Parker was represented by blue-green algae including Anabaena sp., Microcystis sp., Merismopaedia sp., and Aphanizomenon;

Table 5-2. Summary of water quality data for Lake Parker. Numbers shown are mean values with ranges in parentheses.

Parameter	Unit	Canfield (1981)	Several Agencies * (1989/91)	This study (1992/93)
Temperature	°C	N/A	24.95 (16.4-31.8)	24.81 (17.7-30.2)
D.O.	mg/L	N/A	8.5 (4.7-13.2)	9.1 (8.1-10.3)
pH		8.9 (8.8-9.1)	8.9 (5.4-10.9)	8.5 (7.5-9.3)
Alkalinity	mg/L $CaCO_3$	68 (63-76)	55 (40-64)	43.5 (34-50)
Conduct.	µmhos/cm	215 (200-260)	387 (207-470)	247 (207-271)
Nitrogen	mg/L	1.56 (0.98-1.97)	3.15 (0.63-11.4)	3.13 (2.19-3.69)
NH_3-N	mg/L	N/A	0.06 (0.01-0.19)	0.05 (0.02-0.07)
TKN	mg/L	N/A	2.9 (0.6-11.0)	2.8 (1.3-3.2)
Phosphorus	mg/L	0.08 (0.06-0.11)	0.15 (0.02-.34)	0.22 (0.07-0.602)
Chl. a	mg/m^3	33.7 (9.3-49.3)	75.7 (18.6-222.8)	53.8 (43.0-76.6)
Color	Pt units	22 (15-30)	24.2 (6-57)	42.6 (29-53)
Secchi	m	0.5 (0.4-0.6)	0.4 (0.25-1.5)	0.35 (0.25-0.4)
Bacteria	x 10^6	N/A	761.7 (1-7800)	18.7 (1-60)
TSI		N/A	79.6 (62.7-94.0)	82.7 (76.6-86.9)

(*) AGENCY CODES:
USGS; EPA; PCWR; CL; UFC; FGFWFC; CF208; and SWFWMD

Table 5-3. Gross primary productivity for all study lakes expressed as ($mgC.m^{-3}.h$).

LAKE	DEPTH	08/92	12/92	04/93	07/93	MEAN	S.DEV
	0 cm	64.06	78.12	64.53	58.93	66.41	8.21
Parker	50 cm	140.62	203.12	103.86	171.87	154.87	42.51
	100 cm	70.31	62.51	69.43	78.12	70.09	6.39
	0 cm	178.90	437.5	356.78	506.34	369.88	141.2
Holl [a]	50 cm	46.87	31.25	304.65	406.25	197.25	187.4
	100 cm	46.87	30.98	146.35	187.5	102.92	76.06
	0 cm	356.45	468.75	187.5	625.0	409.42	184.4
Hanc [b]	50 cm	187.67	0	0	218.75	101.60	118.0
	100 cm	N/A	N/A	N/A	N/A	N/A	N/A
	0 cm	346.89	15.62	25.46	468.75	214.18	229.1
Gibson	50 cm	278.26	93.75	31.25	265.6	167.21	123.7
	100 cm	156.78	0	43.75	178.46	94.74	86.5
	0 cm	478.34	453.12	345.12	515.6	448.04	73.2
Bonny	50 cm	156.89	187.5	131.25	234.37	177.50	44.34
	100 cm	98.14	93.75	46.87	101.56	85.08	25.67
	0 cm	303.85	109.37	289.56	215.47	229.56	89.0
Hunter	50 cm	423.74	115.62	484.37	203.12	306.71	175.6
	100 cm	205.63	98.45	281.25	218.75	201.02	75.92

(a) Hollingsworth.

(b) Hancock.

Table 5-4. Net primary productivity for all study lakes expressed as ($mgC.m^{-3}.h$).

LAKE	DEPTH	08/92	12/92	04/93	07/93	MEAN	S.DEV
	0 cm	19.22	23.44	19.36	17.68	19.92	2.46
Parker	50 cm	42.18	60.94	31.16	51.56	46.46	12.75
	100 cm	21.09	18.75	20.83	23.44	21.03	1.92
	0 cm	53.67	131.25	107.03	151.90	110.96	42.37
Holl [a]	50 cm	14.06	9.37	91.39	121.87	59.17	56.23
	100 cm	14.06	9.29	43.90	56.25	30.87	22.82
	0 cm	106.93	140.62	56.25	187.5	122.82	55.33
Hanc [b]	50 cm	56.30	0	0	65.62	30.48	35.40
	100 cm	N/A	N/A	N/A	N/A	N/A	N/A
	0 cm	124.41	5.46	8.91	164.06	75.71	80.78
Gibson	50 cm	97.39	32.81	10.94	92.96	58.52	43.29
	100 cm	54.87	0	15.31	62.46	33.16	30.26
	0 cm	167.42	158.59	120.79	180.46	156.81	25.64
Bonny	50 cm	54.91	65.62	45.94	82.03	62.12	15.52
	100 cm	34.35	32.81	16.40	35.55	29.78	8.99
	0 cm	106.35	38.28	101.35	75.41	80.35	31.15
Hunter	50 cm	148.31	40.47	169.53	71.09	107.35	61.46
	100 cm	71.97	34.45	98.44	76.56	70.35	26.58

(a) Hollingsworth.

(b) Hancock.

Table 5-5. Community respiration for all study lakes expressed as ($mgC.m^{-3}.h$).

LAKE	DEPTH	08/92	12/92	04/93	07/93	MEAN	S.DEV
	0 cm	44.84	54.68	45.17	41.25	46.48	5.74
Parker	50 cm	98.44	142.18	72.7	120.31	108.41	29.76
	100 cm	49.22	43.76	48.6	54.68	49.06	4.47
	0 cm	125.23	306.25	249.75	354.44	258.92	98.86
Holl [a]	50 cm	32.81	21.88	213.26	284.38	138.08	131.2
	100 cm	32.81	21.69	102.45	131.25	72.05	53.24
	0 cm	249.52	328.13	131.25	437.5	286.6	129.1
Hanc [b]	50 cm	131.37	0	0	153.13	71.12	82.61
	100 cm	N/A	N/A	N/A	N/A	N/A	N/A
	0 cm	225.48	10.16	16.55	304.69	139.22	148.9
Gibson	50 cm	180.87	60.94	20.31	172.64	108.69	80.40
	100 cm	101.91	0	28.44	116.0	61.59	56.21
	0 cm	310.92	294.53	224.33	335.14	291.23	47.62
Bonny	50 cm	101.98	121.88	85.31	152.34	115.38	28.82
	100 cm	63.79	60.94	30.47	66.01	55.30	16.68
	0 cm	197.5	71.09	188.21	140.06	149.21	57.85
Hunter	50 cm	275.43	75.15	314.84	132.03	199.36	114.1
	100 cm	133.66	64.00	182.81	142.19	130.66	49.35

(a) Hollingsworth.

(b) Hancock.

Table 5-6. Average annual productivity values ($mgC.m^{-3}.h$) (+S.Dev.) at surface (1); 50 cm (2); and 100 cm (3) depths for all study lakes.

		Parker	Hollings.	Hancock	Gibson	Bonny	Hunter
	(1)	66.41 (8.21)	369.88 (141.2)	409.42 (184.4)	214.18 (229.1)	448.04 (73.2)	229.56 (89.0)
Gross prim. product.	(2)	154.87 (42.51)	197.25 (187.4)	101.6 (118.0)	167.21 (123.7)	177.5 (44.34)	306.71 (175.6)
	(3)	70.09 (6.39)	102.92 (76.06)	N/A	94.74 (86.5)	85.08 (25.67)	201.02 (75.92)
	(1)	19.92 (2.46)	110.96 (42.37)	122.82 (55.33)	75.71 (80.78)	156.81 (25.64)	80.35 (31.15)
Net primary product.	(2)	46.46 (12.75)	59.17 (56.23)	30.48 (35.40)	58.52 (43.29)	62.12 (15.52)	107.35 (61.46)
	(3)	21.03 (1.92)	30.87 (22.82)	N/A	33.16 (30.26)	29.78 (8.99)	70.35 (26.58)
	(1)	46.48 (5.74)	258.92 (98.86)	286.6 (129.10)	139.22 (148.91)	291.23 (47.62)	149.21 (57.85)
Respiration	(2)	108.41 (29.76)	138.08 (131.20)	71.12 (82.61)	108.69 (80.40)	115.38 (28.82)	199.36 (114.14)
	(3)	49.06 (4.47)	72.05 (53.24)	N/A	61.59 (56.21)	55.30 (16.68)	130.66 (49.35

greens such as Scenedesmus sp., Closterium sp., Pediastrum sp., Cosmarium sp. and Ankistrodesmus; the dinoflagellate Peridinium sp.; and diatoms of the genus Navicula sp.

A total of 41 zooplankton taxa were identified for Lake Parker (Kolasa, 1993). Rotifers were the most abundant taxa (27 out of 41), and comprised almost twice the number of taxa as all other zooplankton combined. Rotifers comprised 94.7 percent of the zooplankton on an annual basis (Kolasa, 1993). Brachionus sp., Keratella sp., Collurella sp., and Trichocerca sp., were the most numerically abundant rotifers. The most abundant adult crustacean zooplankton was Bosmina longirostris, a cladoceran (Kolasa, 1993).

Lake Parker's fish species richness of 21 species (Canfield and Hoyer, 1992) is expected for a lake of this size and is similar to that found in hypereutrophic Florida lakes. Canfield and Hoyer (1992) reported data from nine 0.08 ha blocknets (six placed in littoral habitats and three in open-water locations), and found that total fish biomass averaged 71.1 $kg.ha^{-1}$, with the average of harvestable sportfish of 31.7 $kg.ha^{-1}$. The greatest total fish biomass was collected in littoral blocknets set in hydrilla and tapegrass, whereas the lowest biomass was harvested in open-water blocknets. The dominant fish species collected in littoral nets were largemouth bass and bluegill, and the dominant fish species collected in open-water nets were gizzard shad and threadfin shad (Canfield and Hoyer, 1992).

ANOVAs were performed using the General Linear Model (GLM) (SAS, 1989) in order to identify significant seasonal limnological factors ($p < 0.05$). The following parameters were analyzed: Secchi, temperature, pH, conductivity, dissolved oxygen, total suspended solids, turbidity, chlorophyll *a*, color, alkalinity, NH_3, TKN, total nitrogen, total phosphorus, orthophosphate, and total bacteria.

The data were collected between 1989 and 1993 and grouped in four seasons: winter (December to February); spring (March to May); summer (June to August); and fall (September to November). The analyses of the data were according to the year of collection, season and the combination of year and season (Table 5-7).

Temperature, conductivity, dissolved oxygen, turbidity, and total bacteria are the parameters that exhibited significant seasonal variation at the 95% confidence level. The other parameters' variability displayed either interannual variations or no variation at all. When year and seasonality were combined, Secchi, conductivity, turbidity, NH_3 and total bacteria are the parameters that displayed significant variability.

Lake Hollingsworth

Lake Hollingsworth is located in Polk County, Florida (Appendix A). The basin lies in the Bartow Embayment division of the Central Lakes District (Canfield, 1981),

Table 5-7. Correlation analysis of water quality data for Lake Parker using the General Linear Model procedure (GLM). Data were combined by year, season and year*season.

PARAMETER	YEAR		SEASON		YEAR*SEASON	
	SIGNIF	N/SIGNF	SIGNIF	N/SIGNF	SIGNIF	N/SIGNF
Secchi	X			X	X	
Temperature		X	X			X
pH	X			X		X
Conductivity	X		X		X	
D.O.		X	X			X
TSS		X		X		X
Turbidity	X		X		X	
Chlorophyll *a*	X			X		X
Color		X		X		X
Alkalinity		X		X		X
NH_3	X			X	X	
TKN	X			X		X
TN	X			X		X
TP		X		X		X
ORT		X		X		X
BACTERIA	X		X		X	

where the bedrock is dominated by phosphatic deposits from the Hawthorn and Bone Valley Formation. Lake Hollingsworth is an urban lake situated within Lakeland, Florida, with a surface area of 145 ha, shoreline length of 4.34 km and mean depth of 1.5 m (Table 5-8).

Table 5-8. Morphometry of Lake Hollingsworth(1)

Surface Area	145 ha
Maximum Depth	2.44 m
Mean Depth	1.13 m
Development of Shoreline	100 %
Drainage Basin Area	495 ha
Shoreline length	4.34 km
Macrophyte Cover	3 %

(1) Data from PCWR (1992).

No known discharges or withdrawals were reported for Lake Hollingsworth. Its drainage basin is totally developed, and land-use within the basin includes residential and commercial development and Florida Southern College. The dominant macrophytic vegetation (defined as greater than or equal to 3% macrophyte cover cf. PCWR, 1992) includes American lotus, elephant ear, and cattails. Chemical control measures are employed for water hyacinth, water lettuce, and hydrilla.

Using the Forsberg and Ryding (1980) classification, Lake Hollingsworth was classified as hypereutrophic during this study. The lake was also classified as hypereutrophic

by Canfield and Hoyer (1992). Past limnological data indicate that this lake has been hypereutrophic for at least the last 25 years (PCWR, 1992). Table 5-9 shows water quality data for Lake Hollingsworth.

Lake Hollingsworth displayed seasonality for its gross primary productivity; the highest values were recorded in late fall and mid-summer. Overall, the surface layer had the highest values, with an annual mean of 369.88 $mgC.^{-3}.h$ ± 141.2; the minimum recorded value was 178.9 $mgC.^{-3}.h$, and the maximum was 506.34 $mgC.m^{-3}.h$ (Table 5-3). Net photosynthesis is reported in table 5-4, community respiration in table 5-5, and table 5-6 shows the average annual productivity values for all study lakes.

Historical phytoplankton data for Lake Hollingsworth from 1968 and 1971 (USGS, 1968, 1971) show the Cyanophyta as the dominant group, especially species of Oscillatoria sp., Lyngbya sp., and Raphidiopsis sp., followed by Chlorophyta and Diatoms. During this study, the phytoplankton community was almost completely dominated by the green algae Ankistrodesmus sp. and Cosmarium sp. and the blue green alga Spirulina sp. The latter was replaced sometimes by Aphanizomenon sp. and Anabaena sp. The diatoms were represented by the genus Navicula.

Rotifers were the dominant zooplankton group throughout the year. Keratella sp., Brachionus sp., and Monostyla sp., were the dominant taxa. The only adult copepod identified

Table 5-9. Summary of water quality data for L. Hollingsworth. Numbers shown are mean values with ranges in parentheses.

Parameter	Unit	Canfield (1981)	Several Agencies * (1989/91)	This study (1992/93)
Temperature	°C	N/A	25.8 (15.5-33.3)	25.3 (18.5-30.9)
D.O.	mg/L	N/A	9.7 (4.8-12.8)	9.4 (6.7-11.0)
pH		9.4 (8.8-10.1)	9.4 (7.8-11.3)	8.7 (7.8-10.43)
Alkalinity	mg/L $CaCO_3$	47 (32-56)	52.4 (38-66)	47.2 (40-58)
Conduct.	µmhos/cm	183 (165-205)	189.8 (142-230)	181.5 (158-209)
Nitrogen	mg/L	N/A	4.04 (0.75-12.2)	5.41 (2.8-8.1)
NH_3-N	mg/L	N/A	0.07 (0.006-0.3)	0.167 (0.036-0.7)
TKN	mg/L	N/A	3.92 (0.75-11.6)	5.15 (2.35-7.9)
Phosphorus	mg/L	0.774 (0.32-1.2)	0.254 (0.015-2.2)	0.268 (0.081-0.61)
Chl. a	mg/m^3	54.8 (26.7-104)	182.59 (30.1-368.1)	186.32 (64.5-285.9)
Color	Pt units	5 (5-5)	34.7 (6-89)	58.5 (37-94)
Secchi	m	1.6 (0.8-2.4)	0.3 (0.2-0.4)	0.22 (0.15-0.4)
Bacteria	x 10^6	N/A	2417.15 (1-17200)	800 (40-3700)
TSI		N/A	89.46 (70.3-103.1)	98.27 (88.2-105.6)

(*) AGENCY CODES:
USGS; EPA; PCWR; CL; UFC; FGFWFC; CF208; and SWFWMD.

was of the genus Diaptomus, but it did not represent a large component of the zooplankton population. Rotifers had an estimated density of 286,000 individuals.m^{-3}, and cladocerans amounted to 68,700 individuals.m^{-3} (Canfield and Hoyer, 1992).

Sixteen species of fish were collected from Lake Hollingsworth (Canfield and Hoyer, 1992). The fish population from 1965 to 1969 for this lake was described as having an overabundance of forage fish such as gizzard shad, threadfin shad, and stunted bluegill (Ware et al., 1971), and total fish biomass estimated with littoral blocknets ranged from 70 to 257 kg.ha^{-1} (Ware et al., 1971). The littoral nets in the Canfield and Hoyer report of 1992 averaged 1050 kg.ha^{-1}. The most abundant species collected by open-water experimental gillnets were gizzard shad and black crappie (Canfield and Hoyer, 1992).

ANOVAs were performed using the General Linear Model (GLM) (SAS, 1989) in order to identify significant seasonal limnological factors ($p < 0.05$). The following parameters were analyzed: Secchi, temperature, pH, conductivity, dissolved oxygen, total suspended solids, turbidity, chlorophyll *a*, color, alkalinity, NH_3, TKN, total nitrogen, total phosphorus, orthophosphate, and total bacteria.

Secchi, temperature, conductivity, turbidity, chlorophyll *a*, color, total phosphorus and total bacteria are the parameters that exhibited significant seasonal

variation at the 95% confidence level (Table 5-10). Other parameters displayed either interannual variations or no variation at all. When year and seasonality were combined, conductivity, turbidity, color, alkalinity and total bacteria were the parameters displaying significant variability.

Lake Hancock

Lake Hancock is located in Polk County, Florida (Appendix A). The basin lies in the Polk and Lake Uplands physiographic region, and it is situated in phosphatic deposits from the Hawthorne and Bone Valley Formations. Lake Hancock is a large lake with a surface area of 1839 ha, shoreline length of 16.7 km, and mean depth of 0.85 m (Table 5-11).

Table 5-11. Morphometry of Lake Hancock(1)

Surface Area	1839 ha
Maximum Depth	1.5 m
Mean Depth	0.85 m
Development of Shoreline	5 %
Drainage Basin Area	34.1 x 10^3 ha
Shoreline Length	16.7 km
Macrophyte Cover	1 %

(1) Data from PCWR (1992).

The lake has historically received wastewater discharges via Banana Lake from the City of Lakeland, and via Lake Lena Run from the City of Auburndale. Both have

Table 5-10 . Correlation analyses of water quality data for Lake Hollingsworth using the General Linear Model procedure (GLM). Data were combined by year, season and year*season.

PARAMETER	YEAR		SEASON		YEAR*SEASON	
	SIGNIF	N/SIGNF	SIGNIF	N/SIGNF	SIGNIF	N/SIGNF
Secchi		X	X			X
Temperature		X	X			X
pH	X			X		X
Conductivity		X	X		X	
D.O.		X		X		X
TSS		X		X		X
Turbidity		X	X		X	
Chlorophyll *a*		X	X			X
Color	X		X		X	
Alkalinity	X			X	X	
NH_3		X		X		X
TKN	X			X		X
TN	X			X		X
TP	X		X			X
ORT		X		X		X
BACTERIA		X	X		X	

been eliminated. Known discharges are coming from a Coca-Cola Citrus processing plant, Adams Citrus processing plant and Florida Distillers via Lake Lena Run. Agrico Chemical discharges directly into the lake. There are no known withdrawals.

Lake Hancock's drainage basin is extremely large and includes portions of the City of Lakeland (75,000 inhabitants) and Auburndale (18,000 inhabitants). Non-urban land-use areas is split between mined areas and agricultural development. The dominant vegetation (defined as greater than or equal to 1% of macrophyte cover cf. PCWR, 1992) includes cattails, pickerelweed, and pennywort. Macrophyte chemical control is employed for water hyacinth, water lettuce, and hydrilla.

Lake Hancock was reported as an alkaline, eutrophic, hard water lake by Canfield (1981). Using the criteria of Forsberg and Ryding (1980), Lake Hancock was classified as hypereutrophic during this study. Lake water quality has been improving for at least the last 10 years, although the lake is still the most eutrophic lake in Polk County (Polk County Water Resources Division, 1990). Table 5-12 shows water quality data for Lake Hancock.

Lake Hancock did not display much seasonality for gross primary productivity. The highest values were always measured at the surface with an annual mean of 409.42 $mgC.m^{-3}.h \pm 184.4$, minimum value of 187.5 $mgC.m^{-3}.h$ in April and

Table 5-12. Summary of water quality data for L. Hancock. Numbers shown are mean values with ranges in parentheses.

Parameter	Unit	Canfield (1981)	Several Agencies * (1989/91)	This study (1992/93)
Temperature	°C	N/A	25.71 (23.1-28.6)	23.65 (23.1-24.2)
D.O.	mg/L	N/A	7.91 (2.48-10.9)	6.54 (2.48-10.6)
pH		9.4 (8.8-10.0)	9.06 (7.84-9.66)	9.31 (9.03-9.59)
Alkalinity	mg/L $CaCO_3$	76 (36-106)	N/A	N/A
Conduct.	μmhos/cm	281 (240-340)	302.8 (232-394)	263 (232-294)
Nitrogen	mg/L	N/A	7.54 (4.07-12.55)	7.64 (4.07-11.22)
NH_3-N	mg/L	N/A	0.029 (0.018-0.057)	0.024 (0.018-0.03)
TKN	mg/L	N/A	7.54 (4.06-12.55)	7.64 (4.06-11.22)
Phosphorus	mg/L	2.44 (1.4-3.2)	0.56 (0.366-0.944)	0.65 (0.37-0.94)
Chl. a	mg/m^3	144 (40.1-217)	184.1 (66-349.5)	221.15 (131.8-310)
Color	Pt units	28 (25-30)	51.25 (37-75)	60 (45-75)
Secchi	m	0.8 (0.5-1.3)	0.2 (0.1-0.3)	0.25 (0.2-0.3)

(*) AGENCY CODES:
USGS; EPA; PCWR; CL; UFC; FGFWFC; CF208; and SWFWMD.

maximum of 625.0 $mgC.m^{-3}.h$ in July (Table 5-3). Values for net photosynthesis are shown in table 5-4, community respiration in table 5-5, and average annual productivity in table 5-6.

During this study, blue-green algae including Spirulina sp. and Anabaena sp. were the most abundant algae, followed by Microcystis sp., and Merismopedia sp. A small number of green algae such as Closterium sp. were also recorded.

The zooplankton community was dominated by rotifers, including Brachionus sp. and Keratella sp., and Diaptomus was the only copepod genus reported (Kolasa, 1993).

Fourteen species of fish were reported for Lake Hancock. Field collections from fifteen-minute electrofishing events in July 1991 and February 1992, indicated that blue tilapia was responsible for 71.4 and 52.0 % of total fish weight, respectively, while values for gizzard shad were 4.4 and 32.0 % by weight for the two sampling periods. The dominant fish species collected were blue tilapia, followed by black crappie in the 1991 sample and gizzard shad in 1992.

ANOVAs were performed using the General Linear Model (GLM) (SAS, 1989) in order to identify significant seasonal limnological factors ($p < 0.05$). The following parameters were analyzed: Secchi, temperature, pH, conductivity, dissolved oxygen, total suspended solids, turbidity, chlorophyll *a*, color, alkalinity, NH_3, TKN, total nitrogen,

total phosphorus, orthophosphate, and total bacteria. Secchi, temperature, conductivity, turbidity, chlorophyll *a*, color and total bacteria exhibited significant seasonal variation at the 95% confidence level (Table 5-13). Interannual variation or no variation at all was reported for the other parameters. When year and seasonality were combined, Secchi, conductivity, turbidity, color and total bacteria were the parameters displaying significant variability.

Lake Gibson

Lake Gibson is located in Polk County, Florida (Appendix A). The basin lies in the Polk Uplands physiographic region and is situated in sandy deposits of the Hawthorne and Bone Valley Formations. Lake Gibson is on the outskirts of the City of Lakeland, with a surface area of 192 ha, a shoreline length of 6.9 km, and a mean depth of 3.5 m (Table 5-14).

Table 5-14. Morphometry of Lake Gibson(1)

Surface Area	192 ha
Maximum Depth	6.1 m
Mean Depth	3.5 m
Development of Shoreline	65 %
Drainage Basin Area	1.1 x 10^3 ha
Shoreline Length	6.9 km
Macrophyte Cover	5 %

(1) Data from PCWR (1992).

Table 5-13. Correlation analyses of water quality data for Lake Hancock using the General Linear Model procedure (GLM). Data were combined by year, season, and year*season.

PARAMETER	YEAR		SEASON		YEAR*SEASON	
	SIGNIF	N/SIGNF	SIGNIF	N/SIGNF	SIGNIF	N/SIGNF
Secchi		X	X		X	
Temperature		X	X			X
pH		X		X		X
Conductivity	X		X		X	
D.O.		X		X		X
TSS		X		X		X
Turbidity	X		X		X	
Chlorophyll *a*		X	X			X
Color	X		X		X	
Alkalinity		X		X		X
NH_3		X		X		X
TKN		X		X		X
TN		X		X		X
TP		X		X		X
ORT		X		X		X
BACTERIA	X		X		X	

Discharge from Padgett Elementary School passes through a domestic wastewater treatment plant then goes into a wetland that overflows north of Lake Gibson. No known outflows were reported. The lake is located in a rapidly urbanizing area. Citrus groves and pastures are giving way to large-scale commercial and moderate-density residential development. Dominant macrophytic vegetation (defined as greater or equal to 5% macrophyte cover cf. PCWR, 1992) includes water primrose, alligator weed, and pennywort. Chemical controls are employed for water hyacinth, water lettuce, and hydrilla.

During this study, Lake Gibson was classified as a eutrophic lake according to the classification of Forsberg and Ryding (1980). Canfield (1981) classified this lake as a slightly acid, mesotrophic, soft-water lake. The Florida TSI for this lake is 56 (the lowest for all lakes included in this study), and it appears that water quality has remained stable over the past decade. The water is slightly tannic (color is 43 cpu), which lowers the Secchi disk values and may bias the TSI value. Table 5-15 shows the water quality data for Lake Gibson.

Lake Gibson displayed noticeable seasonality in its phytoplankton gross primary productivity. As has been reported by other investigators elsewhere in Florida (Nordlie 1976; McDiffet 1980; Beaver and Crisman 1989), maximum algal productivity occurred during summer. The

Table 5-15. Summary of water quality data for L. Gibson. Numbers shown are mean values with ranges in parentheses.

Parameter	Unit	Canfield (1981)	Several Agencies * (1989/91)	This study (1992/93)
Temperature	°C	N/A	25.03 (16.4-30.7)	24.43 (17.6-29.7)
D.O.	mg/L	N/A	7.09 (4.02-9.8)	7.69 (6.57-9.0)
pH		7.2 (6.8-7.6)	7.36 (5.9-10.84)	7.15 (6.4-8.12)
Alkalinity	mg/L $CaCO_3$	17 (9-21)	19.01 (4-44)	27.33 (22-32)
Conduct.	μmhos/cm	138 (120-160)	162.78 (109-205)	172.66 (148-200)
Nitrogen	mg/L	N/A	1.206 (0.1-3.71)	1.183 (0.88-1.69)
NH_3-N	mg/L	N/A	0.076 (0.001-0.3)	0.057 (0.016-0.09)
TKN	mg/L	N/A	1.087 (0.56-3.43)	0.926 (0.35-1.29)
Phosphorus	mg/L	0.650 (0.450-1.0)	0.259 (0.063-0.35)	0.183 (0.16-0.206)
Chl. a	mg/m^3	15.4 (3.2-28.9)	15.37 (2.2-93.1)	8.83 (5.8-12.8)
Color	Pt units	N/A	66.6 (40-133)	60.83 (37-79)
Secchi	m	2.2 (1.4-3.1)	0.87 (0.5-1.4)	1.2 (1.1-1.3)
Bacteria	x 10^6	N/A	287.97 (1-3600)	23.4 (1-60)
TSI		N/A	58.5 (46.7-79.4)	55.18 (53.06-57.3)

(*) AGENCY CODES:
USGS; EPA; PCWR; CL; UFC; FGFWFC; CF208; and SWFWMD.

highest values recorded were at the surface; the annual mean was 214.18 $mgC.m^{-3}.h$ ± 229.1, with the minimum value of 15.62 $mgC.m^{-3}.h$ during winter and 468.75 $mgC.m^{-3}.h$ during summer (Table 5-3). Values for net photosynthesis are shown in table 5-4, community respiration in table 5-5, and average annual productivity in table 5-6.

The phytoplankton community in Lake Gibson was dominated by the blue green algal genus Spirulina. Throughout the year, other blue green genera were represented including Merismopedia sp. and Lyngbya sp., as well as the dinoflagellate Peridinium sp.

The zooplankton assemblage was comprised mainly of rotifers. Brachionus sp., Keratella sp., and Collurella sp., were the most numerically abundant rotifers. The only copepod was Diaptomus sp.

No recent fish population survey was available for this lake. Ten species of fish were reported from the last assessment done in 1979. The dominant species at that time was channel catfish, followed by blue tilapia, with 63.9 % and 26.0 % of total fish weight, respectively.

ANOVAs were performed using the General Linear Model (GLM) (SAS, 1989) in order to identify significant seasonal limnological factors ($p < 0.05$). The following parameters were analyzed: Secchi, temperature, pH, conductivity, dissolved oxygen, total suspended solids, turbidity,

chlorophyll *a*, color, alkalinity, NH_3, TKN, total nitrogen, total phosphorus, orthophosphate, and total bacteria. Temperature, conductivity, dissolved oxygen and chlorophyll *a*, exhibited significant seasonal variation at the 95% confidence level (Table 5-16). Other parameters displayed either significant interannual variations or no variation at all. When year and season were combined, Secchi, conductivity and chlorophyll *a* displayed significant variability.

Lake Bonny

Lake Bonny is located in Polk County, Florida (Appendix A). The lake lies in the Bartow Embayment division of the Central Lake District (Canfield, 1981). Lake Bonny is an urban lake with a surface area of 144 ha, a shoreline length of 8.2 km and a mean depth of 2.0 m (Table 5-17).

Table 5-17. Morphometry of Lake Bonny(1)

Surface Area	144 ha
Maximum Depth	2.5 m
Mean Depth	2.0 m
Development of Shoreline	50 %
Drainage Basin Area	729 ha
Shoreline Length	8.2 km
Macrophyte Cover	10 %

(1) Data from PCWR (1992).

There are no known outflows from Lake Bonny. Land-use in the basin is a combination of commercial and moderate

Table 5-16. Correlation analyses of water quality data for Lake Gibson using the General Linear Model procedure (GLM). Data were combined by year, season and year*season.

PARAMETER	YEAR		SEASON		YEAR*SEASON	
	SIGNIF	N/SIGNF	SIGNIF	N/SIGNF	SIGNIF	N/SIGNF
Secchi	X			X	X	
Temperature		X	X			X
pH		X		X		X
Conductivity		X	X		X	
D.O.		X	X			X
TSS		X		X		X
Turbidity		X		X		X
Chlorophyll *a*	X		X		X	
Color		X		X		X
Alkalinity	X			X		X
NH_3		X		X		X
TKN		X		X		X
TN		X		X		X
TP		X		X		X
ORT		X		X		X
BACTERIA	X			X		X

density residential development. The most commonly encountered plants are Typha sp., Ludwigia repens, and Pontederia cordata (Canfield and Hoyer, 1992). Macrophyte chemical control measures are employed for water hyacinth, water lettuce, and hydrilla.

Lake Bonny was hypereutrophic during this study according to Forsberg and Ryding (1980). Historical data indicate that water quality has been improving since approximately 1986, a time when the lake almost dried completely. Table 5-18 shows water quality data for Lake Bonny.

Lake Bonny did not display any seasonality for gross primary productivity. The only noticeable trend was that phytoplankton productivity was always greater at the surface, with mean annual value of 448.04 $mgC.m^{-3}.h \pm 73.2$; the minimum was 345.12 $mgC.m^{-3}.h$. at April, and the maximum was 515.6 $mgC.m^{-3}.h$ at July (Table 5-3). Net photosynthesis is reported in table 5-4, community respiration in table 5-5, and average annual productivity in table 5-6.

Species of blue green algae including Anabaena sp., Spirulina sp., Merismopedia sp., and Lyngbya sp. dominated the phytoplankton assemblage of Lake Bonny. Green algae (Closterium sp., Cosmarium sp., and Scenedesmus sp.) were present, as well as the dinoflagellate Peridinium sp., and the diatom Stauroneis sp.

Table 5-18. Summary of water quality data for Lake Bonny. Numbers shown are mean values with ranges in parentheses.

Parameter	Unit	Several Agencies* (1989/91)	This study (1992/93)
Temperature	°C	24.87 (16.08-30.55)	24.26 (17.48-29.58)
D.O.	mg/L	7.24 (3.85-11.34)	7.47 (6.24-8.63)
pH		8.14 (5.9-9.79)	7.26 (6.28-8.34)
Alkalinity	mg/L $CaCO_3$	57.88 (48-84)	47.5 (37-56)
Conduct.	μmhos/cm	219.7 (170-257)	194.5 (158-219)
Nitrogen	mg/L	2.94 (0.74-7.2)	2.48 (1.5-4.06)
NH_3-N	mg/L	0.066 (0.001-0.20)	0.079 (0.031-0.13)
TKN	mg/L	2.83 (0.73-6.2)	2.29 (1.1-3.75)
Phosphorus	mg/L	0.14 (0.019-0.36)	0.11 (0.073-0.169)
Chl. a	mg/m^3	62.9 (18.1-182.5)	44.35 (23.3-94.7)
Color	Pt units	37.5 (13-100)	53.66 (39-67)
Secchi	m	0.5 (0.2-1.1)	0.56 (0.3-0.75)
Bacteria	x 10^6	1035.9 (1-13400)	165.83 (10-540)
TSI		76.72 (61.84-96.61)	73.46 (68.06-86.28)

(*) AGENCY CODES:
USGS; EPA; PCWR; CL; UFC; FGFWFC; CF208; and SWFWMD.

The Lake Bonny zooplankton community was dominated by rotifers and copepods with 274,000 and 85,200 individuals.m^{-3}, respectively (Canfield and Hoyer, 1992). The dominant rotifers were Brachionus sp. and Keratella sp. Diaptomus was the most abundant copepod.

Sixteen species of fish were collected in Lake Bonny (Canfield and Hoyer, 1992). Gizzard shad was the most abundant fish collected in open-water using experimental gillnets, followed by Florida gar, with 17.0 and 11.3 fish.net^{-1}.24hr, respectively (Canfield and Hoyer, 1992).

ANOVAs were performed using the General Linear Model (GLM) (SAS, 1989) in order to identify significant seasonal limnological factors ($p < 0.05$). The following parameters were analyzed: Secchi, temperature, pH, conductivity, dissolved oxygen, total suspended solids, turbidity, chlorophyll *a*, color, alkalinity, NH_3, TKN, total nitrogen, total phosphorus, orthophosphate, and total bacteria. Temperature, conductivity, dissolved oxygen, turbidity, chlorophyll *a*, color, TKN, total nitrogen and total phosphorus exhibited significant seasonal variation at the 95% confidence level (Table 5-19). Interannual variation or no variation was reported for the other parameters. When year and season were combined, Secchi, pH, conductivity, TSS, turbidity, chlorophyll *a*, color, and total phosphorus displayed significant variability.

Table 5-19. Correlation analyses of water quality data for Lake Bonny using the General Linear Model procedure (GLM). Data were combined by year, season and year*season.

PARAMETER	YEAR		SEASON		YEAR*SEASON	
	SIGNIF	N/SIGNF	SIGNIF	N/SIGNF	SIGNIF	N/SIGNF
Secchi	X			X	X	
Temperature		X	X			X
pH	X			X	X	
Conductivity		X	X		X	
D.O.		X	X			X
TSS	X			X	X	
Turbidity	X		X		X	
Chlorophyll *a*	X		X		X	
Color	X		X		X	
Alkalinity		X		X		X
NH_3		X		X		X
TKN	X		X			X
TN	X		X			X
TP	X		X		X	
ORT		X		X		X
BACTERIA	X			X		X

Lake Hunter

Lake Hunter is located in Polk County, Florida (Appendix A). The lake lies in the Bartow Embayment division of the Central Lakes District (Canfield, 1981). Lake Hunter is a small urban lake located within the city limits of Lakeland, with a surface area of 41 ha, a shoreline length of 2.6 km, and a mean depth of 1.5 m (Table 5-20).

Table 5-20. Morphometry of Lake Hunter(1)

Surface Area	41 ha
Maximum Depth	2.7 m
Mean Depth	1.5 m
Development of Shoreline	80 %
Drainage Basin Area	208 ha
Shoreline Length	2.6 km
Macrophyte Cover	0.2 %

(1) Data from PCWR (1992).

Development in the watershed has caused Lake Hunter to be used as a stormwater retention area. Twelve stormwater discharge inlets enter the lake, six of which can be considered as a major source of pollution. The lake watershed is almost entirely developed and consists of commercial and moderate to large residential developments. Most typical of urban systems, Lake Hunter experiences excessive sedimentation, poor water quality, and rapid growth of undesirable aquatic vegetation. The lake's macrophyte dominant vegetation (defined as greater than or equal to 5% macrophyte cover) includes elephant ear, water

primrose, and pennywort. There is no aquatic weed control program for Lake Hunter.

According to Forsberg and Ryding (1980), Lake Hunter was hypereutrophic during this study. In 1983, Lake Hunter was completely drained, and water quality has continued to decline since then. Table 5-21 show water quality data for Lake Hunter.

Lake Hunter does not have much seasonality in its gross primary productivity (Table 5-3). The most productive layer is at 50 cm water depth. The mean for the 50 cm depth was 306.71 $mgC.m^{-3}.h$ ± 175.6 throughout the year, with a minimum value of 115.62 $mgC.m^{-3}.h$ in December and a maximum of 484.37 $mgC.m^{-3}.h$ in April. Values for net photosynthesis are shown in table 5-4, community respiration in table 5-5, and average annual productivity in table 5-6.

The phytoplankton community in Lake Hunter had Cyanophytes as dominants with Chlorophytes being co-dominant. Species of blue-green algae such as Lyngbya sp., Spirulina sp., Microcystis sp.,Merismopedia sp., and Anabaena sp. were present as well as green algae such as Closterium sp., and Scenedesmus sp.; the dinoflagellate Peridinium sp. were abundant. No diatoms were reported for this lake.

Rotifers and copepods comprised the dominant zooplankton community for Lake Hunter. Dominant rotifers were Keratella sp., Brachionus sp., and Monostyla sp.

Table 5-21. Summary of water quality data for Lake Hunter. Numbers shown are mean values with ranges in parentheses.

Parameter	Unit	Several Agencies* (1989/91)	This study (1992/93)
Temperature	°C	25.50 (16.50-31.51)	24.98 (17.87-30.09)
D.O.	mg/L	9.60 (4.72-13.90)	10.46 (9.88-11.8)
pH		9.48 (7.88-11.35)	8.74 (8.33-9.45)
Alkalinity	mg/L $CaCO_3$	65.25 (46-99.9)	66.6 (62-71)
Conduct.	μmhos/cm	199.6 (139-253)	202.8 (160-276)
Nitrogen	mg/L	2.24 (0.51-4.92)	2.50 (1.58-3.56)
NH_3-N	mg/L	0.067 (0.01-0.292)	0.042 (0.032-0.056)
TKN	mg/L	2.14 (0.51-4.81)	2.21 (1.42-3.05)
Phosphorus	mg/L	0.150 (0.019-0.26)	0.18 (0.158-0.239)
Chl. a	mg/m^3	70.69 (25.6-152.8)	69.75 (43.3-112.3)
Color	Pt units	28.6 (15-55)	38.66 (25-60)
Secchi	m	0.38 (0.2-0.6)	0.366 (0.2-0.6)
Bacteria	x 10^6	1236.3 (1-20000)	342.5 (140-600)
TSI		79.18 (66.36-94.52)	81.58 (71.94-88.88)

(*) AGENCY CODES:
USGS; EPA; PCWR; CL; UFC; FGFWFC; CF208; and SWFWMD.

Diaptomus was the only copepod identified. Data from Canfield and Hoyer (1992) show a rotifer density of 31,200 individuals.m^{-3}, and a copepod density of 14,000 individuals.m^{-3}.

Eighteen species of fish were collected in Lake Hunter (Canfield and Hoyer, 1992). Historically, Lake Hunter was a hypereutrophic lake with an overabundance of forage fish, principally gizzard shad and threadfin shad (Huish, 1955; Ware and Horel, 1971). After the lake restoration done in 1983 and 1984, 10,000 largemouth bass and 1,000 sunshine bass were stocked to establish a predator population (Moxley et al. 1984). After 8 years, the most abundant open-water species collected in experimental gillnets were gizzard shad and sunshine bass, with 163 and 8 fish.net^{-1}.24hr, respectively (Canfield and Hoyer, 1992).

ANOVAs were performed using the General Linear Model (GLM) (SAS, 1989) in order to identify significant seasonal limnological factors ($p < 0.05$). The following parameters were analyzed: Secchi, temperature, pH, conductivity, dissolved oxygen, total suspended solids, turbidity, chlorophyll *a*, color, alkalinity, NH_3, TKN, total nitrogen, total phosphorus, orthophosphate, and total bacteria.

The data were collected between 1989 and 1993, and grouped into four seasons: winter (December to February); spring (March to May); summer (June to August); and fall (September to November). The results in the Table 5-22 are

Table 5-22. Correlation analyses of water quality data for Lake Hunter using the General Linear Model procedure (GLM). Data were combined by year, season and year*season.

PARAMETER	YEAR		SEASON		YEAR*SEASON	
	SIGNIF	N/SIGNF	SIGNIF	N/SIGNF	SIGNIF	N/SIGNF
Secchi	X			X	X	
Temperature		X	X			X
pH	X			X	X	
Conductivity	X		X			X
D.O.		X	X		X	
TSS	X		X			X
Turbidity	X			X	X	
Chlorophyll *a*		X	X			X
Color	X			X		X
Alkalinity		X		X		X
NH_3		X		X		X
TKN	X			X		X
TN	X			X		X
TP	X			X		X
ORT		X	X			X
BACTERIA		X		X		X

reported as significant at $p < 0.05$. Temperature, conductivity, dissolved oxygen, TSS, chlorophyll *a*, and orthophosphate exhibited significant seasonal variation. Other parameters displayed either interannual or no variation. When year and season were combined, Secchi, pH, dissolved oxygen and turbidity displayed significant variability.

Summary

For the scope of this study, the lakes were divided into two groups: [1] lakes considered as having blue tilapia as the major component of its filter-feeder fish population (Lakes Parker, Hollingsworth, and Hancock), and [2] lakes were gizzard shad is the dominant rough fish (Lakes Gibson, Bonny and Hunter). Evaluating the limnology of the study lakes, there seems to exist differences among them. Figure 5-1 has information on gross primary productivity ($mgC.m^{-3}.h$), community respiration ($mgC.m^{-3}.h$), phytoplankton (NU/ml) and zooplankton ($individuals.m^{-3}$) for all lakes.

Lakes Parker and Hollingsworth have a significantly larger (ANOVA, $p < 0.05$) zooplankton population than Lakes Bonny and Hunter, which could be indicative of the different proportionality of filter feeder fish in the population. Lakes where blue tilapia have proportionally high population tend to have higher number of zooplankton, especially Lake

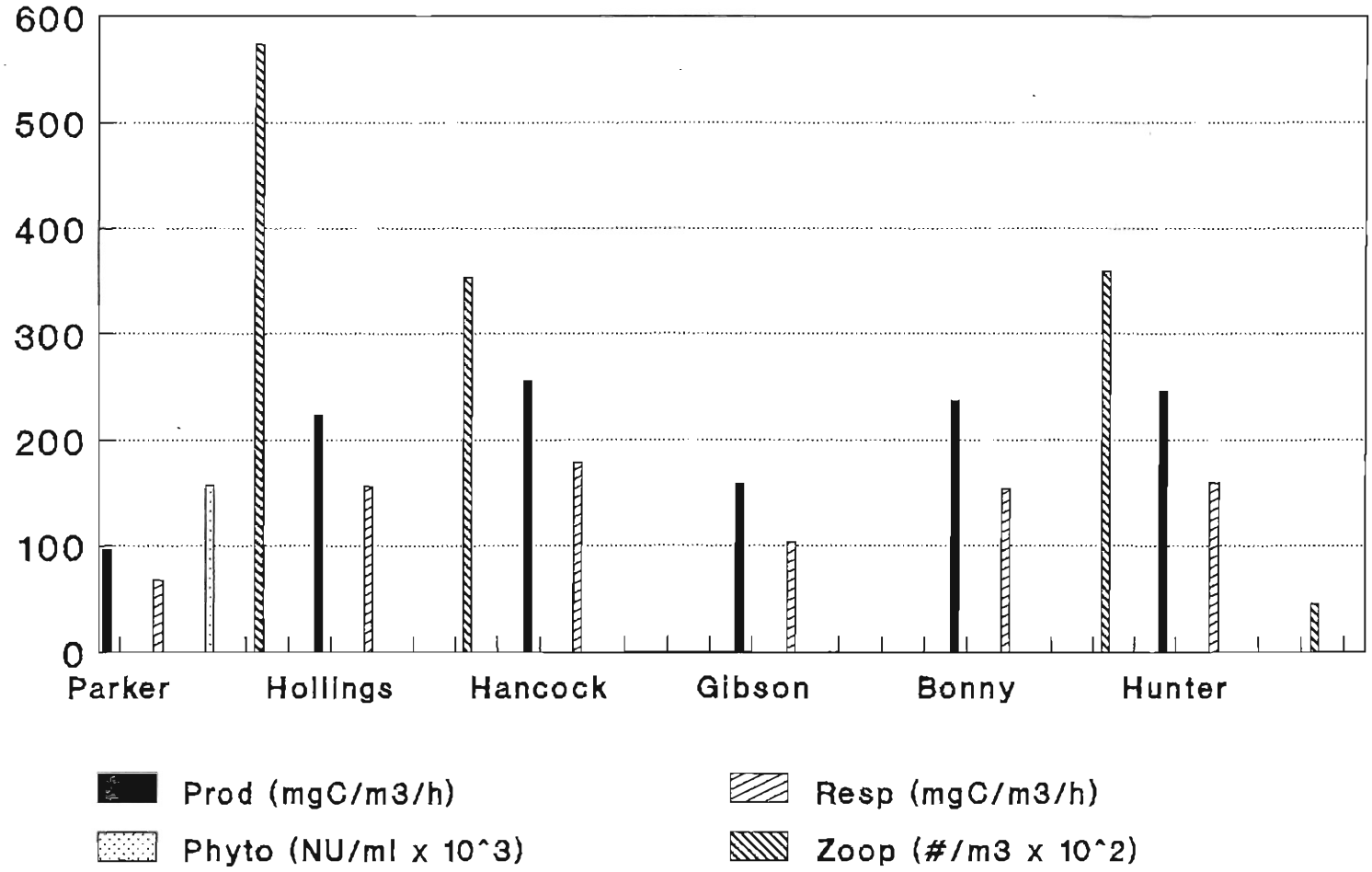

Figure 5-1. Phytoplankton and zooplankton population for all study lakes. Values for primary productivity and community respiration are shown.

Parker, where the zooplankton population was two times larger than the other lakes, data that are in agreement with Crisman and Beaver (1988).

Lakes considered as having more blue tilapia also have proportionally lower gross primary productivity (Figure 5-1) than the group where gizzard shad is considered dominant, which is consistent with the results of Crisman and Beaver (1988).

Regarding to the fish population, the two extremes were in the group where blue tilapia is prevalent. Lake Parker has the smallest fish population of the six studied lakes (total fish biomass = 71.1 $kg.ha^{-1}$) (Canfield and Hoyer, 1991), and Lake Hollingsworth the largest fish population (total fish biomass = 1050 $kg.ha^{-1}$) (Canfield and Hoyer, 1992). The small fish population in Lake Parker could also help to explain the presence of the large zooplankton population in that lake. There were no noticeable differences in the size of the fish populations for Lakes Bonny and Hunter, both belonging to the group where gizzard shad was dominant.

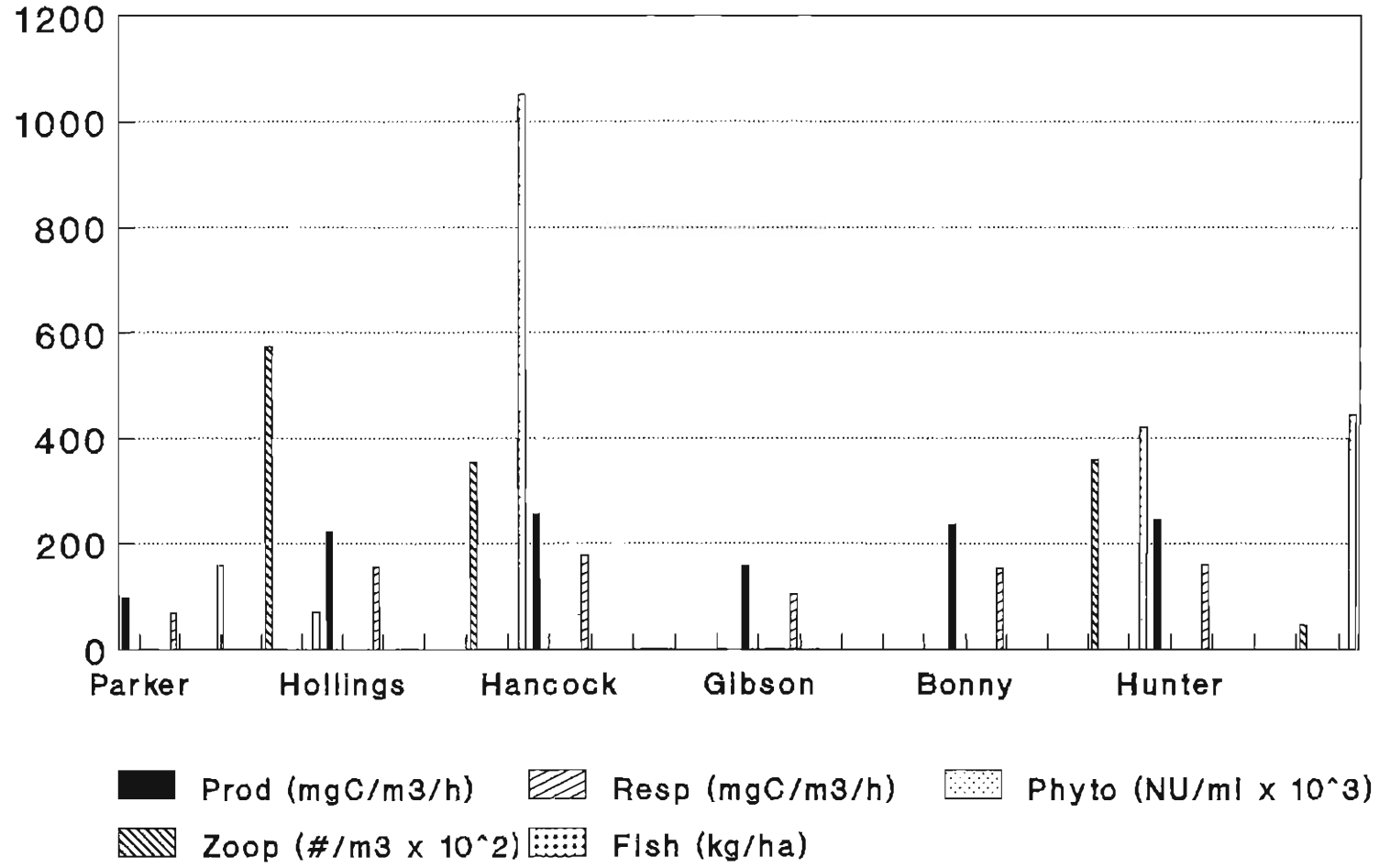

Figure 5-2. Fish, phytoplankton and zooplankton population for all study lakes. Values for primary productivity and community respiration are shown.

CHAPTER 6
SUMMARY

Fish community impact on lake ecosystems was addressed from the point of view of the effects of replacement of a native filter feeder, gizzard shad, by the exotic blue tilapia which has been present in central Florida eutrophic lakes for the last thirty years. Evaluation of the impact of gizzard shad and blue tilapia was through the characterization and estimation of the fecal material produced by each respective fish species. Analyses were performed in order to determine how the different kinds of feces could affect bioavailability of nutrients in the systems.

A mesocosm experiment was performed to evaluate past and present lake conditions. Paleolimnological data were coupled with ambient water quality data in order to create the necessary framework to permit further investigation of system alterations due to modifications in the biological community.

Several issues addressed in this study provided information that led to a mosaic of interpretations. Historical limnological data were analyzed for the six lakes studied. Physicochemical and limnological data were

collected for more than twelve months in all the lakes. Phytoplankton primary productivity and community respiration were measured. Phytoplankton, zooplankton and fish communities were identified. Sediment cores were taken from two lakes, and sediment core information was gathered for two of the others. Finally, the fecal composition of the two major filter feeder fish in the systems was determined and relative impact estimated.

All lakes except one were classified as hypereutrophic, according to the classification of Forsberg and Ryding (1980). Lake Gibson was classified as eutrophic. For analytical purposes, I grouped the lakes into two categories: lakes where the prevalent filter feeder fish (defined as >50% of the filter-feeder fish population) was gizzard shad and lakes where blue tilapia was the prevalent filter feeder fish. Lakes Gibson, Bonny, and Hunter were dominated by shad during this study, while Lakes Parker, Hollingsworth and Hancock were tilapia dominated systems.

Fish feces experiment

There were significant differences in the amount of feces hand-stripped from both species of fishes. Blue tilapia releases its fecal material in pellet form enveloped in mucilage, whereas gizzard shad releases its feces as a flocculent material. Blue tilapia produced the greatest amount of feces when compared with gizzard shad. The bulk

density of blue tilapia feces was two times the bulk density for gizzard shad feces. On average, shad feces was composed of 37% more organic matter, 50% more protein, and had a 60% higher caloric content than did blue tilapia feces. Furthermore, blue tilapia feces consisted of 50% more dry matter than gizzard shad feces.

I carried out two bioassays to evaluate fecal nutrient composition and algal content. I measured the change in fluorescence for each treatment group. The three treatment groups were: [1] feces plus Microcystis aeruginosa; [2] feces plus Selenastrum capricornutum; and [3] control (no feces). Each treatment group was inoculated with algae in both a medium deprived of nitrogen and a medium deprived of phosphorus.

Gizzard shad fecal samples always yielded greater fluorescence values regardless of the algal strain or chemical composition of the growth medium (-N or -P) used. In treatment group feces plus Selenastrum capricornutum, a significant increase in fluorescence occurred in the first day of experiment; whereas in treatment group feces plus Microcystis aeruginosa, a significant increase occurred on the fourth day of the bioassay (Figures 3-3 and 3-4).

Results were different in the flasks inoculated with blue tilapia fecal material. Despite high initial fluorescence values, in the presence of treatment group feces plus Microcystis aeruginosa, fluorescence values

remained constant and a decrease was noticed after the fourth day of experimentation. When the treatment group feces plus Selenastrum capricornutum was evaluated, high initial fluorescence values decreased after the first day of the bioassay, reaching minimum values on the fourth day of the experiment. (Figures 3-3 and 3-4). Control flasks in the medium deprived of phosphorus and in the medium deprived of nitrogen exhibited algal fluorescence readings of zero or near zero.

Chlorophyll *a* readings for blue tilapia feces were very high in the beginning of the experiment (ten-fold greater than the values measured for gizzard shad feces). The results showed that gizzard shad fecal material started the bioassay at low chlorophyll *a* value in the flasks containing blue green algae both in the nitrogen and in the phosphorus deprived medium. At the end of the experiment, chlorophyll *a* values increased ten-fold in the nitrogen deprived medium and 13.5 times in the phosphorus deprived medium. Blue tilapia feces displayed, on the other hand, under the same conditions, increased chlorophyll *a* only at the end of the experiment and only 5 to 6 times the initial value, an increase which is only half of that produced by gizzard shad feces.

Phytoplankton biomass in gizzard shad and blue tilapia feces was indicative of the different impact these fishes can have in aquatic systems. While phytoplankton biomass was

suppressed (zero growth) in 73% of all samples containing blue tilapia feces, all the samples containing gizzard shad feces displayed a ten-fold increase.

Paleolimnological analysis

Analyses of sediment cores from Lakes Bonny and Gibson, both defined in this study as having a prevalent filter-feeder population of gizzard shad, were done to assess the distribution of organic deposits throughout the lake basin and for sediment age determination. Sediment characterization was based on one core collected from the deepest part of each lake.

Lake Bonny showed high organic matter content in the superficial layers (58% of organic matter). TP and TN were constant throughout the whole core. Bulk density and dry matter began to decline after 1950, which coincides with urban development of the watershed. Total phosphorus reconstructions near the base of the core suggest that the lake has always been mesotrophic to eutrophic or hypereutrophic. However, the lake has experienced periods of much higher nutrient enrichment, and values for total phosphorus in excess of 5.0 $mg.g^{-1}$ were common for layers that post-dated the ^{137}Cs peak after 1960. Detectable ^{137}Cs, injected into the atmosphere as a consequence of nuclear weapons testing in the early 1950s, matched well with the determined ^{210}Pb chronology.

Lake Gibson had less organic matter in its superficial layers than did Lake Bonny. However, values of up to 40% organic content were registered throughout the core, which coincides with the average organic matter content of dry bulk sediment in Florida lakes of 39.7% (Brenner and Binford, 1988). Total nitrogen and total phosphorus declined at depths > 24 cm in the core. Below 48 cm in the core, total phosphorus was undetectable.

Total phosphorus reconstructions at the base of the core suggest that the lake was oligotrophic for at least the last century. Nevertheless, values for TP exceeding 3.0 $mg.g^{-1}$ were registered for the sediment corresponding to the last 25 years, reflecting changes in the watershed use. Using ^{210}Pb determinations, it appears that the lake had much less organic sediment before the 1950s. Detectable ^{137}Cs coincided with the determined ^{210}Pb chronology.

Lakes Parker and Hollingsworth, both defined in this study as having a blue tilapia dominant fish population, were the subject of a sediment survey done by Schelske et al. (1992). They stated that Lake Parker has always been mesotrophic to eutrophic. Analyses of diatom assemblages indicated the presence of alkaline conditions. Determination of total phosphorus in the topmost 15 cm of sediment suggested eutrophic conditions. However, these analyses further indicated a progressive decline in water column total phosphorus that may signal some reversal of cultural

enrichment (Schelske et al., 1992). Diatom reconstructions of historical water column total phosphorus concentrations indicate that Lake Hollingsworth has been eutrophic to hypereutrophic for the past 100 years (Schelske et al., 1992). During the last ten years, however, total phosphorus inferences (cf. Schelske et al., 1992) suggest that the rate of eutrophication remained relatively constant.

Since both of these lakes were classified as having a blue tilapia dominant fish population in this study, it is possible that some of the trophic reversal can be credited to a filter feeder fish replacement. Additional support for this hypothesis can be drawn indirectly from the absence of an improvement trend in the sediment nutrient concentration in Lakes Bonny and Gibson, both gizzard shad dominant lakes. However, additional study is required in order to confirm this hypothesis.

Lakes assessment

The lakes were analyzed for the years 1989 to 1993 and separated by season within each of these years. The seasons were defined as winter (December, January and February), spring (March, April and May), summer (June, July and August), and fall (September, October and November). ANOVAs were performed using General Linear Model (GLM) (SAS, 1989) in order to identify significant seasonal limnological factors ($p < 0.05$).

Cluster analysis (Pearson Correlation Coefficient) (SAS, 1989) divided the six lakes into 4 aggregations. Lakes Hunter and Bonny, both with a prevalent population of gizzard shad, were the most statistically similar lakes in this study. Lake Parker, a blue tilapia dominant lake, displayed limnological characteristics that differed significantly from the mean values reported for the other study lakes.

Cluster analyses of lakes by fish species yielded similar relationships. Lake Parker again did not show any correlation with the other study lakes. Seasonally, summer and fall appeared to be the most stable seasons of the year for the study lakes, and winter is the one which exhibited the greatest statistical variability. This observation is certainly a consequence of temperature, with minimum annual temperatures occurring during the winter months.

Primary productivity determinations showed high gross photosynthesis for most of the lakes, with a mean of 334.22 $mgC.m^{-3}.h \pm 106.35$. The only deviation occurred in Lake Parker, where a mean value of 66.41 $mgC.m^{-3}.h$ was recorded.

Conclusion

In view of the evidence presented in this study, coupled with the findings of mesocosm experiments performed by Crisman and Kennedy (1982), Drenner et al. (1986), and Crisman and Beaver (1988), it seems possible that

biomanipulation techniques employing fish as a major element could lead to a significant improvement in water clarity in subtropical and tropical lakes. This study certainly provides evidence that gizzard shad and blue tilapia feces have a different composition and may stimulate different plankton community dynamics.

The results of this investigation suggest intriguing possibilities for the potential role of filter feeding fish in the control of eutrophication in Florida lacustrine systems. Gizzard shad feces clearly stimulated algal growth, while blue tilapia feces appeared to suppress algal growth. However, additional research is necessary in the use of filter feeding fish in biomanipulation schemes.

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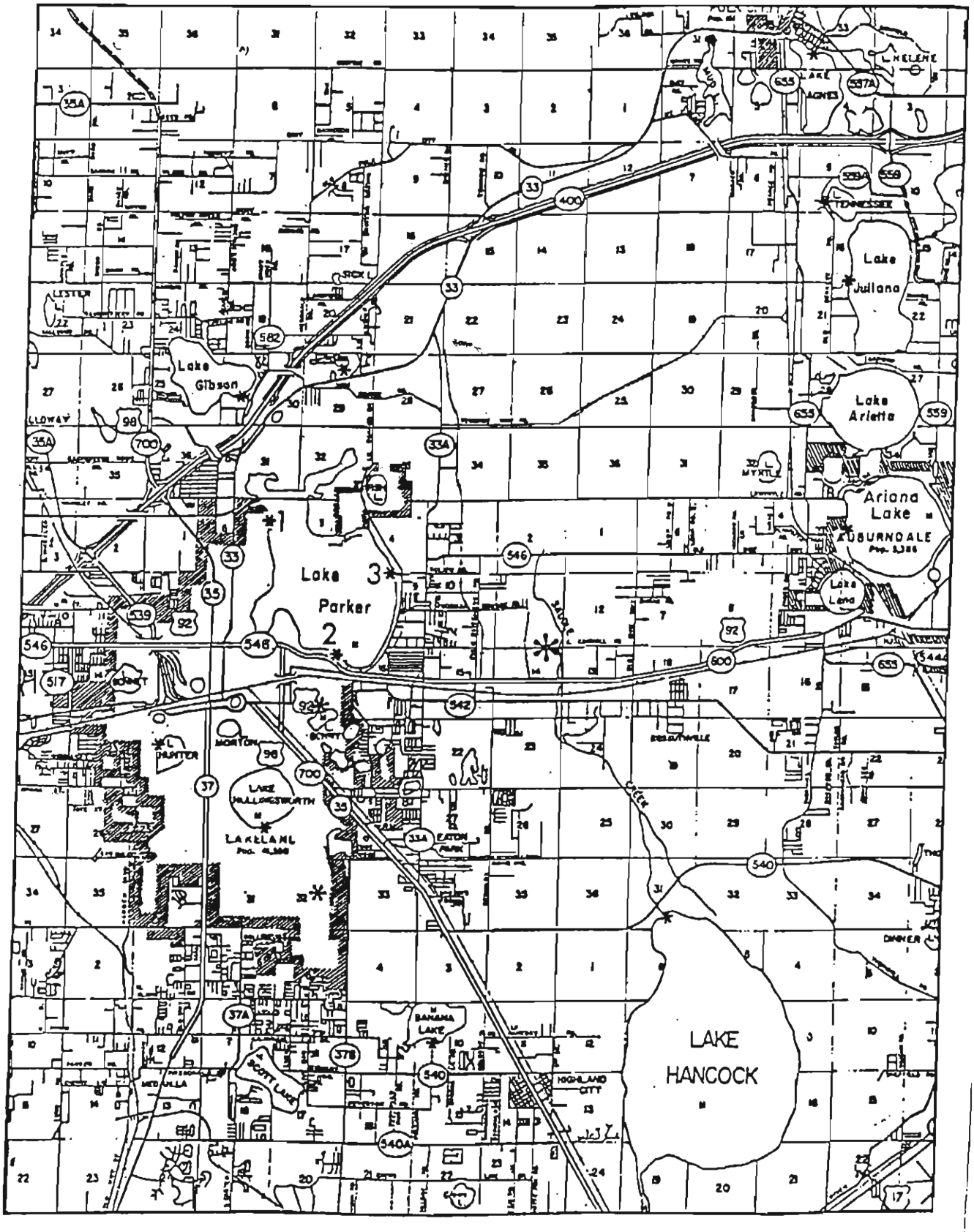

Map 1 * - Shows approximately where ramps are

_____ - Shows where canals are

APPENDIX B
Lake Bonny sedimentation rate ($g.cm^{-2}.yr$) by year.

LAKE BONNY

Core Sectio (cm)		Bulk Density (g/cm3)	Unsupp. Pb-210 (pCi/g)	Unsupp. Pb-210 (pCi/cm2	Cum.Res. Uns.Pb-21 (pCi/cm2)	Age (yrs BP) (1993)	Sed. Rate (g/cm2.yr)	Cs-137 (pCi/g)
0	4	0.029	18.72	2.17	31.84	0.00	0.05	1.5
4	7	0.050	12.30	1.84	29.67	2.27	0.08	3
7	9	0.061	12.50	1.53	27.83	4.32	0.07	4.13
9	11	0.066	11.22	1.48	26.31	6.13	0.07	4.01
11	13	0.074	8.46	1.25	24.83	7.99	0.09	3.65
13	15	0.078	8.14	1.27	23.57	9.65	0.09	3.20
15	17	0.079	5.27	0.83	22.30	11.43	0.13	4.02
17	19	0.090	6.99	1.26	21.47	12.65	0.10	4.56
19	22	0.092	8.92	2.46	20.21	14.59	0.07	4.49
22	26	0.092	6.51	2.39	17.75	18.76	0.08	3.28
26	30	0.090	7.00	2.52	15.36	23.40	0.07	2.67
30	34	0.089	6.00	2.14	12.84	29.16	0.07	1.07
34	38	0.091	3.80	1.38	10.70	35.01	0.09	0.85
38	42	0.102	3.43	1.40	9.32	39.44	0.08	0.92
42	46	0.164	3.48	2.28	7.92	44.67	0.07	0.87
46	50	0.174	3.17	2.21	5.64	55.56	0.06	0.78
50	54	0.142	1.91	1.08	3.43	71.52	0.06	0.91
54	58	0.137	1.94	1.06	2.35	83.64	0.04	0.59
58	64	0.124	0.00	-0.10	1.29	102.86	ERR	0.14
64	72	0.125	0.00	0.15	1.39	100.56	ERR	0.33
72	84	0.126	0.00	1.24	1.24	104.23	ERR	0.37

APPENDIX C
Lake Gibson sedimentation rate ($g.cm^{-2}.yr$) by year.

LAKE GIBSON

Core Sectio (cm)		Bulk Density (g/cm3)	Unsupp. Pb-210 (pCi/g)	Unsupp. Pb-210 (pCi/cm2	Cum.Res. Uns.Pb-21 (pCi/cm2)	Age (yrs BP) (1993)	Sed. Rate (g/cm2.yr)	Cs-137 (pCi/g)
0	2	0.113	11.86	2.68	28.29	0.00	0.07	4.41
2	4	0.167	9.48	3.16	25.61	3.20	0.08	5.02
4	6	0.221	9.31	4.11	22.45	7.43	0.08	4.18
6	8	0.233	6.76	3.15	18.34	13.92	0.08	3.63
8	10	0.271	5.72	3.10	15.19	19.97	0.08	2.96
10	12	0.284	6.07	3.45	12.09	27.30	0.06	2.96
12	14	0.229	2.33	1.07	8.64	38.09	0.12	1.32
14	16	0.178	2.63	0.94	7.57	42.32	0.09	1.28
16	18	0.168	2.61	0.88	6.64	46.56	0.08	0.84
18	20	0.160	3.34	1.07	5.76	51.11	0.05	0.75
20	24	0.161	2.29	1.47	4.69	57.70	0.06	0.71
24	28	0.185	1.11	0.82	3.22	69.78	0.09	0.75
28	32	0.208	1.42	1.18	2.40	79.22	0.05	0.57
32	36	0.183	2.00	1.46	1.22	100.95	0.02	0.30
36	40	0.161	1.42	0.91	-0.24	ERR	-0.01	0.25
40	44	0.177	0.12	0.08	-1.15	ERR	-0.30	0.37
44	48	0.269	1.41	1.52	-1.23	ERR	-0.03	0.19
48	52	0.517	0.39	0.81	-2.75	ERR	-0.22	0.03
52	56	1.167	0.00	-0.93	-3.56	ERR	ERR	0.04
56	60	1.644	0.00	-2.63	-2.63	ERR	ERR	-0.05

APPENDIX D
Physical and chemical properties of Lake Parker sediment core (cf. Schelske et al., 1992)

Interval (cm)	Mid-depth (cm)	Rho ($g.dry.cm^{-3}$)	Organic Matter (%)	C_{tot} (%)	N_{tot} ($mg.g^{-1}$)	P_{tot} ($mg.g^{-1}$)
0-2	1	0.01820	71.7	34.9	63.7	4.9
2-4	3	0.02575	71.4	35.3	N/A	N/A
4-6	5	0.02702	70.1	34.3	71.8	5.5
6-8	7	0.02895	63.4	31.9	N/A	N/A
8-10	9	0.03554	59.1	28.6	23.3	4.5
10-12	11	0.03667	54.0	26.3	N/A	N/A
12-14	13	0.03652	52.1	27.2	23.5	4.5
14-16	15	0.04000	52.0	26.6	N/A	N/A
16-18	17	0.03919	52.3	27.7	25.4	4.9
18-20	19	0.04368	50.1	26.8	22.7	5.1
22-24	23	0.04846	48.6	24.9	24.8	5.5
26-28	27	0.04934	47.7	24.6	25.6	5.9
30-32	31	0.05047	48.8	26.3	22.7	6.7
34-36	35	0.05258	48.4	26.6	23.8	6.5
38-40	39	0.05625	41.5	20.7	22.3	6.0
42-44	43	0.05751	37.1	20.4	19.0	6.5
46-48	47	0.05741	37.4	21.7	19.6	6.8

APPENDIX E
Physical and chemical properties of Lake Parker sediment core (cf. Schelske et al., 1992) Cont.

Interval (cm)	Mid-depth (cm)	Rho ($g.dry.cm^{-3}$)	Organic Matter (%)	C_{tot} (%)	N_{tot} ($mg.g^{-1}$)	P_{tot} ($mg.g^{-1}$)
50-52	51	0.06073	40.4	23.4	18.8	5.4
54-56	55	0.05879	41.0	21.6	21.4	5.4
58-60	59	0.06078	49.1	28.8	26.8	5.7
60-62	63	0.06016	44.3	23.6	25.0	5.7
66-68	67	0.07233	49.6	28.3	25.6	5.5
70-72	71	0.07785	54.6	31.3	28.0	4.5
74-76	75	0.07166	54.3	30.9	29.2	4.1
78-80	79	0.06960	55.0	31.0	28.6	4.2
82-84	83	0.07834	56.6	32.4	28.3	4.1
86-88	87	0.07616	57.5	32.6	23.2	4.7
90-92	91	0.08900	55.8	31.3	25.1	3.4

APPENDIX F
Physical and chemical properties of Lake Hollingsworth sediment core (cf. Schelske et al., 1992).

Depth (cm)	Density ($g.dry.cm^{-3}.wet$)	Organic Matter (%LOI)	C_{tot} (%)	N_{tot} (%)	P_{tot} ($mg.g^{-1}$)
0-2	0.01772	54.3	27.6	3.04	6.99
2-4	0.02082	51.8	27.4	3.35	7.24
4-6	0.02487	54.5	27.4	2.89	6.42
6-8	0.02693	52.8	26.0	3.02	7.32
8-10	0.02846	52.8	26.7	N/A	N/A
10-12	0.03139	52.2	26.1	2.92	7.03
12-14	0.03583	48.7	25.5	2.76	6.47
14-16	0.03796	48.8	25.4	2.71	6.58
16-18	0.04688	40.7	20.8	2.36	5.43
18-20	0.06550	30.5	14.9	2.17	5.13
20-22	0.09093	24.2	13.9	1.34	5.18
22-24	0.08989	25.8	13.8	1.66	6.47
24-26	0.09053	28.9	15.0	1.67	8.87
26-28	0.08030	34.3	17.3	1.97	9.78
28-30	0.08537	30.6	16.3	1.53	9.64
30-32	0.10669	25.7	14.8	1.35	7.52
40-42	0.07557	43.4	24.3	2.43	6.22
50-52	0.07180	49.8	27.8	2.70	4.30
60-62	0.07493	55.5	34.3	3.04	3.57
70-72	0.07412	56.7	33.8	3.16	3.96
80-82	0.06584	62.0	35.3	3.53	3.20
90-92	0.07100	55.5	34.4	2.99	2.29
98-100	0.08207	48.4	28.4	2.81	2.48

APPENDIX G
Sediment core dating for Lake Hollingsworth
(cf. Schelske et al., 1992).

Depth (cm)	Total ^{210}Pb ($dpm.g^{-1}$)	^{214}Bi ($dpm.g^{-1}$)	Excess ^{210}Pb ($dpm.g^{-1}$)	^{137}Cs ($dpm.g^{-1}$)	Age (years)	Date (AC)	Sed.rate ($g.cm^{-2}.yr^{-1}$)
0-4	31.41	7.90	23.62	2.79	1.5	1990	0.051
4-8	32.98	8.25	24.84	3.06	3.8	1988	0.046
8-12	29.01	8.32	20.78	3.04	6.2	1986	0.051
12-16	28.63	7.46	21.26	3.28	9.4	1983	0.045
16-18	24.99	5.93	19.17	3.06	11.4	1981	0.046
18-20	18.24	6.54	11.77	2.84	13.3	1979	0.071
20-22	19.40	7.53	11.93	3.69	16.1	1976	0.065
22-24	20.12	9.35	10.82	2.89	18.8	1973	0.066
24-26	23.04	11.99	11.11	2.29	21.9	1970	0.059
26-28	26.71	13.99	12.80	3.17	25.4	1967	0.046
28-30	21.73	10.54	11.26	2.58	29.0	1963	0.047
30-32	18.20	9.14	9.11	1.57	33.2	1959	0.051
32-40	17.05	8.70	8.39	1.20	51.6	1940	0.040
40-42	15.90	8.26	7.67	0.82	56.7	1935	0.030
42-50	11.51	6.46	5.07	0.70	75.5	1916	0.031

APPENDIX H
Sediment core dating for Lake Hollingsworth
(cf. Schelske et al., 1992)

Depth (cm)	Total ^{210}Pb ($dpm.g^{-1}$)	^{214}Bi ($dpm.g^{-1}$)	Excess ^{210}Pb ($dpm.g^{-1}$)	^{137}Cs ($dpm.g^{-1}$)	Age (years)	Date (AC)	Sed. rate ($g.cm^{-2}.yr^{-1}$)
50-52	7.13	4.66	2.48	0.59	78.7	1913	0.045
52-60	5.83	3.68	2.16	0.47	93.7	1898	0.039
60-62	4.53	2.70	1.84	0.35	98.2	1894	0.034
62-70	5.22	2.85	2.38	0.33	144.8	1847	0.013
70-72	5.91	3.00	2.92	0.31	N/A	N/A	N/A
72-80	4.68	3.06	0.00	0.21	N/A	N/A	N/A
80-82	3.44	3.11	N/A	0.11	N/A	N/A	N/A
82-90	3.56	3.08	N/A	0.12	N/A	N/A	N/A
90-92	3.68	3.06	N/A	0.12	N/A	N/A	N/A
92-98	4.11	2.78	N/A	0.17	N/A	N/A	N/A

BIOGRAPHICAL SKETCH

Carlos A. Fernandes was born in Mossoró, State of Rio Grande do Norte, Brazil, on April 11, 1950. He received a Bachelor of Science degree in molecular biology from University of Brasília, Brasília, Brazil, in 1976. He began working as a professor at the University of Brasília, in 1977. He received his Master of Science degree in ecology, with area of specialization in limnology from University of Brasília, in 1979. Later that year he started working as a biologist for the Water and Wastewater Company of Brasília (CAESB). From 1984 to 1989 he worked as a director for Pollution Control for the Secretariat of Environment, Science and Technology (SEMATEC), in Brasília. His desire for more knowledge brought him to the University of Florida in 1990. He earned a Doctor of Philosophy degree from the University of Florida, Department of Environmental Engineering Sciences in 1994. He is a mostly proud father of two boys, Carlos Filho and Diego.

I certify that I have read this study and that in my opinion it conforms to acceptable standards of scholarly presentation and is fully adequate, in scope and quality, as a dissertation for the degree of Doctor of Philosophy.

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This dissertation was submitted to the Graduate Faculty of the College of Engineering and to the Graduate School and was accepted as partial fulfillment of the requirements for the degree of Doctor of Philosophy.

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