Development of Biomanipulation Strategies for the Remediation of Eutrophication Problems in an Urban Reservoir, Lago Paranoá, Brazil.

by

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To my Family

DECLARATION

I hereby declare that this Thesis has been composed entirely by myself and has not been submitted in any previous application for a degree.

The work has been carried out by myself. The nature and extent of any work carried out by, or in conjunction with, others has been specifically acknowledged by reference.

This is also to certify that I have carried out the experimental work and produced the scientific content and composition of the join paper entitled:

"Experimental investigation of the feasibility of improving water quality by controlling exotic planktivore over-population in the eutrophic Paranoá Reservoir (Brasília, Brazil)."

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SUMMARY

The feasibility of improving water quality by food web manipulation in an urban eutrophic tropical reservoir, Lago Paranoá (Brasília, Brazil) was investigated. The potential of promising biomanipulation strategies was experimentally tested in limnocorrals: reduction of internal nutrient loading by controlling tilapia over-population, and suppression of nuisance cyanobacteria by stocking non reproducing filter-feeding silver carp.

A four-month test in two large littoral isolated areas of the reservoir (1,000 m^2 each) stocked with high vs. low tilapia biomass revealed that fish over-population promoted blooms of *Microcystis* and decreased water clarity by supplying phytoplankton with additional nutrients (mainly phosphorus, "P"). Since reduction of tilapia biomass from 150 to 40 g/m^3 resulted in significant water quality improvements, the release of commercial fisheries using cast-nets was recommended.

Field experiments in floating net-cages (10 m^3) were conducted to evaluate silver carp adaptation to Lago Paranoá and to determine fish growth rates when maintained feeding exclusively on the abundant plankton in the reservoir. High growth rates (up to 6 g/day) and survivorship (> 90%) of young-of-the-year (72 g), juveniles (300 g) and adult silver carp (1100 g) during both dry and rainy season were observed. Efficient ingestion of nuisance algae was evidenced by the dominant presence of *Microcystis aeruginosa* and *Botryococcus braunii* colonies in the fore-intestine of experimental fishes. Results indicate the great potential for silver carp cage culture as a low-cost and environmentally beneficial economic activity.

Following indications from literature and previous studies in Lago Paranoá, the optimum range of silver carp biomass which maximize phytoplankton control was determined in ten large replicated limnocorrals (50 m^3). Although a significant decrease in *Microcystis* abundance was achieved by stocking silver carp at all biomass levels, net-phytoplankton biomass was only significantly suppressed at moderate fish stocking densities of 40 and 60 g/m^3.

Two additional limnocorral experiments during dry and rainy seasons were performed to test the effectiveness of both biomanipulation strategies when adopted separately or simultaneously. Water quality improvements through control of tilapia abundance (from 100 to 40 g/m^3) and stocking with silver carp (at moderate stocking rates of 40-50 g/m^3) induced significant decreases in total phosphorus (21-31%), cyanobacteria density (40-44%) and phytoplankton biomass (22-38%). As those strategies were found to act independently, the combination of both enhanced water quality benefits by reducing total phosphorus by 38%, cyanobacteria density by 75% and phytoplankton biomass by 60%.

To predict the internal P-loading reduction that could be achieved by implementing both biomanipulation strategies on a whole-reservoir scale, P excretion rates were quantified in indoor tanks for 16 and 40 g tilapia (0.527-1.576 µg SRP/g ww/h) and silver carp (0.391-0.737 µg SRP/g ww/h). It was estimated that the tilapia contribution to internal P-loading (5.4 mg TP/m^2/day for 1,300 kg/ha) is equivalent to external P input to the Bananal Branch (6.0 mg TP/m^2/day). A 60% reduction in tilapia biomass plus stocking of silver carp at densities of up to 60 g/m^3 after three years would not change internal P-loading (5.6 mg TP/m^2/day) but would (1) reduce by 60% the input of "new" phosphorus to the water column from tilapia bottom feeding, and (2) increase grazing on phytoplankton following silver carp introduction.

In addition to the ecological advantage of replacing the use of copper sulphate by silver carp grazing to control undesirable nuisance cyanobacteria, important socio-economic benefits would be achieved from initiating a commercial tilapia fishery using cast-nets, as a subsistence activity to the low-income population of Brasília. The present work also discusses the applicability of such alternative biomanipulation approaches to other eutrophic ecosystems in the tropics and subtropics.

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CHAPTER I

GENERAL INTRODUCTION

I.1. Eutrophication

Eutrophication is generally defined as the process of nutrient enrichment of a water body in terms of phosphorus and nitrogen associated with enhanced autotrophic production (algae and macrophytes) that the added loads stimulate (Reynolds, 1992). The term originally was used to refer to the long-term natural aging process of lakes, in which nutrients from allochthonous (or external) sources and organic matter of terrestrial origin accumulate in a lake basin, gradually decreasing lake volume through sedimentation and filling in, and increasing autochthonous production which slowly enrichs the lake (Welch, 1980). The speeding up of such natural and geological time-scale process by human interference, resulting in increased external nutrient supply and drastic water quality deterioration over few decades, is termed cultural eutrophication (Rast & Thornton, 1996). Man-made sources of nutrients accompanying human population growth and agricultural and urban development include wastewater (domestic and industrial sewage), rural and urban runoff associated with the use of fertilizers and detergents, and activities that increase the availability of nutrients originally present in the soil from the drainage basin of the water body (Margalef, 1983).

General causes and consequences of eutrophication process

Among the three macronutrients potentially responsible for eutrophication process, both nitrogen and carbon are generally abundantly available as a result of atmospheric input while phosphorus, at least in temperate systems, is most frequently considered the limiting nutrient for algal and rooted plant growth in the majority of freshwater water bodies (Wetzel, 1993).

The external inorganic nutrient input to a lake or reservoir from its surrounding drainage basin increases photosynthetic production rate and plant biomass, reducing the depth of light penetration and limiting further phytoplankton and macrophyte growth due to self-shading. In extreme situations, the development of nuisance algal blooms (mainly cyanobacteria) may result in the ecological tragedy of massive fish mortality and decomposition due to oxygen depletion. Alternatively, excessive macrophyte growth restricts the flow of water, impedes navigation and provides shelter and favorable conditions for the development of vectors of diseases such as mosquitos and snails (Suess & Dean, 1980). High decomposition rates of excessive plant material and accelerated nutrient cycling promote sediment enrichment with organic matter, creating anoxic conditions in the hypolimnion which favor further internal nutrient release. As eutrophication proceeds, bottom sediments tend to act mostly as a source rather than as a sink for phosphorus, generating the so-called internal nutrient loading (Margalef, 1983).

Effects on the food chain

The most important and harmful symptom of eutrophication is represented by the excessive growth of cyanobacteria at the autotrophic level (Walsby, 1992). Key factors for explaining the prevalence of such phytoplankton group include the competitive advantage for the uptake of inorganic carbon at high pH (Shapiro, 1973), the ability of fixing nitrogen directly from atmosphere and the capacity of performing

vertical migration due to the presence of gas vesicles for buoyancy (Sevrin-Reyssac & Pletikosic, 1990).

The progressive dominance of large-sized, unpalatable and frequently toxic cyanobacteria prevents grazing from zooplankton community by clogging the filtering apparatus of large-bodied herbivores (Gliwicz, 1990). As a consequence, a shift in zooplankton size-structure towards small-bodied forms such as rotifers and protozoans tends to occur in eutrophic systems (Crisman, 1986). Such microzooplankton organisms are able to explore increased bacterial biomass associated with net-phytoplankton bloom more efficiently than macrozooplankton (Bays & Crisman, 1983). The concomitant enhancement of fish biomass and change from piscivores to more tolerant planktivores (mainly cyprinids) with progressive eutrophication (Persson *et al.*, 1988) intensify predation pressure on macrozooplankton and also contribute to microzooplankton dominance. The result of the eutrophication process is thus a decrease in the efficiency of energy transfer between producer and consumer levels in the pelagia with the replacement of herbivory (grazing) by dominant detritivory pathway in the food chain (De Bernardi & Giussani, 1995).

As a consequence of such "bottle-neck" between dominant phytoplankton and zooplankton, secondary production may be dominated by benthic macroinvertebrates and bottom-feeding fish which efficiently explore abundant detritus and sedimented material (Tátrai, 1995). Although an enhancement of overall fish production with trophic state is expected from its direct relationship with primary production, the number of species decreases and substantial shifts in species composition towards low-value cyprinid species usually occur (Biró, 1995). Moreover, when extreme hypertrophic conditions are reached, fish yields tend to stabilize or even decrease as a

result of massive mortality caused by adverse environmental conditions (Barica, 1980).

Such detrimental consequences of nutrient enrichment on each trophic level are generally events common to ecosystems located in all geographical/climatic regions and represent a source of water quality degradation on a global scale. However, limnological specificities make tropical and subtropical water bodies especially susceptible to reach higher trophic state than equivalent temperate ecosystems (Nilssen, 1984; Lewis, 1987).

Tropical versus temperate ecosystems

As discussed by Lewis (*op.cit.*), fundamental distinctive physical properties of tropical lakes are higher mean solar irradiance and water surface temperature, resulting in elevated primary production. Such higher production at low latitudes is sustained by a more efficient nutrient recycling system responsible for both a faster release of limiting nutrients from non-living organic matter to producers and a higher rate at which nutrients are re-supplied to the euphotic zone from deeper layers and bottom sediment. Temperature regime also predisposes deep lakes in the inter-tropic regions to a stable stratification and therefore to a deoxygenation of deep layers (Rast & Thornton, 1996). This trend towards lower hypolimnetic oxygen concentration at lower latitudes is magnified by longer stratification season and higher oxygen demand due to higher microbial production when compared to temperate lakes (Lewis, 1987). Increased and faster rates of production, decomposition and recycling processes in warm-waters explain the generally higher nutrient concentrations representing the boundary between oligotrophic, mesotrophic and eutrophic conditions in tropical lakes, making them probably best comparable to temperate systems of higher trophic state in many respects (Nilssen, 1984; Rast & Thornton, 1996).

Despite such differences in terms of degree of eutrophication between lakes located at contrasting latitudes, the control measures used in the temperate zone should also work in the inter-tropical regions, although with contrasting efficiencies (Rast & Thornton, 1996).

1.2. Restoration

For many decades, limnological research was directed towards the identification of the primary causes of eutrophication and quantification of its adverse effects, notably nuisance algal blooms (Reynolds, 1992). During the 1970's, consequences of nutrients enrichment at each trophic level became well understood and empirical relationships between phosphorus and water quality response parameters were developed for temperate lakes (Rast & Thornton, 1996).

In recent years, the worldwide progressive water quality deterioration caused by nutrient enrichment associated with fast and uncontrolled human population growth, motivated limnologists to draw special attention on applied research investigating possible measures to control eutrophication (Bernhardt, 1992).

In a broad sense, restoration measures may be divided in the following two categories: (a) control of external nutrient loading, and (b) reduction of internal sources, symptoms or consequences of nutrient enrichment to the ecosystem (Koschel, 1995).

Traditional engineering approaches to control external nutrient loads

The first and definitely most important step in reversing the eutrophication of a given water body is to determine and control the main sources of nutrient input to the ecosystem (Bjork, 1994). In contrast with diffuse or non-point-sources of nutrients, such as agricultural runoff, which are difficult both to identify and to control, point-

sources of nutrients of anthropogenic origin, such as sewage discharges, can usually be identified, quantified and eliminated (Crisman, 1986). The two main techniques to control point-sources are diversion away from the drainage basin of the water body, and advanced wastewater treatment to remove nutrients (Welch, 1980).

Sewage diversion can be effective, provided the receiving water body has adequate capacity to assimilate the incoming nutrient loading. Results from 16 lakes submitted to sewage diversion revealed the occurrence of water quality improvements in the majority of cases (Welch, 1980). However, in many circumstances, exporting sewage away from the watershed simply displaces the problem of excessive nutrient from one location to another, making costly sewage treatment recommended.

Advanced tertiary wastewater treatment represents the most modern and efficient technique capable of removing organic matter and nutrients (N and P) through biological (bacteria) or chemical (precipitation) means. The main restrictions in its use are the high costs involved and the extra sludge generated for disposal (Welch, 1980).

The non-point-sources of nutrients, although very difficult to control, can be reduced by dilution of influents or lake water with groundwater or water from other sources. Because of the large volume of water required, reduction of nutrient concentration by adding low-nutrient water is more restricted to reservoirs or even lakes supplied with inflow from large streams or rivers (Crisman, 1986).

Independent of the technique adopted, the significant reduction of external nutrient loading to a given lacustrine ecosystem is expected to reflect in water quality improvements, as was the case with Lake Washington, Green Lake, Shagawa Lake and Moses Lake, in the United States; Lake Malaren and Lake Norrviken, in Sweden; Lake Zurich, in Switzerland; Lake Lyngby-So, in Denmark; Lake d'Annecy, in

France and Lake Tegernsee, in Germany (Welch, 1980; Willén, 1987; Rast & Thornton, 1996).

However, the literature contains numerous examples where no appreciable improvement in water quality had occurred following many years of controlling the external nutrient loading (Sas, 1989). Case studies such as Lakes Lansing, Sammamish, Snake, Stone, and Okeechobee in the United States (Welch, 1980; Havens *et al.*, 1996), Lake Rotsee, Switzerland (Welch, *op.cit*), Lakes Trummen and Trehorningen, Sweden (Bjork, 1994; Rast & Thornton, 1996), Esthwaite Water, in United Kingdom (Heaney *et al.*, 1992), Lakes Gross-Glienicker and Haussee, in Germany (Wolter, 1994; Koschel, 1995) and Lake Balaton, Hungary (Padisák, 1994), illustrate the failure of reducing external loading alone and clearly demonstrate that additional in-lake measures are required to ensure lake restoration.

The main reason for lakes being buffered against reduced external loading is that nutrient and organic matter which largely accumulate at the sediment surface during the eutrophication process, may act as an enormous nutrient pool sufficient to sustain high autotrophic production in the lake water long after the elimination of external nutrient inputs (Boers *et al.*, 1991; Wetzel, 1993). The release of nutrients stored in the sediments back to the water column constitutes the main source of the so-called internal loading. It is well recognized today that not only internal nutrient loading from bottom sediment but also biologically-induced acceleration of nutrient recycling rates (Phillips *et al.*, 1994; Koschel, 1995) have a decisive role in the success of lake restoration (Bjork, 1994).

In-lake restoration techniques

Several restoration methods have been developed to remove nutrients from the water column or to prevent the release of nutrients from organic sediments as complementary measures to control external nutrient input. The main in-lake

techniques applied to reduce eutrophication symptoms and to counteract internal nutrient loading by transforming the ecosystem from a nutrient source to a nutrient sink are flushing, nutrient precipitation, sediment inactivation, drawdown, aeration, dredging and control of primary producers (Wetzel, 1993; Bjork, 1994).

Flushing or shortening the residence time in reservoirs by deliberate manipulations of the hydrological regime represents a physical method aiming at wash out nutrients and phytoplankton biomass during critical periods of algal blooms (Tundisi, 1986). This technique does not attempt to reduce nutrients release from sediment and requires the availability of a high volume of water to be renewed within a short time (Crisman, 1986).

Nutrient precipitation by application of phosphorus-binding compounds to the water is a chemical method for reducing phosphorus availability to primary producers through its inactivation (Wolter, 1994). This process is induced by adding to the water aluminum (aluminum sulfate-*alum* or sodium aluminate) or iron compounds (iron III-chloride, $FeCl_3$) to form relatively stable phosphorus-binding compounds which settle in the form of gelatinous flock (Wetzel, 1993). Although one application may last a couple of years, this treatment is best suited for small lakes due to the costs involved and also faces the disadvantage of potentially causing smothering on benthos and toxicity in fish (Welch, 1980). Moreover, addition of iron chloride or aluminum sulfate to water bodies with low alkalinity may reduce the buffer capacity and cause acidification (Wolter, *op.cit.*).

If such compounds are applied in large amounts to the water or directly to the sediment, they will form a layer on the sediment surface impeding phosphorus release to the overlying water and promoting the **sediment inactivation** (Wetzel, 1993). Other combinations of chemicals, such as nitrate and iron, have been successfully used to enhance the retention efficiency of sediments, as nitrate stimulates natural

process of denitrification and oxidation of organic matter and delays the reduction of iron and the associated release of phosphate (Ripl, 1994). A modified method based on natural calcite precipitation or biogenic decalcification to reduce the amounts of phosphorus has been experimentally tested in Germany (Klapper, 1992). The lake lime in the deep layers of bottom sediment is flushed out with a rinsing jet equipment and the water-calcite suspension pumped and spread over the phosphorus-rich uppermost sediment to promote the co-precipitation of calcite with phosphorus from the water column.

Floculent sediments resulting from increased sedimentation rate of sestonic material can be easily resuspended favoring nutrient release. Therefore, sediment desiccation, as a result of lake **drawdown** or water-level lowering, may be use as an alternative method of sediment isolation through compaction. According to Crisman (1986), the surface layer of such exposed sediment becomes so durable that subsequent lake-filling does not affect its integrity.

Another process to reduce nutrient release from sediment is to keep its surface oxidized so to create aerobic conditions in the hypolimnion through **aeration** or artificial circulation (Crisman, 1986). Hypolimnetic aeration alone without destratification or complete artificial circulation of the water column can also induce precipitation of phosphorus, iron manganese, increase pH, and reduce internal phosphorus loading (Verner, 1994). Artificial circulation also function as a phytoplankton control measure, reducing the average time spent by phytoplankton in the euphotic zone and causing light to become a limiting factor for its production and growth (Oskam & Breemer, 1992). Intermittent destratification proved to be more efficient in controlling cyanobacteria blooms than permanent destratification as the created conditions of alternating disturbance and quiescence as well as drastic and sudden changes in light regime prevent cyanobacteria from adapting and

outcompeting eukaryotic algae (Steinberg & Gruhl, 1992). As intermittent destratification not only stimulates phytoplankton sedimentation (or decaying) but also herbivorous zooplankton grazing on more edible algal forms and light limitation, this process is accompanied by an overall reduction of phytoplankton biomass (Steinberg & Gruhl, *op. cit.).*

Sediment removal or **Dredging** represents the most definite, but also perhaps the most costly in-lake restoration technique (Welch, 1980). It involves the physical removal of surface layers of the organically-enriched sediment, based on the premise that nutrient content of the sediment decrease with depth and, thus, relatively nutrient-poor deeper sediments will be exposed (Crisman, 1986). Sediment removal is recommended only for shallow lakes suffering from accelerated filling in from highly organic sediment whenever all above-mentioned simpler methods have proved ineffective (Bjork, 1994). When nutrient-rich top sediment is pumped, the proportion of sediment and water should be constant and caution should be taken to avoid increasing tubidity by sediment resuspension (Bjork, 1994). Lake Trummen is a very illustrative example of successful lake restoration through sediment removal (Andersson, 1988; Bjork, *op.cit.*). Immediately following the removal of 0.5-1 m of black, phosphorus-rich sediments from one km^2 area through suction dredging, drastic reductions of phosphorus, nitrogen and phytoplankton biomass took place and were maintained for many years (Graneli, 1987). Lake Trummen experienced a surprising deterioration of the water quality five years after sediment removal, but it was a result of increased biomass of benthivorous and planktivorous fishes negatively affecting trophic conditions. Subsequent large scale reduction of cyprinid fish biomass restored water quality to the previous desirable levels (Andersson, 1988).

Among all in-lake restoration methods, the **control of primary producers** represents the only one fighting the symptom rather than the cause of eutrophication.

Consequently, most of such ecologically-based techniques of controlling excessive autotrophic biomass are frequently viewed only as palliative measures of limited efficacy and restricted applicability being also subdued to considerable debate (Wetzel, 1993). Methods to control the excessive growth of macrophytes resulting from eutrophication processes include water level reductions, introduction of infectious pathogens, reduction of light availability by shading, chemical alteration of sediments and harvesting of plants by mechanical means, herbivorous insects or fishes (Crisman, 1986).

Despite the success associated with many of those techniques, emphasis is given here to the biological methods using phytophagous fish as control agent. Congo tilapia (*Tilapia rendalli*) and grass carp (*Ctenopharyngodon idella*) are among the species with the greatest potential for removing macrophyte biomass (see Crisman, 1986 for useful review). Adaptation to a wider range of temperatures and restricted breeding requirements preventing uncontrolled proliferation in new ecosystems are features favoring the selection of grass carp over tilapia for this purpose. The main point against the use of herbivorous grass carp is the possible development of undesirable algal biomass associated with increased nutrient availability due to fish feeding on macrophytes. According to Crisman (*op.cit.*), the best strategy for the most effective macrophyte control involves the integration of mechanical, chemical and biological techniques. Preliminary use of mechanical and chemical methods to reduce plant biomass followed by introduction of a biological control agent would ensure that plant populations remain low for a longer period of time.

Similarly, mechanical, chemical and biological methods have also been used to control phytoplankton biomass in lacustrine ecosystems. Mechanical harvesting of surface-accumulating algal scums using a portable algal-skimming unit has been tested but considered to be of limited practical application (Crisman, 1986).

Chemical methods, particularly the use of algicides such as copper sulphate, represent the most frequently used technique to control undesirable algal growth worldwide (Branco, 1986). The main advantages of copper sulphate are its high specificity to kill blue-green algae (cyanobacteria) and its relative low cost compared to other chemicals. Contamination of sediments with accumulating copper and toxicity to heterotrophic organisms such as zooplankton, benthos and fishes are among the main disadvantages associated with this product (Crisman, 1986).

Biological control of phytoplankton using pathogens and grazers exemplifies a promising technique derived mainly from laboratory tests but to date not sufficiently evaluated under practical field conditions (Shapiro *et al.*, 1982; Crisman, 1986). Although pathogenic organisms such as fungi, virus and bacteria have been shown to have a great potential for controlling cyanobacteria under laboratory conditions, large-scale field experiments are still lacking. The use of grazers such as zooplankton and fish to control algal abundance in lacustrine ecosystems is part of the so-called food web manipulation or biomanipulation and will be discussed in details in the following section.

1.3. Food web manipulation

Traditionally, limnologists had not paid much attention to the role of fishes as top consumers capable of affecting water quality so that the management of lakes and reservoirs had rarely included fisheries management (Gophen, 1990a; Hrbácek, 1994). Aquatic food webs have for long been considered to be structured solely according to an ascendant flux of energy from producers to consumers in the presence of nutrients and solar radiation (Straskraba, 1965).

The idea of managing lacustrine ecosystems suffering from eutrophication through food web manipulation was originated in the 1960's thanks to the pioneering works of Hrbácek *et al.* (1961) and Brooks & Dodson (1965). These studies provided the first field evidences for the indirect effect of fish on phytoplankton through size-selective predation on zooplankton. In the absence of size-selective visual planktivores, the zooplankton community shifted from small to large-bodied organisms (mainly Daphnids) which outcompete rotifers and small cladocerans by feeding more efficiently on a wider algae size range (Crisman, 1986; Gulati, 1995a). As a consequence of a trophic-cascade effect, phytoplankton biomass was then controlled by increased grazing pressure from *Daphnia* and transparency levels were improved.

This alternative feedback-controlled pathway in the food chain has been called "trophic cascade effect" or "top-down force" (Carpenter *et al.*, 1985; McQueen *et al.*, 1986). The term biomanipulation was first coined by Shapiro *et al.* (1975) to designate any deliberate interference in a given key component of the food chain aiming at lake restoration. Although a number of other measures to reduce the dominance of cyanobacteria such as artificial destratification and CO_2 injection have also been recommended and successfully tested (Shapiro *et al.*, 1982), biomanipulation approaches have mostly aimed at enhancing the grazing pressure on excessive phytoplankton biomass (Gophen, 1990a).

"Classical" biomanipulation based on increasing zooplankton grazing

Few years following the preliminary field observation of predation-induced food web modifications promoting water quality improvements in fishless vs. fish stocked ponds in Eastern Europe (Hrbacék *et al.*, 1961), Brooks & Dodson (1965) formulated the "size-efficiency hypothesis". According to this hypothesis, phytoplankton biomass and sucession are directly regulated by the grazing pressure of herbivorous

zooplankton, itself controlled by the planktivorous fish predation pressure. If size-selective predation by planktivorous fish on large-sized cladocerans is limited, such efficient phytoplankton feeders may be able to maintain phytoplankton biomass at low levels.

The possibility of controlling algal growth through food web manipulation as a simple, low cost, and "environment-friendly" alternative to traditional enginnering approaches, created a enormous enthusiasm among limnologists during the past two decades (Kasprzak *et al.*, 1993). Consequently, in limnological publications, there has been a remarkable interest in documenting "top-down" effects of fish (Northcote, 1988).

A large number of biomanipulation experiments confirmed the great potential of this approach based on the removal of planktivorous fishes as a complementary restoration technique in temperate zones (see Lazzaro, 1987; Northcote, 1988; Gophen, 1990a; Lammens *et al.*, 1990; De Mello *et al.*, 1992; Gulati, 1995a; Kasprzak, 1995, for useful reviews). Many studies quantified predator-prey relationships in fishes by using laboratory experiments, modeling, and behavioral observations (e.g. Confer & Blades, 1975; O'Brien *et al.*, 1976; Drenner *et al.*, 1978; Werner & Hall, 1974). Subsequently, experimental limnologists intensified the use of enclosures to examine the responses of zooplankton populations to selective fish predation and its cascading effects on phytoplankton (e.g. Anderson *et al.*, 1978; Lynch & Shapiro, 1981; Tatrai *et al.*, 1985). Additionally, whole-lake experiments involving reductions or removals of planktivores, and/or introductions of piscivores into lacustrine ecosystems provided evidences for the use of zooplankton-phytoplankton-transparency interactions as a tool to improve water quality (e.g. Stenson, 1978; Henrikson *et al.*, 1980; Benndorf *et al.*, 1984; Shapiro & Wright, 1984; Wright & Shapiro, 1984).

The occurrence of side-effects, as well as some unexpected outcomes and the short term nature of the beneficial results, indicated the need to include some other components of the food web in order to elucidate the complex phytoplankton-zooplankton relationship (Lammens *et al.*, 1990). The number of components that were studied increased as nutrients and fish mediated nutrient effects, bacteria, microzooplankton, picoplankton and macrophytes were also considered and sometimes incorporated into models (Gophen, 1990b). Apart from a unique example of successful restoration by biomanipulation alone without any considerable reduction of external nutrient loading (Gulati, 1995b), experiences accumulated over nearly three decades of food web manipulations emphasise the need to conduct fish management in conjunction with reduction in external nutrient loading (Gulati, 1995a).

Although the control of planktivores by selective fishing, poisoning or piscivores stocking was generally found to be effective in maintaining larger *Daphnia* population capable of controlling phytoplankton biomass and producing a longer and increased clear-water phase, such desirable effect rarely lasted more than a couple of years (Reynolds, 1994). Instability of the ecosystem which tended to return to its original situation as soon as the biomass of the planktivore population started to recover, makes periodic adjustments through re-application of biomanipulation, necessary to maintain and stabilize the positive effects of the food web management (Matena *et al.*, 1994). More recently, Shapiro (1990) proposed the use of a series of refuges for large-bodied zooplankton such as light, temperature, oxygen, macrophytes, and behavioral refuges in order to ensure these grazers continued to coexist in lakes and extend the effects of biomanipulation.

As pointed out by Lammens *et al.* (1990), successes and failures accumulated over three decades of experiences on this "classical" biomanipulation approach

demonstrate that crucial obstacles for oligotrophication of lacustrine ecosystems in temperate regions are: (1) ecosystem stability and resilience (Carpenter *et al.*, 1992), (2) the high complexity of the food web, giving unexpected indirect effects of biomanipulation (Benndorf, 1992) and (2) dominance of large-sized ungrazeable cyanobacteria in phytoplankton assemblage during eutrophication (De Bernardi & Guissani, 1990).

An alternative approach based on direct phytoplankton grazing by filter-feeding planktivores has been considered as a particularly promissing echotechnology to tropical and subtropical ecosystems, although still poorly evaluated (Nilssen, 1984; Lazzaro, 1987; Gophen, 1990a).

Alternative biomanipulation based on phytoplanktivorous fish

Tropical and subtropical lacustrine ecosystems differ greatly from temperate lakes in the basic food web configuration as large bodied herbivorous zooplankton from the genus *Daphnia* are rarely reported (Fernando, 1980) and the size range of prevailing small-bodied grazers is too small to efficiently exploit dominant large-sized algae (Nilssen, 1984). This frequent uncoupling between phyto- and zooplankton link in tropical ecosystems accentuate the general tendency for detritivory to overcome herbivory in the food chain, as eutrophication proceeds (De Bernardi & Guissani, 1995).

In tropical and subtropical systems, size-selective visual zooplanktivores are generally replaced by filter-feeding omnivores (Nilssen, 1984). This type of planktivorous fish is simultaneously able to prey on dominant microzooplankton and compete with such small-bodied grazers for large-sized algae and other sestonic food. Although the impacts of filter-feeding fish on plankton are much less documented than visual feeders, it is well known that filter-feeding planktivores directly suppress zooplankton (mainly less evasive organisms) and net-phytoplankton but indirectly

promote the development of small algal forms such as pico- and nanoplankton (Lazzaro, 1987).

The potential for cropping nuisance net-phytoplankton (mainly cyanobacteria) through fish grazing has been suggested and evaluated in a number of studies (see Lazzaro, 1987; Northcote, 1988 and Gophen, 1990a for useful review). In general, intense fish grazing pressure on net-phytoplankton shifts size structure of phytoplankton assemblages towards small edible forms without necessarily reducing total algal biomass (Drenner *et al.*, 1984a, 1984b, 1987; Vinyard *et al.*, 1988).

By reviewing data from 48 published experimental studies, Drenner *et al.* (1996) showed that, in the large majority of cases, omnivorous filter-feeding fish enhance phytoplankton biomass and primary productivity despite suppressing net-phytoplankton. Nanoplankton, which are not grazed by fish and which have comparatively higher growth rates, take benefits from fish mediated effects of removing both competitors (net-phytoplankton) and consumers (herbivorous zooplankton) and transfering nutrients from the sediment. Drenner *et al.* (*op. cit.*) concluded that omnivorous fish seems to interact synergistically with lake trophic state making its effect more intense with increased eutrophication.

Nevertheless, the frequent inclusion of many well-known benthivorous fishes such as common carp, bream and roach under the same category of omnivores makes it dangerous to generalize about the effects of filter-feeding fish on water quality. For instance, among those 48 studies reviewed by Drenner *et al.* (1996), almost half involved benthivorous species and only some one third referred to typical phytophagous fish (tilapia and silver carp) which were exactly the ones accounting for the few cases of successfull algal control.

The American clupeid gizzard shad (*Dorossoma cepedianum*), listed in nine of the above studies, represent another intensively studied and controversial filter-feeding

omnivore. Although this species effectively feeds on phytoplankton larger than 40 μm in addition to detritus and zooplankton (Drenner *et al.*, 1986; Mummert & Drenner, 1986), poor digestion efficiency of cyanobacteria (Crisman & Beaver, 1990), bottom feeding habits (Schaus *et al.*, 1997) and high carrying capacity in terms of total biomass attained in lakes (Lazzaro *et al.*, 1992), are among the main reasons for the enhancement of phytoplankton abundance by gizzard shad.

Taking this fish species as example, Crisman & Beaver (1990) considered it difficult to generalize the role of filter-feeding planktivores as a group and suggest that future research should concentrate on defining the influence of individual fish taxa on phytoplankton. These authors emphasize, however, that the greatest potential for biomanipulation in eutrophic subtropical and tropical lakes lies on phytoplanktivorous fish.

The most recommended filter-feeding planktivore to biological control of undesirable algae is the Chinese silver carp (*Hypophthalmichthys molitrix*). Laboratory grazing experiments (Herodek *et al.*, 1989; Smith, 1989) demonstrated the high retention efficiency of silver carp for particles > 10 μm due to the combination of a powerful suction ability and a well-developed filtering apparatus with mesh size averaging 14-20 μm (Wilamovski, 1972; Spataru & Gophen, 1985).

However, results from various field tests involving silver carp stocking into enclosures, fish ponds, lakes and reservoirs are rather contraditory (see Costa-Pierce, 1992 for useful review). Among the key factors explaining successes and failures of silver carp are the share of large-sized algae in phytoplankton community (Laws & Weisburd, 1990), location of the ecosystem (Spataru & Gophen, 1985), strength of the zooplankton-phytoplankton link (Arcifa *et al.*, 1995) and fish stocking density and biomass (Milstein, 1992; Starling, 1993a). Besides, environmental features associated with decomposition processes and nutrient cycling such as water temperature, mixing

pattern, residence time and trophic state may obscure beneficial effects of filter-feeding planktivores (Lazzaro, 1987).

According to Crisman (1986) and Lazzaro (*op. cit.*), this alternative trophic cascade pathway based on the use of herbivorous fish should be more experimentally explored in ecosystems located outside temperate regions considering the inadequacy of the "classical" biomanipulation approach based on zooplankton grazing.

As summarized by Lazzaro (1997), the applicability of this "classical" food web manipulation to tropical lakes is mainly challenged because top-down links between piscivores and planktivores are weakened by: (i) higher carrying capacity of dominant filtering omnivores due to lack of food limitation and weak predation pressure when compared to that on zooplanktivores and (ii) frequent replacement of large, open-water strictly piscivores by less efficient predators such as small-sized, sedentary, sit-and-wait carnivores. Simultaneously, zooplankton grazing pressure on phytoplankton is weakened by: (i) all year-round intensive invertebrate predation preventing any increase in abundance of large herbivores, (ii) lack of large-bodied *Daphnia,* (iii) a quasi-permanent size-selective predation pressure from fish larvae and juveniles resulting from year-round spawning of many tropical fish species, and (iv) enhanced nutrient cycling and regeneration mediated by higher planktivore biomass and smaller zooplankton size.

1.4. Background data from Lago Paranoá

Lago Paranoá (15° 48' S and 47° 50' N) is a urban man-made lake created mainly for aesthetic and recreational purposes in 1959, when Brasília, the new capital of Brazil, was constructed. Inserted in a drainage basin of 1015 km^2, Lago Paranoá (38 km^2 surface area; 498.10^6 m^3 volume; 40 m total depth, 13 m mean depth and 299

days of total retention time) is formed by five main tributaries: Bananal and Torto to the north and Gama, Riacho Fundo and Cabeça do Veado to the south.

General climatic conditions are characterized by a well defined rainy season (October-March, with mean monthy precipitation of 204 mm) and dry season (April-September, with average precipitation of 33 mm). Several limnological studies identified this ecosystem as a typical warm monomitic reservoir with mixing period in winter (June-July) and stratification with an anoxic hypolimnion maintained throughout the rest of the year in deeper areas (Mattos *et al.*, 1992).

During the first decade after filling, Lago Paranoá started developing eutrophication symptoms due to the inflow of untreated and inadequately treated domestic sewage throughout the Southern and Northern Wastewater Treatment Plants (Palmer, 1969; Oliveira & Krau, 1970; Branco, 1976). During the late 1960's and early 1970's, dominant Desmidiacea were rapidly replaced by filamentous Cyanobacteria (*Cylindrospermopsis raciborskii*), which established a situation of permanent bloom averaging more than 90% of phytoplankton biomass (Cronberg, 1977). By conducting field bioassays, Lindmark (1977) identified phosphorus as the limiting factor for algal growth and Pinto-Coelho (1983) verified that dominant *Cylindrospermopsis* was not subdued by intense grazing pressure by zooplankton community.

In 1978, an extended drought period created favourable conditions for a massive bloom of *Microcystis aeruginosa* in the hypertrophic area of Riacho Fundo Branch. This ecological disaster resulted in a severe fish mortality and caused enormous distress to the inhabitants of Brasília. Within ten years of its first appearance at harmful densities, *Microcystis* became also widespread in Bananal Branch and Central Region of the reservoir, making it necessary to control abundance by addition of copper sulphate (Altafin *et al.*, 1995).

As a result of an international cooperation program with the United Nations Development Cooperation (UNDP), several aspects of Lago Paranoá eutrophication process were evaluated by a research group from the Institute of Limnology in Lund, Sweden (Graneli, 1987). The accumulation of a nutrient-rich organic sediment layer was detected nearby the Southern Wastewater Treatment Plant (Ennel, 1977) and budget calculations showed that over 70% of the phosphorus input into the reservoir enters via sewage (Hilmer, 1977). Consequently, Bjork (1979) recommended the diversion or the tertiary treatment of all sewage from the catchment basin as the main measure for lake restoration.

In the early 90's, Lago Paranoá Recovery Program was initiated with the construction of new biological nutrient removal plants capable of treating sewage at tertiary level using Modified Bardenpho Process together with final chemical polishing (Cavalcanti *et al.*, 1997). Following a 75% reduction in the external nutrient loading to the ecosystem, there has been a significant improvement in water quality parameters such as phosphorus, nitrogen, oxygen and transparency. Although chlorophyll-a has shown a trend towards lower concentrations throughout the reservoir in the past few years, persistence of nuisance colonial algal blooms (*Microcystis aeruginosa* and *Botryococcus braunii*) makes the application of copper sulphate still necessary (Pereira & Cavalcanti, 1996).

In a recent taxonomic study conducted by Branco & Senna (1996), a total of 76 phytoplanktonic taxa were listed and the cyanobacteria *Cylindrospermopsis raciborskii* remained the dominant species comprising up to 99% of total algal standing stock. Zooplankton community is still largely dominated by rotifers and small cladocerans and large-bodied herbivores are virtually lacking (Mattos *et al.*, 1997). However, following almost three decades of disapearance, *Daphnia gessneri* was detected again in the central area of the reservoir during the past four years, but

densities are still very low and restricted to the depths of 10-15 meters (Starling, *unpublish. data*).

In contrast to the well-documented evolution of limnological variables, data on fish community from Lago Paranoá are rather scarce. Since the early 1960's, exotic fish species such as tilapia (*Tilapia rendalli* and *Oreochromis niloticus*), common carp (*Cyprinus carpio*) and bluegill sunfish (*Lepomis macrochira*) have been stocked into this ecosystem (França *et al*, 1964). According to Dornelles & Dias Neto (1985), further introductions of exotic fish species have been carried out in 1964 and 1978, although details are not available.

In 1978-1980, a team from the Brazilian Organization for Fisheries Development (SUDEPE) conducted a survey to identify fish species present in the ecosystem. The following exotic fish species were captured: Congo tilapia (*Tilapia rendalli*), Nile tilapia (*Oreochromis niloticus*), common carp (*Cyprinus carpio*), bluegill sunfish (*Lepomis macrochira*), Pichcock pike (*Cichla ocellaris*), "tamoatá" (*Callichthys callichthys*), and a number of ornamental species (*Tricogaster sp., Xiphophorus spp.* and *Carassius auratus*). In addition, some native species originated from the tributaries such as "saguirú" (*Curimata elegans*), catfish (*Rhamdia sp.*), "traíra" (*Hoplias malabaricus*), "lambarí" (*Astyanax sp.*), "sarapo" (*Gymnotus carapo*), "acará" (*Geophagus brasiliensis*), "cascudo" (*Plecostomus sp.*) were also recorded.

From this survey using experimental gill-nets throughout the reservoir, it was found that tilapia and common carp were the most representative fish species in number and weight, respectively, and Riacho Fundo Branch was the area with the highest fish abundance in the entire reservoir. The implementation of professional fisheries using cast-nets to control tilapia over-population in this hypertrophic area was considered as a promising fisheries management strategy (SUDEPE, 1980).

Simultaneously, the total fish stock of this ecosystem was estimated as 3.000 metric tons by an echosounding survey (Dornelles & Dias-Neto, 1985).

The following and most recent study on fish community from Lago Paranoá was conducted during the late 1980's. Using a similar set of gill-nets in five different sampling stations of the reservoir, Grando (1989) investigated the feeding ecology of four abundant fish species. Tilapias (*Tilapia rendalli* and *Oreochromis niloticus*) showed a similar omnivorous feeding habit, exploiting bottom sediment (organic matter), vascular plants, plankton and insect larvae as main food items. Bluegill (*Lepomis macrochira*) and "saguirú" (*Curimata elegans*) had carnivorous and detritivorous feeding preferences, respectively. Tilapia over-population in Riacho Fundo Branch was revealed: tilapia capture in this area was five times higher than the corresponding average fish biomass in the other branches and the central body of the reservoir.

1.5 Preliminary Biomanipulation Experiments

The potential for food-web manipulation in Lago Paranoá was first evaluated during the late 80's through a combination of laboratory feeding trials and an enclosure experiment using four planktivorous fish species (Starling, 1989). The impacts of two abundant exotic species inhabiting the reservoir (Congo tilapia, *Tilapia rendalli* and bluegill sunfish, *Lepomis macrochira*) and two others recommended for introduction but not yet present so far in Lago Paranoá ("tambaqui", *Colossoma macropomum* and silver carp, *Hypophthalmichthys molitrix*) were compared when fish were stocked into 2 m^3 bag-type enclosures at high biomass (3,000 kg/ha).

From laboratory trials, feeding behaviour and rates were determined using fish fed with plankton from Lago Paranoá in 120-L aquaria. Bluegill, as particulate feeder, had values of feeding rate constant increasing with particle size, with maximum rates on adult copepods. However, all other fish species were observed to display a single filter-feeding mode, with maximum feeding rates on zooplankton prey of intermediate to smaller size. Silver carp presented the highest feeding rates among fish species for the majority of prey items and were the only species capable of efficiently grazing on filamentous cyanobacteria *Cylindrospermopsis raciborskii*.

Contrasting evolution of zooplankton abundances during the 31 days of an enclosure experiment occurred according to the stocked fish species (Starling & Rocha, 1990). Comparisons with fishless control enclosures showed that rotifers increased in the presence of tilapia, bluegill and tambaqui, but were drastically suppressed by silver carp. Silver carp also significantly affected phytoplankton abundance, as dominant cyanobacteria were reduced, when compared to enclosures containing other fish species. However, the absence of a significant contrast in phytoplankton biomass between silver carp and fishless enclosures was attributed to the nutrient release from live and dead fish.

By comparing these data from enclosures containing tilapia and silver carp, Starling (1993b) suggested that high tilapia abundance stimulates eutrophication in Lago Paranoá by enhancing microzooplankton (rotifers) and net-phytoplankton (cyanobacteria) biomasses through the speeding-up of nutrient cycling, while silver carp had a great potential for controlling nuisance cyanobacteria by grazing.

A further experiment in larger tube-type enclosures (6.5 m^3 each) showed that a relatively lower silver carp biomass (850 kg/ha) was capable of producing significant water quality improvements when compared to fishless enclosures (Starling, 1993a). Despite suppressing microzooplankton, silver carp significantly reduced net-

phytoplankton abundance and total chlorophyll-a and therefore was considered as a promissing biomanipulation tool. These results, as compared to those of the previous enclosure experiment (Starling & Rocha, 1990), demonstrated that a reduction in the magnitude of silver carp negative effects on zooplankton community and nutrient contribution were attained by lowering fish biomass. Consequently, the side effect of nanoplankton enhancement was prevented and an efficient control of total phytoplankton biomass could be achieved.

As emphasized by Arcifa *et al.* (1995), the above mentioned food web experimental studies conducted in Lago Paranoá provided preliminary evidence for both the negative impacts of abundant reservoir fish (mainly tilapia), and the suitability of using phytophagous silver carp to improve water quality.

1.6. Aims of the Present Work

The main objective of this study is to experimentally test the feasibility of controlling internal nutrient loadings by reducing tilapia over-population and cropping of nuisance cyanobacteria through silver carp grazing activity. Subsequent chapters describe a sequence of experiments, from laboratory trials to fish manipulations in limnocorrals, net-cages and large isolated littoral areas, carried out to evaluate the feasibility of implementing the two biomanipulation approaches on a whole reservoir scale.

In ***Chapter II***, a large-scale field test of the benefits to water quality from removal of 2/3 of tilapia biomass in littoral hypertrophic areas is carried out in order to enable a preliminary estimate of the importance of fish P excretion relative to the external P loading to Lago Paranoá.

Chapter III summarises a series of experiments carried out to evaluate the adaptation of caged silver carp to reservoir conditions, in which the sole source of food was plankton. The suitability of implementing a large-scale silver carp cage culture operation to crop nuisance cyanobacteria is discussed.

In ***Chapter IV***, the biomass range of free-roaming silver carp required to optimise the control of nuisance cyanobacteria is defined, based on a field experiment conducted in large replicated limnocorrals.

In ***Chapter V***, the results from two limnocorral experiments carried out to assess the benefits of controlling tilapia and/or stocking silver carp on water quality are presented .

Chapter VI details the results from laboratory measurements of tilapia and silver carp phosphorus excretion rates. The data are used to improve assessments of the importance of the fish community in the overall P budget of Lago Paranoá. Data are also used to explore biomanipulation strategies under contrasting scenarios to fulfil environmental management and socio-economic objectives.

Chapter VII sumarises conclusions and recommendations concerning the implementation of alternative biomanipulation techniques into tropical and subtropical ecosystems suffering from accelerated eutrophication process.

CHAPTER II

TILAPIA BIOMASS MANIPULATION IN LARGE AREAS

II.1. Introduction

Traditionally, limnologists have not paid much attention to fish as a key food-web component capable of affecting water quality in different ways (Gophen, 1990a). The important contribution of nutrient recycling by fish through excretion, egestion and mechanical stirring of bottom sediments to the enhancement of phytoplankton biomass has only been demonstrated recently (Threlkeld, 1987; Brabrand *et al.*, 1990; Kraft, 1992; Schindler *et al.*, 1993).

Enclosure experiments conducted in Lago Paranoá (Starling & Rocha, 1990) have revealed that a high tilapia biomass (up to 3,000 kg/ha in shallow hypertrophic areas) was associated with eutrophication symptoms (i.e. high phytoplankton abundance and low transparency). Starling (1993b) suggested that controlling exotic tilapia proliferation in this ecosystem would represent an appropriate management practice to reduce algal blooms in critical areas of the reservoir.

The purpose of this chapter is to conduct a practical test of fish-biomass mediated effects on a large scale, as close as possible to the reservoir conditions, by monitoring two 1,000 m^2 enclosed littoral areas of Lago Paranoá.

II.2. Material and Methods

During 1993 dry season (August 3-5), two littoral areas were created within a small bay located in one of the most eutrophic regions of the reservoir (Bananal Branch). A team of 5 experienced divers completely isolated these areas from the surrounding water, using PVC-reinforced sheets (Figure 2.1).

These sheets were composed of plastic canvas connected to metal poles inserted into the bottom sediment at 10 m intervals in a linear array. A plastic hose filled with sand was inserted into the lower part of each sheet to ensure a complete conformation with the bottom slope. To prevent water exchange between the experimental areas and the reservoir, a flap of the sheet was staked with clamps into the sediment. To prevent fish from jumping in or out, a plastic net (2 mm mesh) was erected up to 1 m above the water surface. Details of the vertical barrier constructed using PVC-reinforced sheets are given in Figure 2.2

On August 6-11, the two areas plus the adjacent open water reservoir area were sampled with seines and cast-nets, using a similar fishing effort, to estimate fish biomasses after the disturbance caused by setting up the barrier. All fishes captured were identified, measured and weighted. The overall average biomass obtained from the areas and adjacent reservoir was considered as that originally present in that region of the reservoir (Table II.1).

Fish community composition in the two experimental areas was quite similar, although the abundance of one (called HIGH BIOMASS area) was twice that of the other (designated LOW BIOMASS area). Indeed, many flooded trunks and roots plus dense littoral vegetation provided shelter for fishes in the HIGH BIOMASS (H.B.)

area, whereas the LOW BIOMASS (L.B.) area was free of hiding places, had comparatively less aquatic vegetation and consequently sheltered a lower fish biomass comparable to that of the adjacent opened area of the reservoir. Because of these differences, comparisons of H.B. vs. L.B. and adjacent reservoir (LAKE) areas were used to test the effects of reducing reservoir tilapia biomass.

Overall, 13 fish species were captured from the two areas and the adjacent reservoir: *Tilapia rendalli, Oreochromis niloticus, Curimata elegans, Aequidens portalegrensis, Lepomis macrochira, Cyprinus carpio, Xiphophorus sp., Astyanax bimaculatus, Poecilia reticulatus, Hoplias malabaricus, Callychthys callychthys, Gymnotus carapo, Planaltina myersi* and *Simbranchus marmoratus.* However, *Tilapia rendalli* and *Oreochromis niloticus* were the most abundant species, with 90% and 5% of the overall fish biomass, respectively. Therefore, on August 12-14, the original fish communities captured from the H.B. and L.B. areas were re-established using exclusively the two species of tilapias in the above proportion. Experiment started on August 16 by monitoring limnological characteristics of the areas and the adjacent reservoir at 3-7 days intervals.

Using a small boat, a 5-litre surface composite sample was collected from each area, by pooling samples from five stations within the area for temperature, pH, turbidity, Secchi depth, total alkalinity, dissolved oxygen, conductivity, total phosphorus (TP), orthophosphate (PO_4P), ammonia, nitrate, total Kjeldahl nitrogen, chlorophyll-a, phytoplankton primary productivity (dark and light bottles incubated for 2 hours around noon), phytoplankton (raw sample under inverted microscope at 400x magnification) and zooplankton (2-L filtered through 45 µm plankton net and

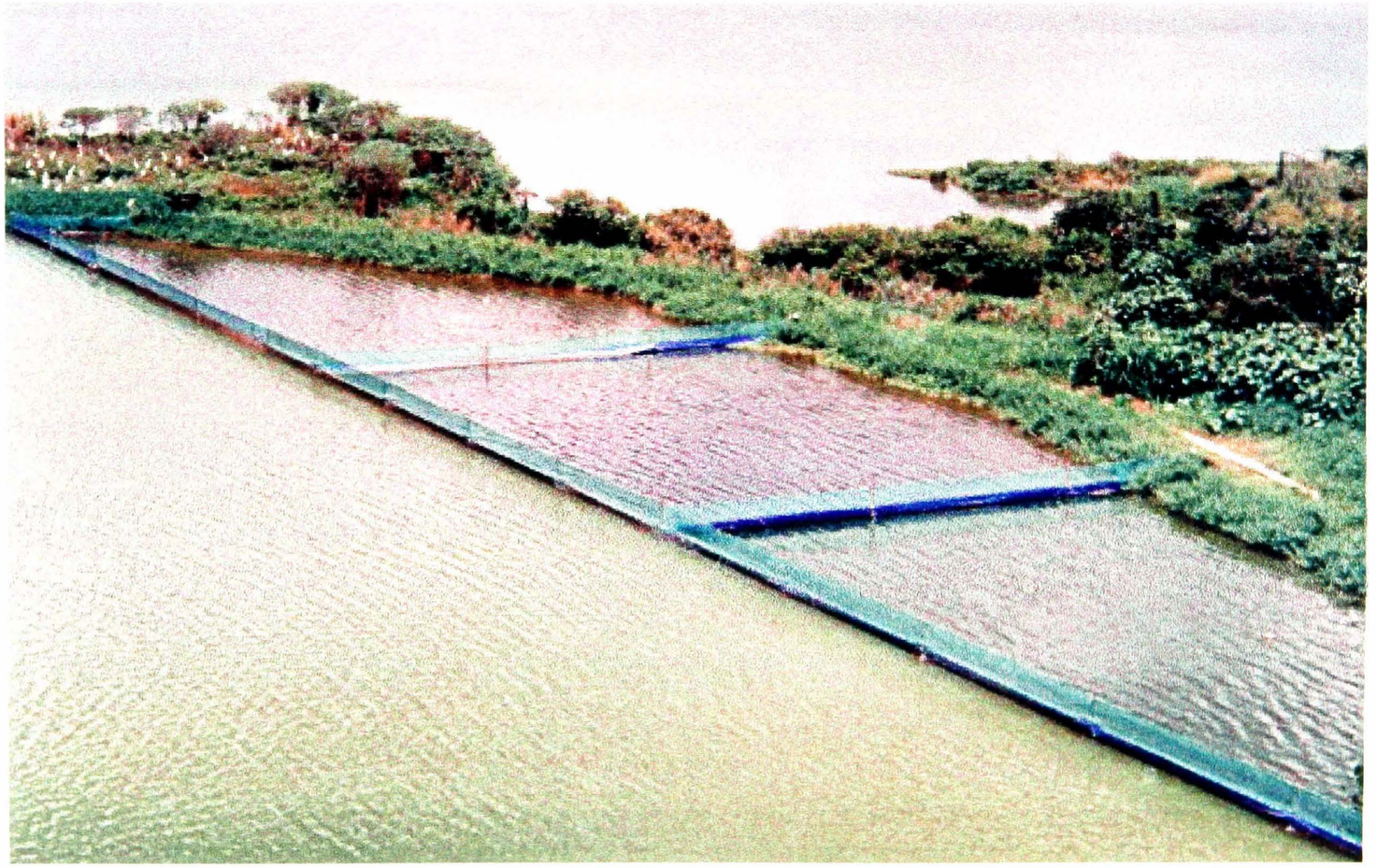


Figure 2.1: Overview of the experimental sub-areas isolated by PVC-reinforced sheets nearby the Northern Sewage Plant in the Bananal Branch, Lago Paranoá.

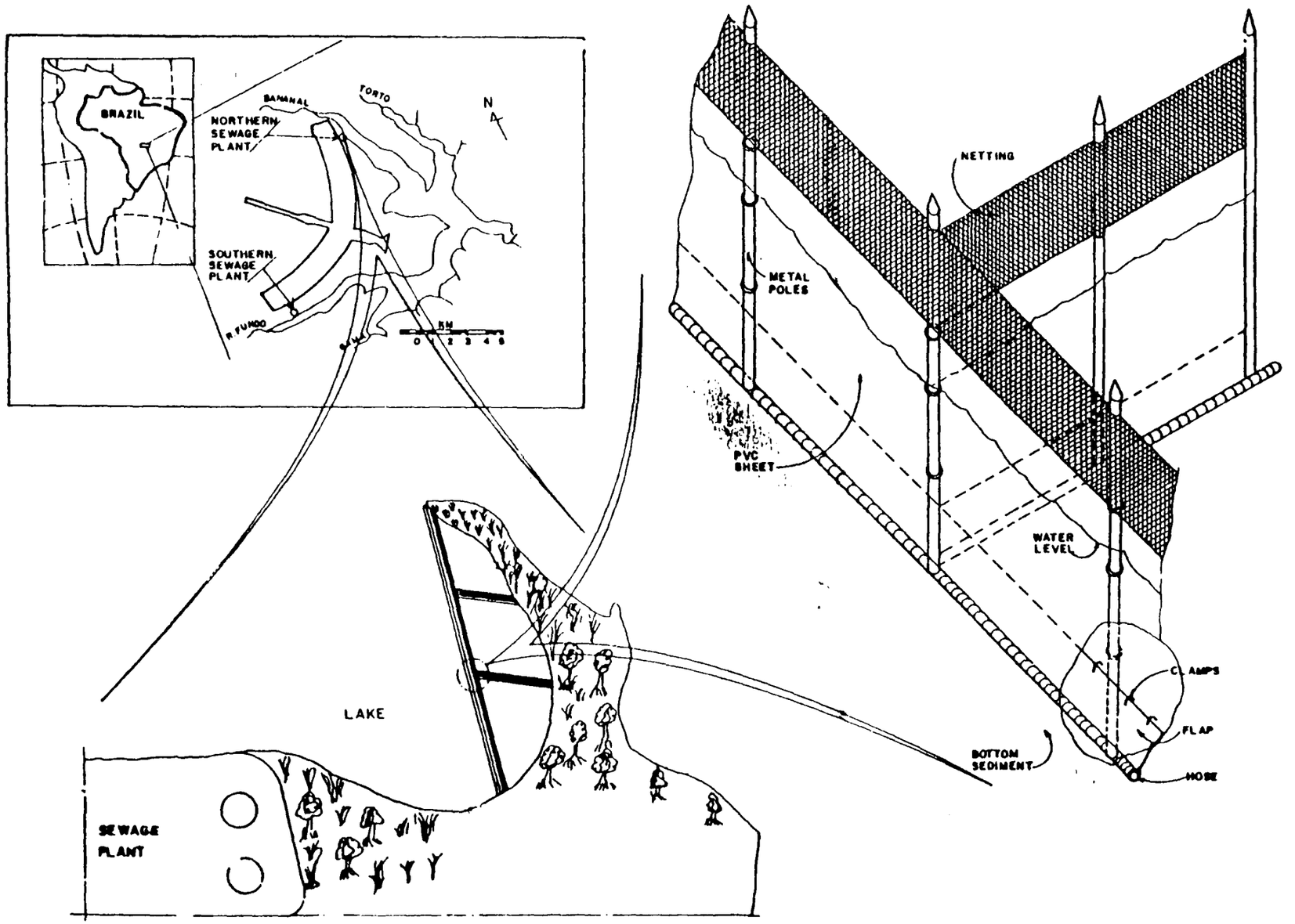


Figure 2.2: Details of the construction of the PVC-reinforced sheets barriers to isolate large littoral areas within a hypertrophic bay in Bananal Branch, Lago Paranoá.

enumerated using Sedwich-Rafter cells) counts (APHA, 1985). Verification of the complete isolation of each area from the adjacent reservoir were performed by scuba diving twice a month.

The experiment lasted 113 days, after which all fishes were recovered from each area using a combination of seines, cast-nets, gill-nets, electrofishing, rotenone and bayluscide (chemical commonly used to control snails, Marking, 1992). Repeated-measures ANOVAs without replication (mixed model) were applied on water quality variables to test differences between H.B. and L.B. areas, using MINITAB Statistical Package. Because of no replication and low statistical power, a probability level of alpha < 0.10 was used to detect significant differences in responses between TRIAL and CONTROL areas.

Table II.1: Initial fish biomass estimates and characteristics in experimental areas and the adjacent reservoir.

Areas	Surface (m^2)	Volume (m^3)	Mean Depth (m)	Fish biomass kg/ha (ind./ha)
H.B. (High Biomass)	834	619	0.75	547 (13,297)
L.B. (Low Biomass)	1,065	1,015	0.95	206 (8,500
LAKE (opened lake)	-	-		374 (6,174)

II.3. Results

The tilapia over-population in the H.B. area as compared to the L.B. area resulted in increased turbidity (P=0.001), water temperature (P=0.029), conductivity (P=0.002), pH (P=0.071), dissolved oxygen (0.071), nitrate (P<0.001), ammonia (P<0.001), TKN (P<0.001), total phosphorus (P=0.001), chlorophyll-a (P=0.001) and *Microcystis aeruginosa* abundance (P=0.096), but in reduced Secchi depth (0.023), *Cylindrospermopsis raciborskii* abundance (P=0.001) and total zooplankton (P=0.006), *Anuraeopsis sp.* (P=0.001) and *Bosmina spp.* (P=0.007) densities (Figs. 2.3 to 2.5).

Important initial differences between L. B. and H.B. areas were detected only for conductivity and nutrients (mainly nitrogen). These might have resulted from a differential effect of bottom disturbance due to seining. Indeed, bottom sediment was richer in TKN nitrogen (P=0.003), total phosphorus (P=0.020) and % of organic matter (P<0.001) in the H.B. area (621 p.p.m., 2,004 p.p.m. and 2 %, respectively) than in the L.B. area (253 p.p.m., 415 p.p.m and 0.5 %, respectively) at the beginning of the experiment (Paired t-test, N=5).

L.B. and LAKE areas were similar in terms of physico-chemical variables. Differential evolution only occurred for the densities of the most important phytoplankton species (*C. raciborskii* and *M. aeruginosa*) and zooplankton. This illustrate an artifact effect which resulted from lower turbulence and longer residence time within the isolated areas as compared to the open reservoir, providing advantages to *Microcystis* growth and zooplankton maintenance.

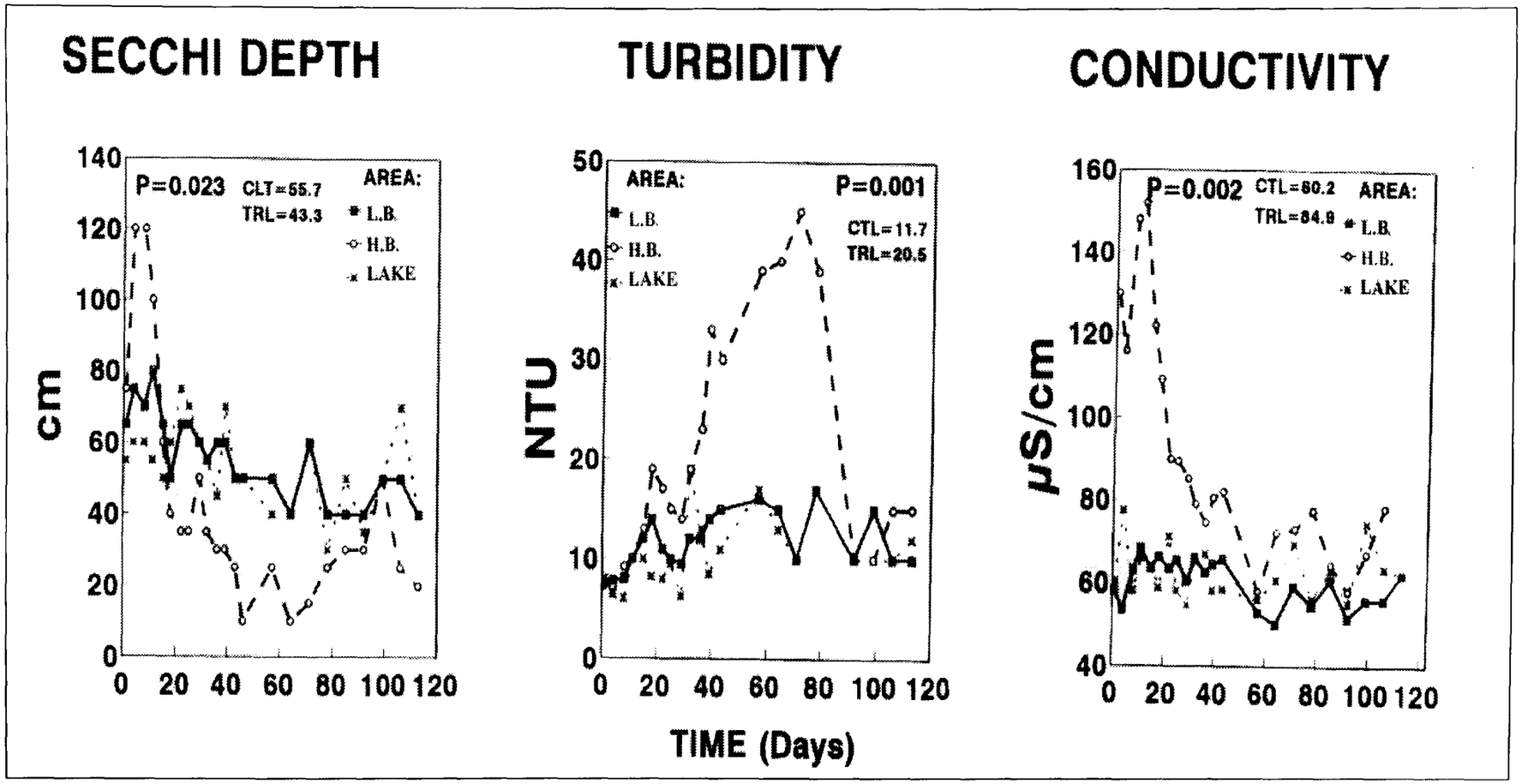


Figure 2.3. Evolution of main physico-chemical variables during the experiment for High Biomass (H.B.), Low Biomass (L.B.) and adjacent reservoir (LAKE) areas. Probability values (P) from repeated-measures ANOVA (mixed model) as well as overall mean values for High and Low Biomass areas are given at the top of each graph.

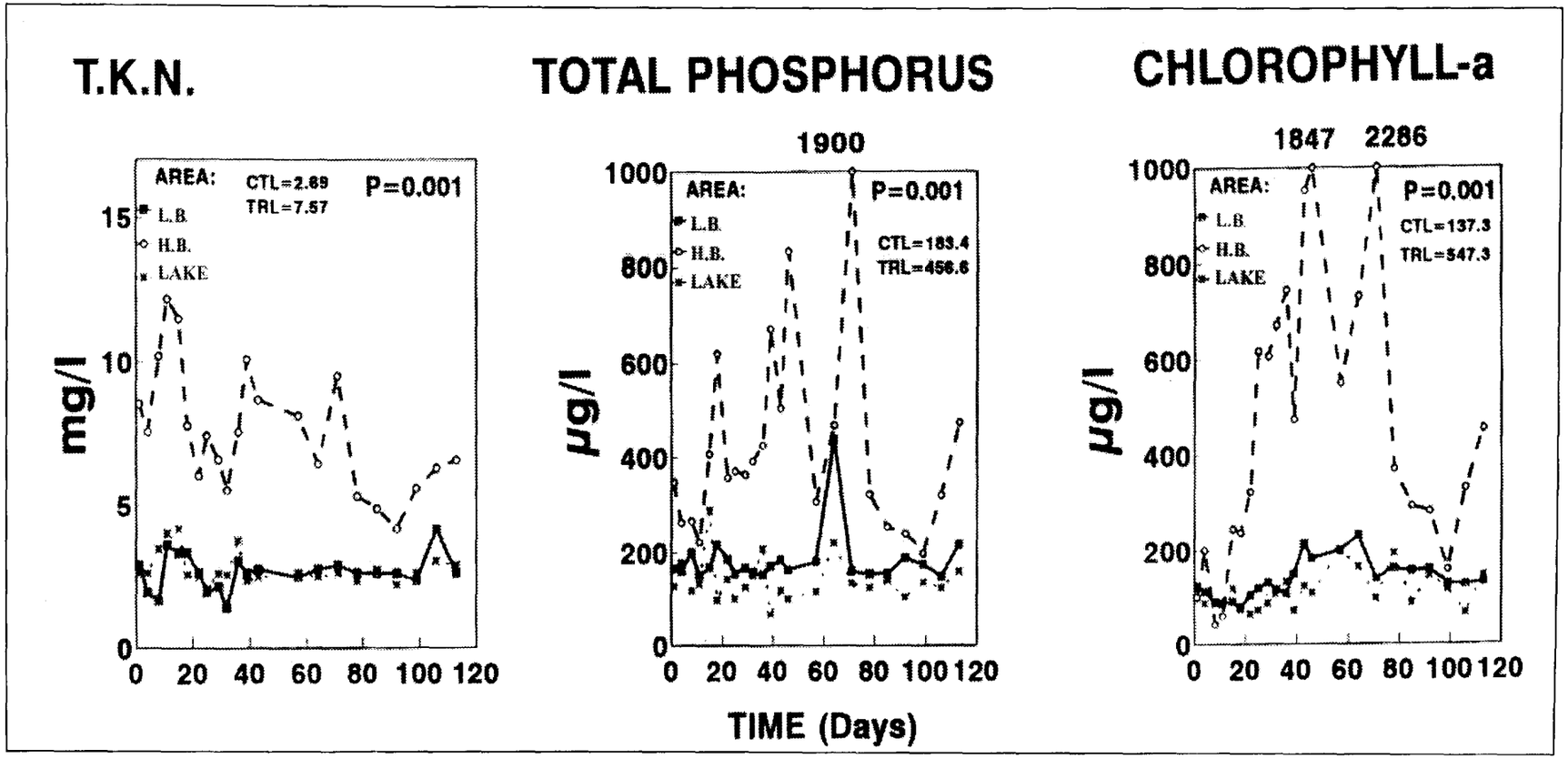


Figure 2.4. Evolution of nutrients (total Kjeldahl nitrogen and total phosphorus) and chlorophyll-a during the experiment for High Biomass (H.B.), Low Biomass (L.B.) and adjacent reservoir (LAKE) areas. Probability values and overall mean values as in Figure 2.3.

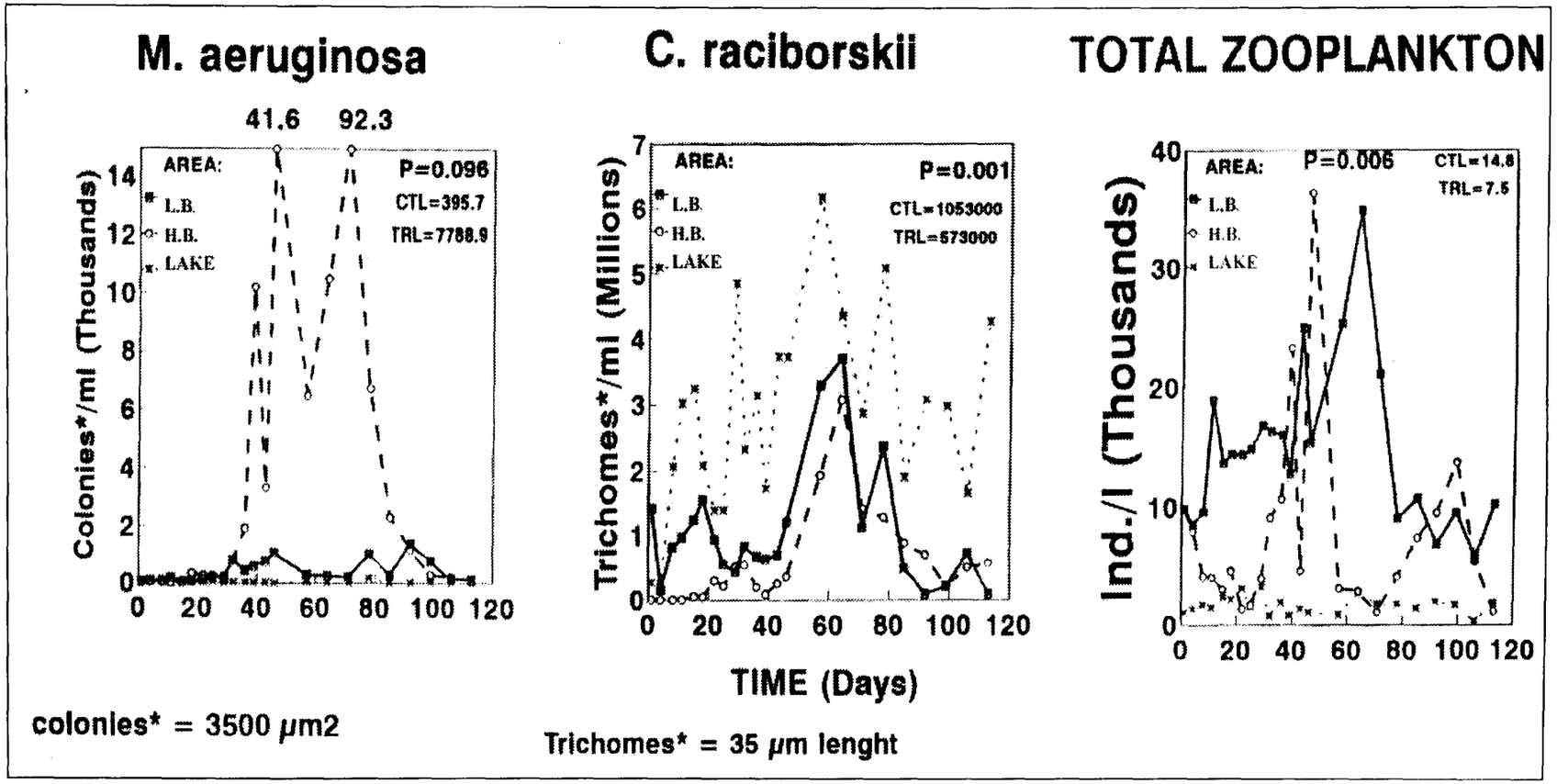


Figure 2.5: Evolution of dominant phytoplankton species and total zooplankton densities during the experiment for High Biomass (H.B.), Low Biomass (L.B.) and adjacent reservoir (LAKE) areas. Probability values and overall mean values as in Figure 2.3.

Fish biomasses recovered from each area at the end of the experiment were at least twice that initially estimated (Table II.2). Numerous fishes belonging to species other than tilapia remained in the areas during the course of the experiment because they escaped the initial sampling. Although densities and biomasses of each fish species captured in L.B. and H.B. areas were different, the relative abundance of tilapia and piscivorous "traíra", *Hoplias malabaricus*, were similar, averaging 60% and 20% of total fish abundance in each area, respectively.

Table II.2: Final fish numbers and biomass captured in the L.B. and H.B. areas.

Species	Final number (Biomass in g)	
	L.B. area	**H.B. area**
T. rendalli	204 (15,502)	837 (34,928)
O. niloticus	103 (8,459)	270 (18,128)
C. carpio	39 (9,760)	30 (6,893)
H. malabaricus	27 (10,432)	41 (15,445)
A. portalegrensis	35 (1,271)	180 (8,839)
C. callychthys	17 (1,567)	68 (7,118)
L. macrochira	20 (110)	0
C. elegans	1 (28)	4 (151)
A. bimaculatus	4 (14)	77 (430)
G. carapo	0	2 (189)
Total:		
Number	450	1,509
Weight (in g)	39,808	92,121
kg/ha	403	1,105
g/m^3	42	149
Fish/ha	4,225	18,094

II.4. Discussion

The lower fish biomass in L.B. area was directly associated with increased Secchi depth and zooplankton abundance, and lower nutrient concentrations (N and P) and phytoplankton biomass, compared with the H.B. area. Moreover, tilapia over-population established in the H.B. area promoted a severe bloom of *M. aeruginosa* out-competing the typical dominance of cyanobacteria *C. raciborskii* (L.B. and LAKE areas).

At least two mechanisms may explain the enhanced algal growth in presence of higher fish biomass: tilapia may have indirectly favoured phytoplankton by (1) preying upon zooplankton, and consequently reducing the grazing pressure, and/or (2) supplying nutrients via excretion and/or physical disturbance of bottom sediment. Given the weakness of the zooplankton-phytoplankton link in such an ecosystem dominated by small herbivorous zooplankton (mainly rotifers) and unpalatable and inedible cyanobacteria (Pinto-Coelho, 1983), it seems more plausible that tilapia may have supplied nutrients to algal growth.

Bottom sediment from the H.B. area was richer in both organic matter and nutrients (N and P) than the L.B. area. Therefore, the bottom-feeding habits of tilapia may have added "new" nutrients (otherwise trapped in the sediment) to the water column at highest rates in presence of enhanced fish biomass. Indeed, sediment comprises 50% of the food amount (on a volumetric basis) consumed by either *Tilapia rendalli* or *Oreochromis niloticus* in Lago Paranoá (Grando, 1989). Furthermore, tilapias are omnivores in Lago Paranoá (Grando, *op.cit.*). By consuming various types of food, such as zooplankton, phytoplankton, macrophytes, benthic invertebrates, fish larvae and eggs, in addition to organic matter from the sediment, tilapia may have increased the availability of phosphorus in the water column, which

otherwise, would have remained immobilised in these compartments for longer period.

The supply and recycling of nutrients by fish have been suggested as mechanisms acting in experiments conducted in 'bag-type' enclosures stocked with extremely high tilapia biomass in Lago Paranoá (3,000 kg/ha; Starling & Rocha, 1990). Indeed, some recent studies conducted in lacustrine ecosystems have demonstrated that fish play an important role in providing nutrients and/or accelerating their recycling rate (Vanni & Findlay, 1990; Brabrand *et al.*, 1990; Kraft, 1992; Schindler *et al.*, 1993).

Estimates of fish P release rates

Although the rate of phosphorus excretion by tilapia in Lago Paranoá has not yet been determined, the use of published data makes it possible to get an approximation of the total amount of PO_4-P supplied by the tilapia community enclosed in the experimental areas. According to Brabrand *et al.* (1990), because of differences due to feeding habits, age and species, excretion rates of orthophosphate by fish vary between 2 and 8 µg P/g ww/h (at 22 °C, fish > 2.8 g). Table II.3 shows estimates of fish P release rates using an average value of 5 µg P/g ww/h on a 24 hs basis. The calculated daily fish release rates (4.71 µg PO_4P/L for L.B. area and 17.80 µg PO_4P/L for H.B. area) account for a considerable fraction of the orthophosphate present in the water column as overall mean concentrations of orthophosphate in the water were 13.33 µg PO_4P/L and 21.61 PO_4P/L, for L.B. and H.B. areas, respectively.

Table II.3: Fish characteristics in the L.B. and H.B. areas, and estimates of phosphorus release (as orthophosphate or soluble reactive phosphorus, SRP) by the overall fish community and tilapia, using Brabrand et al.'s (1990) data on < 10 g roach (*Rutilus rutilus)* at 17 °C, (*) assuming an average release rate of 5 µg PO_4P/g wet weight/h (i.e., 120 µg PO_4P/g ww/d).

Parameters	L.B.	H.B.
Area (m^2)	1,065	834
Volume (m^3)	1,015	619
Overall fish biomass (kg ww)	39.81	92.12
Tilapia biomass (kg ww)	23.96	53.06
Overall mean fish body weight (g ww)	88.46	61.05
Mean tilapia body weight (g ww)	78.05	47.93
Estimate of overall fish P release * (µg SRP/d)	4.78	11.05
Estimate of overall fish P release * (µg SRP/l/d)	4.71	17.8

Relative importance of fish P release rates and external P load.

To evaluate the relative contribution of fish in the P balance of Lago Paranoá, estimates of external P load to the branch of the reservoir in which the experiment has been performed (Bananal Branch) and P supplied by the overall fish population in that area can be compared:

(a) Estimate of P release by the overall fish population in Bananal Branch:

Fish biomass = 400 Kg/ha (estimate from captures in the L.B. area).

Overall fish biomass in Bananal Branch = 400 Kg/ha x 973 ha = 389.2 10^6 g

Mean specific P excretion rate = 5 µg PO_4P/g ww/h or 120 µg PO_4P/g ww/d (Brabrand *et al.*, 1990).

Overall daily fish P release rate = 389.2 10^6 g x 120 µg PO_4P/g ww/d = **46.7 kg PO_4P/d**

(b) Watershed P load to Bananal Branch (CAESB data):

Mean P load before tertiary sewage treatment = 135 Kg TP/d or **13.5 kg PO_4P /d**

Mean P load in 1994, after tertiary sewage treatment = 58 Kg TP/d or **5.8 kg PO_4P/d**

Therefore, for the Bananal Branch, the fish contribution to P cycling is more important than the external P load from the watershed. Indeed, in terms of PO_4P, the estimated P release by the overall fish population is 3.5 and 8.0 times higher than the external P load before and after the implementation of the tertiary sewage treatment by CAESB, respectively. Despite the drastic 60% reduction in P load to this ecosystem from 418 kg TP/d in 1992 to 169 kg TP/d in 1994, no significant decline in both TP and chlorophyll-a concentrations in the water column has been observed so far (Altafin *et al.*, 1995). These results demonstrate the need for controlling the internal load of phosphorus represented by tilapia over-population and/or nutrient enriched bottom sediment in Lago Paranoá, as a supplementary measure to the decrease in nutrient input (mainly P) to the Reservoir by the new sewage plants.

A more accurated picture on the reservoir fish contribution to the total P budget will be given in Chapter VI by crossing measurements of tilapia excretion rates with estimates of their biomass in littoral areas of Lago Paranoá

II.5. Conclusions

- The hypertrophic littoral zone of the Bananal Branch in Lago Paranoá holds a dense fish population, highly dominated by tilapia (*Tilapia rendalli* and *Oreochromis niloticus).*
- Tilapia over-population in such shallow areas promotes the development of eutrophication symptoms such as low transparency and high phosphorus and chlorophyll-a levels.
- Blooms of nuisance cyanobacteria (*Microcystis aruginosa*) in shallow hypertrophic areas of Lago Paranoá may be prevented by controlling additional nutrient supply mediated by excessive tilapia population.
- Tilapia feeding on detritus from sediment enhances phytoplankton biomass by supplying additional nutrients to the water column from a "new" source via excretion and disturbance of nutrient-enriched bottom sediment.
- The phosphorus contribution from fish excretion was estimated to be at least as important as the current external nutrient load to the Bananal Branch after the implementation of Tertiary Sewage Treatment.
- Reduction of internal nutrient loadings by fish excretion and/or sediment release represents a key argument to speed up the undergoing restoration of Lago Paranoá.

CHAPTER III

SILVER CARP CAGE CULTURE

III.1. Introduction

In recent years, the world production of carps (including both Chinese and common carp) has reached 6 million metric tons, representing nearly 60% of the freshwater finfish production. Silver carp production accounts for one third of the total carp production rates and around 20% of the global finfish aquaculture production (FAO, 1996). Examples of silver carp production rates as high as 4000 kg/ha/year have been reported as this fish species utilizes food resource (phytoplankton) at the base of the food chain (Liang *et al.*, 1981).

Extensive cage culture in eutrophic waters using a fish species that feeds at the bottom of the food chain, such as silver carp, has the greatest potential for low-cost fish production (Little & Muir, 1987). Although fish culture in floating cages has been adopted in Asia since the end of the last century (Beveridge, 1996), its use in lakes and reservoirs has only become increasingly popular in recent decades (Sifa & Senlin, 1995). In addition to rearing fish from juvenile to marketable size, extensive cage culture of planktivorous Chinese carps has been targeted at the production of large-sized fingerlings prior to stocking in lakes and reservoirs, in order to increase fish survivorship (Sifa & Senlin, 1995) and for cropping excessive phytoplankton biomass and reducing the incidence of nuisance algal blooms (Ling, 1982).

Despite the potential of converting high primary production of nutrient-enriched ecosystems such as Lago Paranoá into edible fish biomass for human consumption, a number of possible environmental constraints to cage aquaculture, such as low

oxygen levels, presence of contaminants and high biomass of unpalatable and/or toxic cyanobacteria may become important as eutrophication proceeds. The frequent occurrence of nuisance *Microcystis* blooms and fish mortality in hypertrophic areas of Lago Paranoá provides a background of uncertainty concerning the acclimatization of silver carp to this ecosystem.

The main purpose of this chapter is to evaluate the adaptation and performance of silver carp to the hypertrophic conditions of Lago Paranoá by measuring fish growth and mortality rates when maintained for long periods in net-cages without supplementary feeding. Specific objectives include the consumption of colonial bloom-forming algae, the evaluation of the effects of fish stocking density on growth and the environmental impacts of cage culture on the vicinity of the experimental site. The study specifically addresses the following questions:

i) Will silver carp adapt to reservoir eutrophic conditions, displaying low mortality rates when kept in net-cages without supplementary food?

ii) Do silver carp graze on dominant nuisance cyanobacteria in the reservoir (mainly colonial *Microcystis aeruginosa*)?

iii) Will silver carp grow "satisfactorily" when confined in net-cages feeding exclusively on natural plankton from Lago Paranoá?

iv) Will there be differences in silver carp growth rates as a consequence of contrasting stocking rates used in net-cages?

v) Will the water quality in the vicinity of the experimental area be affected by net-cage aquaculture?

III.2. Material and Methods

In order to address the above questions, two 6-month (experiment I with juvenile silver carp, and experiment II with adult silver carp) and one 3-month (experiment III with fingerling silver carp) net-cage experiments were performed. Four floating net-cages (2.5 m x 2.5 m, 1.7 m depth, 6.25 m^2 in surface area, 10 m^3 in effective volume, 12 mm mesh opening for experiments I and II and 8 mm mesh for experiment III) were secured to a pier located in the vicinity of the Northern Sewage Treatment Plan, one of the most eutrophic areas of Lago Paranoá (Figures 3.1 to 3.3).

Two replicated treatments: low biomass (L.B., 400 $g.m^{-3}$) and high biomass (H.B., 800 $g.m^{-3}$) were assigned to cages using one-year-old juvenile silver carps (300 g average weight, 25 cm average standard length) in experiment I, two-years-old silver carps (1100 g average weight, 40 cm average standard length) in experiment II and young-of-the-year silver carps (72 g average weight) in experiment III, obtained from a private fish farm. Based on a literature review, low and high silver carp stocking densities were set at 400 g/m^3 and 800 g/m^3 respectively. Each cage in the high biomass treatment was stocked with 26 fishes during the first trial, 8 fishes during the second, and 81 fishes during the third trial, while each cage from low fish biomass treatment received 12 fishes in the first experiment, 4 fishes in the second, and 40 fishes in the third experiment.

Experiments I and II lasted 6 months each, the first from 20 July 1995 until 19 January 1996, covering mainly the rainy season, while the second instead from 17 April 1996 until 16 October 1996, covering mainly the dry season. Experiment III also covered the wet season as it was performed from 23 December 1996 until 24 March 1997 and lasted 3 months. During experiments I and II, all fishes from each

cage were recovered every two months in order to obtain individual weights. Due to the shorter term of experiment III, only initial and final fish weights were recorded.

Limnological conditions inside cages as well as in the adjacent reservoir water were evaluated at monthly intervals in experiment I, and every 15 days in experiments II and III. Surface water samples were analyzed for water temperature, dissolved oxygen, conductivity, pH, turbidity, total Kjeldahl nitrogen, total phosphorus, total chlorophyll-a, colonial floating algae abundance and zooplankton enumeration (APHA, 1985). Water transparency (Secchi disk) and fish mortality were also monitored.

At the end of each experiment, at least one fish from each cage (except for experiment III) was preserved for further quantitative digestive tract contents analysis using Hyslop's (1980) numeric method. After measuring the total length of the digestive tract, the contents of the anterior one third of the intestine (foregut) were removed by washing with distilled water, diluted up to 100 ml, preserved in 5% formalin and treated as a plankton sample. Sub-samples of planktonic food items (zooplankton organisms and colonial bloom-forming algae) were counted in triplicate using Sedgewick-Rafter cells observed under a compound microscope (x 100). Due to the high variability in size of colonies of bloom-forming algae (*Microcystis aeruginosa* and *Botryococcus braunii*), a standard colony size represented by the area of a square in the Kellner eyepiece (90 μm x 90 μm; 8100 μm^2) was arbitrarily established for counting purposes.

For both experiments I and II, data on fish growth rates at high and low stocking densities were statistically compared using repeated-measures ANOVA (Wilkinson, 1989). Given the low replication and statistical power, a probability level $\propto < 0.10$ was chosen to reduce the chance of committing a type II error, i.e., failing to reject a false null hypothesis.

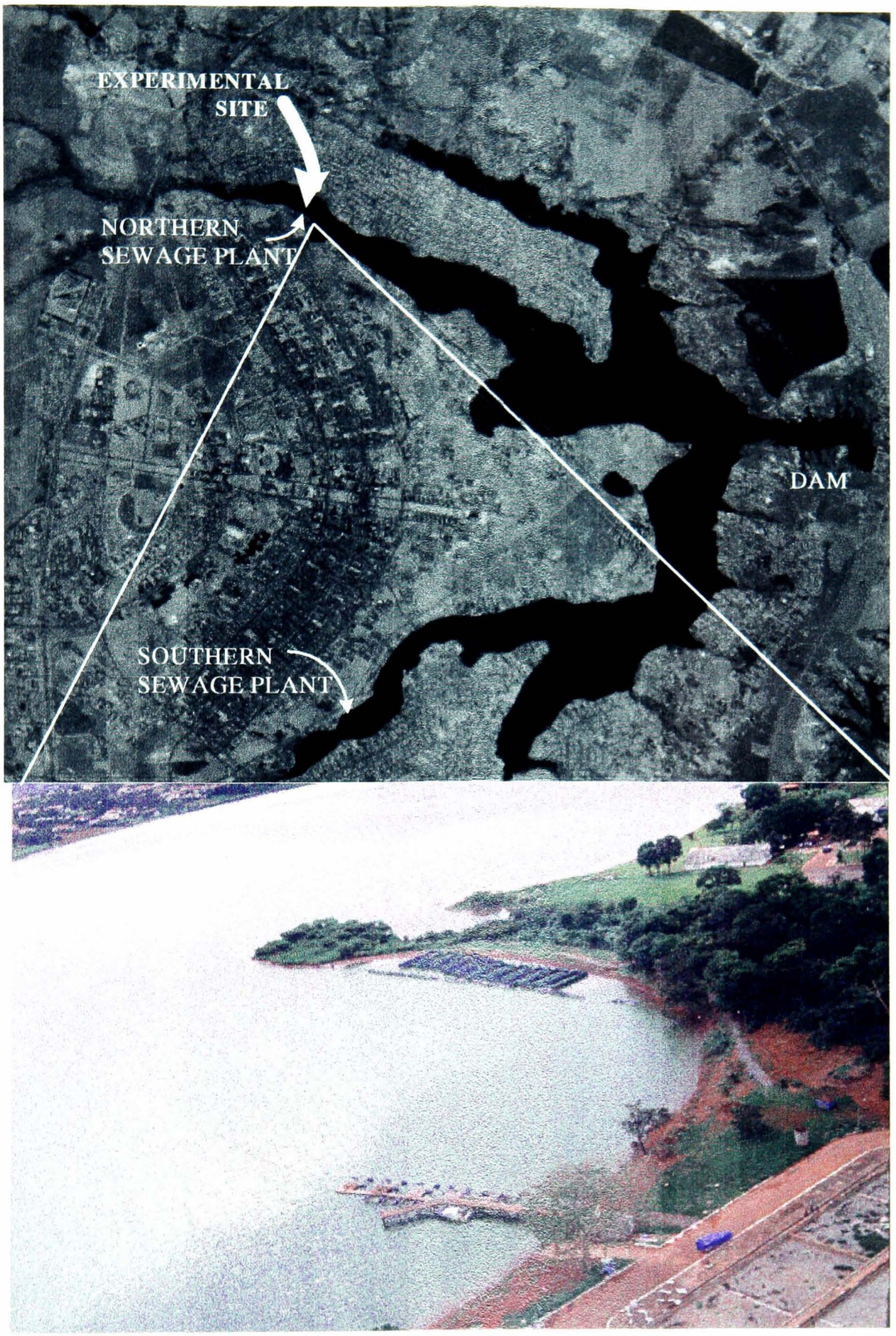


Figure 3.1: Location of the Experimental Site in the Bananal Branch, Lago Paranoá.

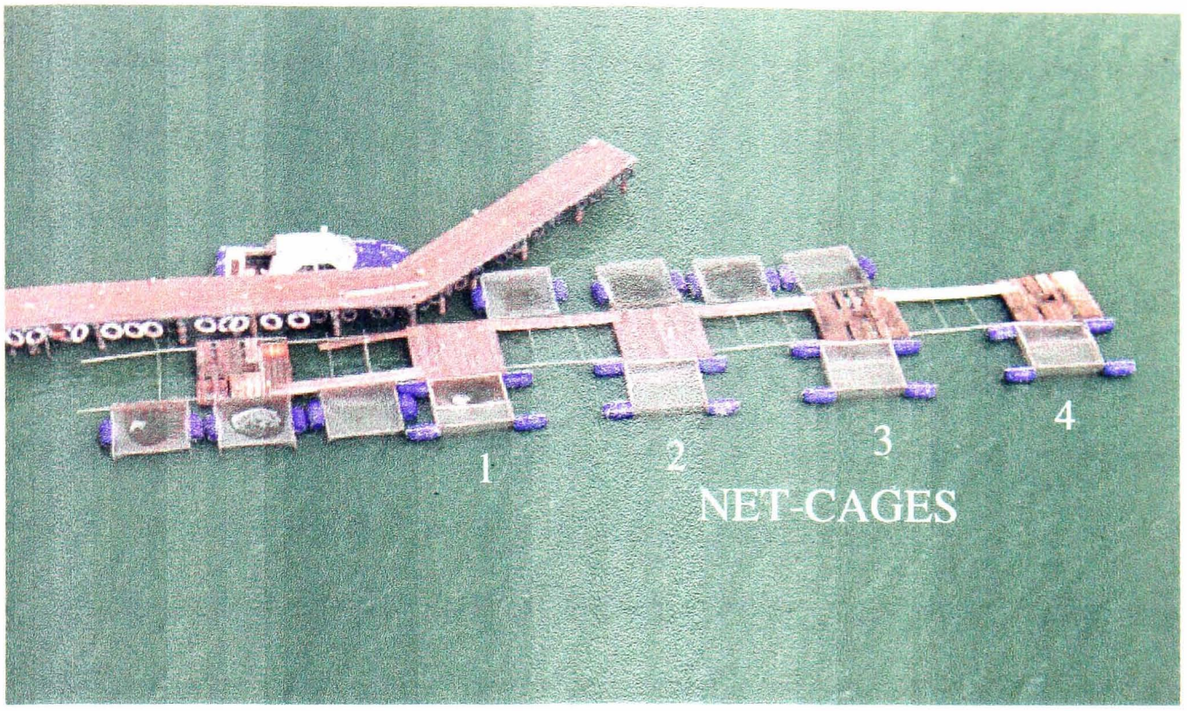


Figure 3.2: Overview of the experimental area in Lago Paranoá showing net-cages 1 to 4 in a linear array.

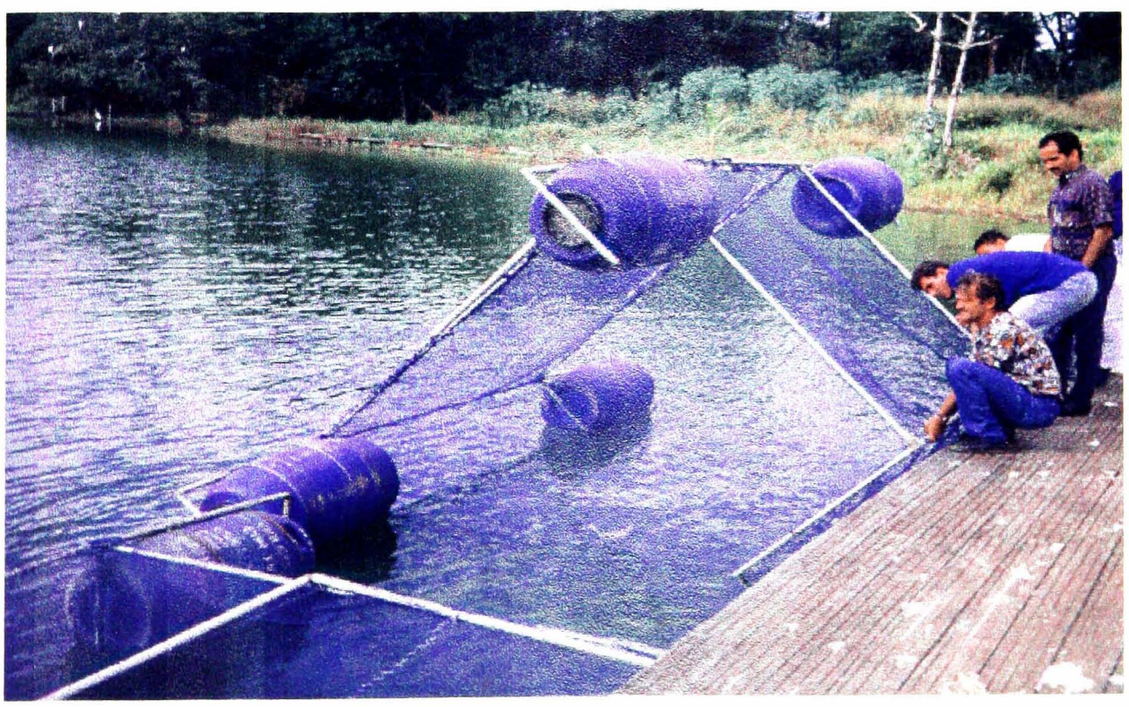


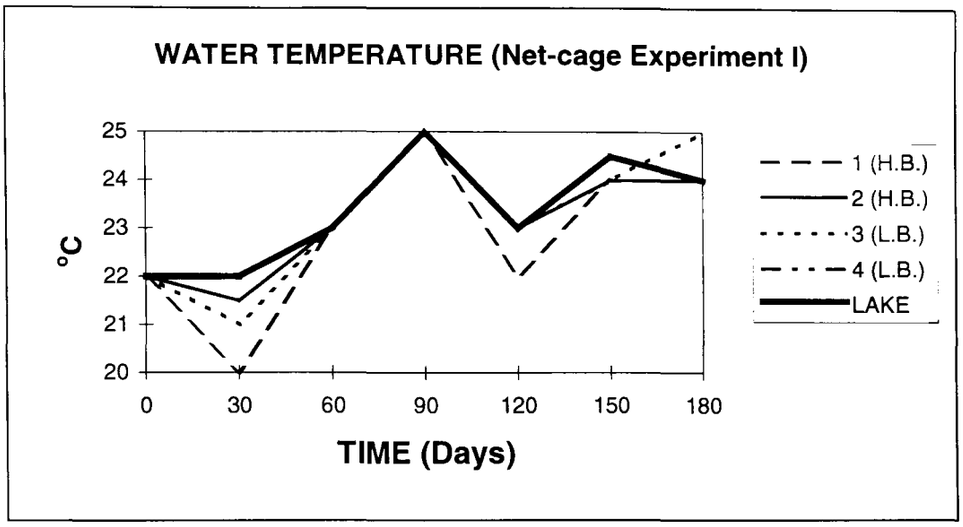
Figure 3.3: Detail of a net-cage being installed in the experimental area.

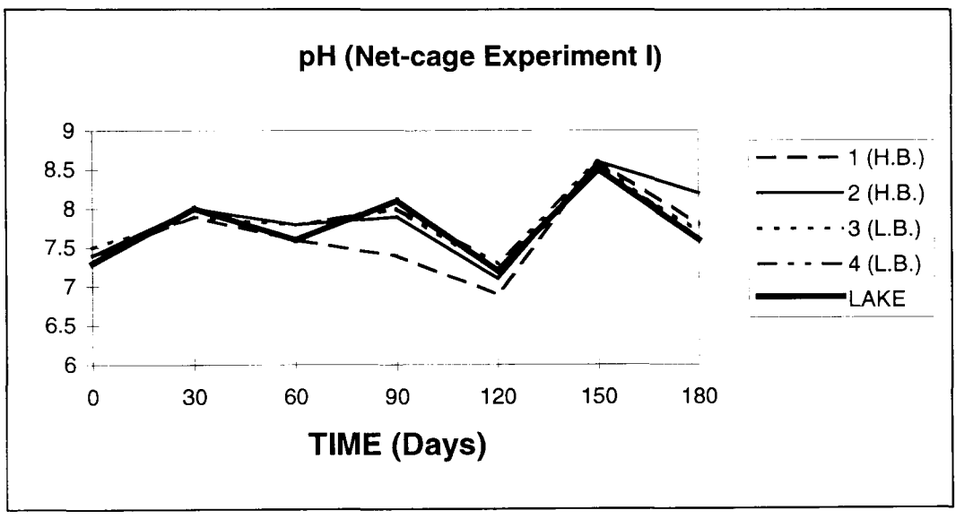
III.3. Results

The change in major water quality parameters during the course of experiment I is shown in Figure 3.4. Apart from some restrictive conditions for fish survivorship imposed by dissolved oxygen levels below 3 mg/l on day 120, all water quality variables ranged within expected limits. In general, there were no noticeable differences between the cages and the adjacent reservoir for limnological parameters throughout the period. Only zooplankton, which are patchily distributed, presented some variation between cages and the adjacent lake sampling point.

Taking these results into account, the water quality monitoring program was modified for the following two experiments: sampling was restricted to only one cage and the adjacent reservoir area but intensified by doubling the frequency of water sampling. The change in major water quality parameters throughout experiment II and III are presented in Figures 3.5 and 3.6. As in experiment I, with the exception of a drop in dissolved oxygen (down to 2.0 mg/l at 30 days of study) during experiment II, all variables ranged within expected limits. Again, there were no remarkable contrasts between cage and adjacent lake sampling points, except for the patchily distributed planktonic organisms on some sampling dates.

During experiment I, all fishes stocked in the cages gained weight. For cages 1, 2, 3, and 4, final average individual fish weights were respectively 2.25, 2.37, 2.44 and 3.00 times higher than initial stocking weight. Figure 3.7 illustrates the change in mean silver carp body weight for each cage and mean gain of weight per treatment during experiment I. A repeated-measures ANOVA showed that fish growth rate was significantly higher at low fish biomass ($P=0.099$) as compared with high biomass.





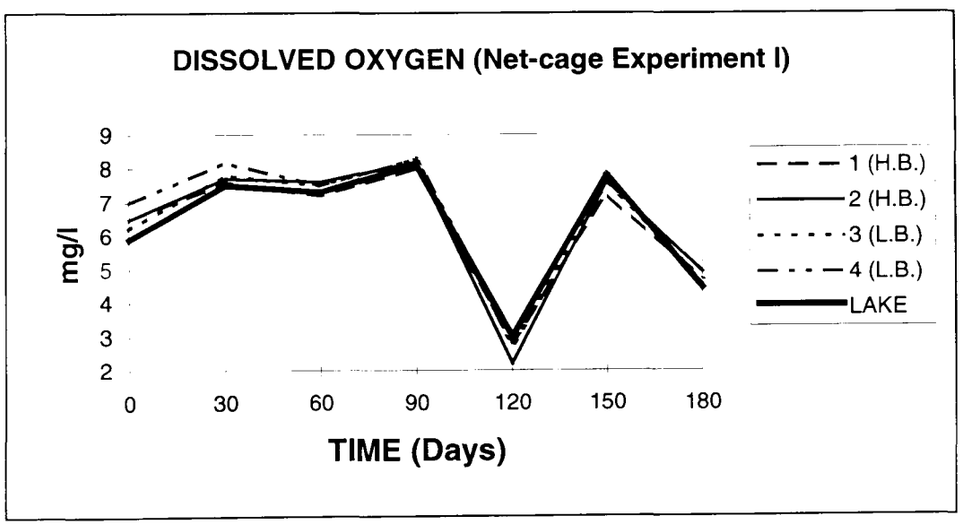
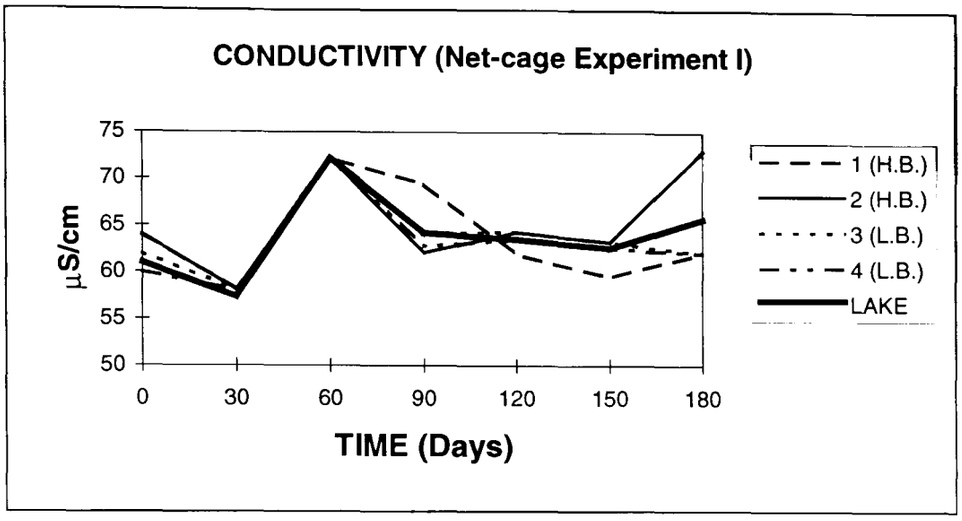
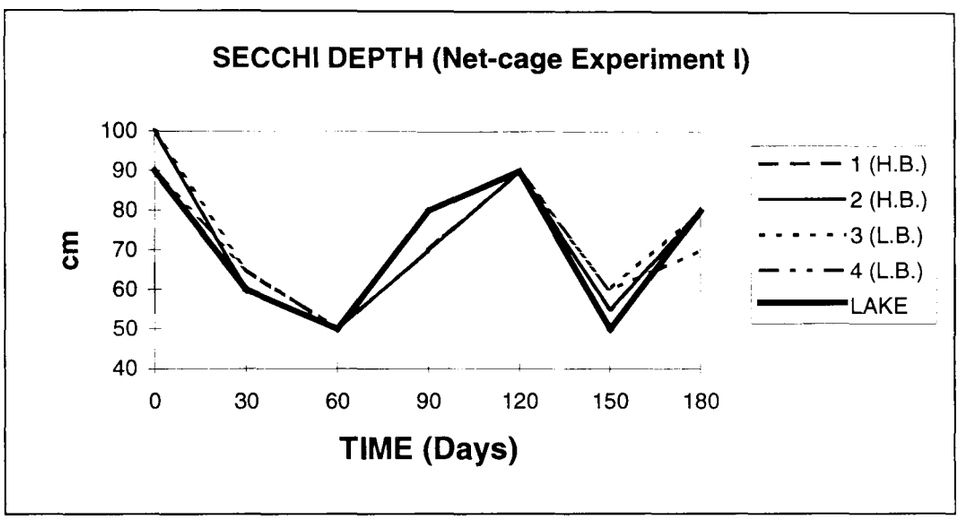


Figure 3.4: Evolution of main water quality parameters in cages (H.B.= high biomass, L.B.= low biomass) and adjacent reservoir (LAKE) during experiment I.





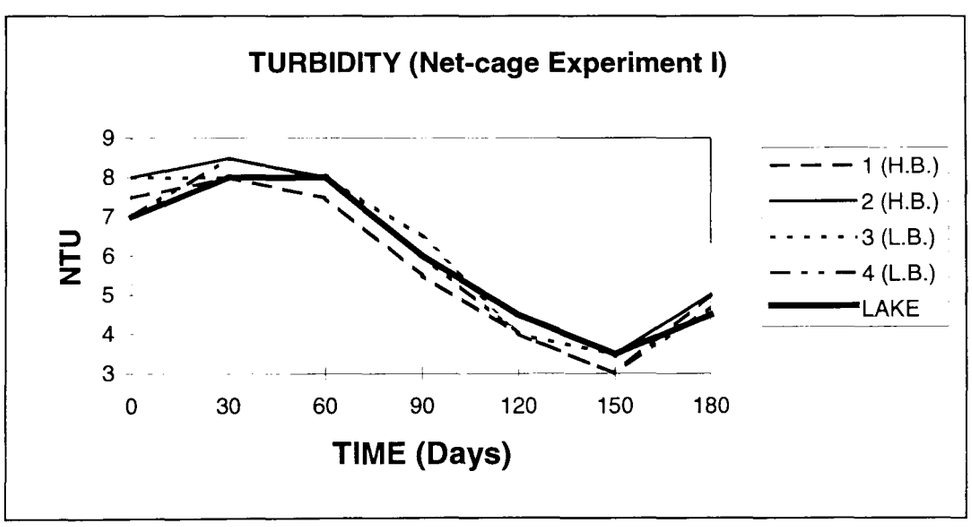
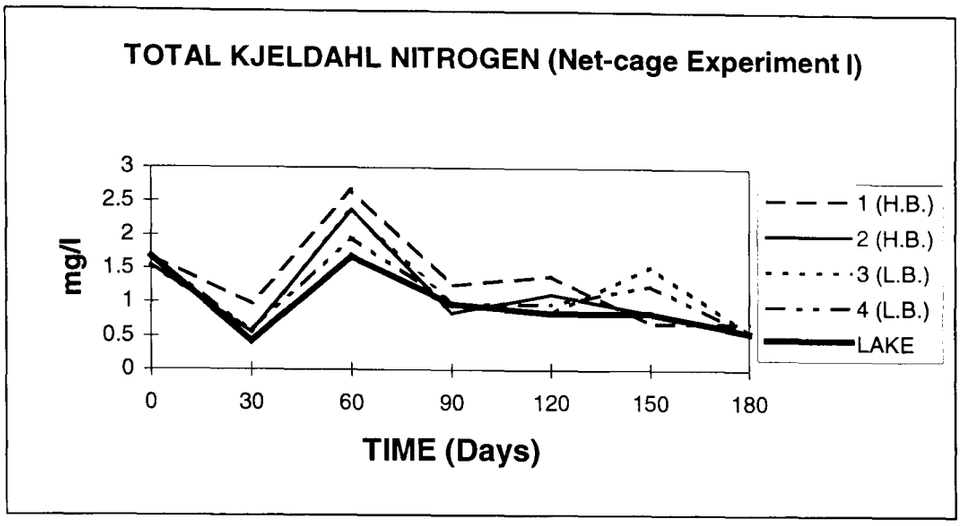
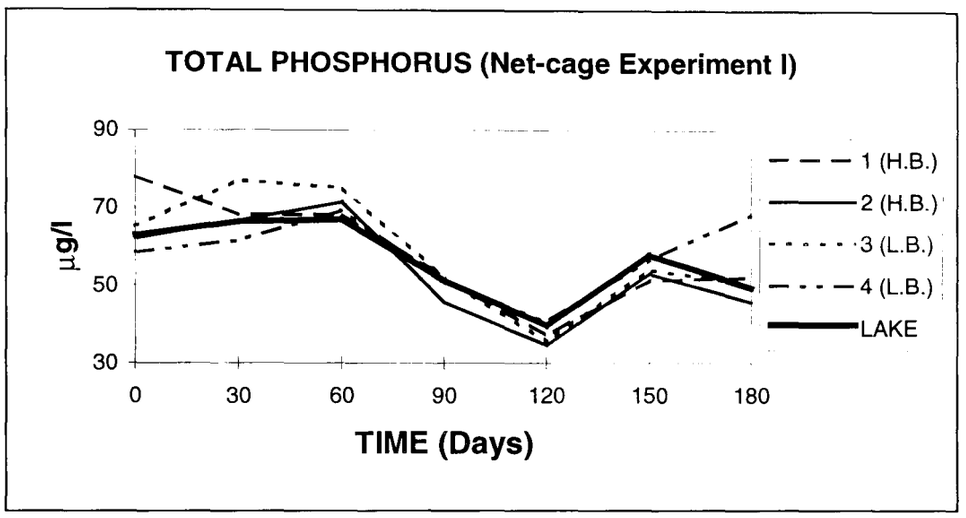


Figure 3.4: (cont.) Evolution of main water quality parameters in cages (H.B.= high biomass, L.B.= low biomass) and adjacent reservoir (LAKE) during experiment I.





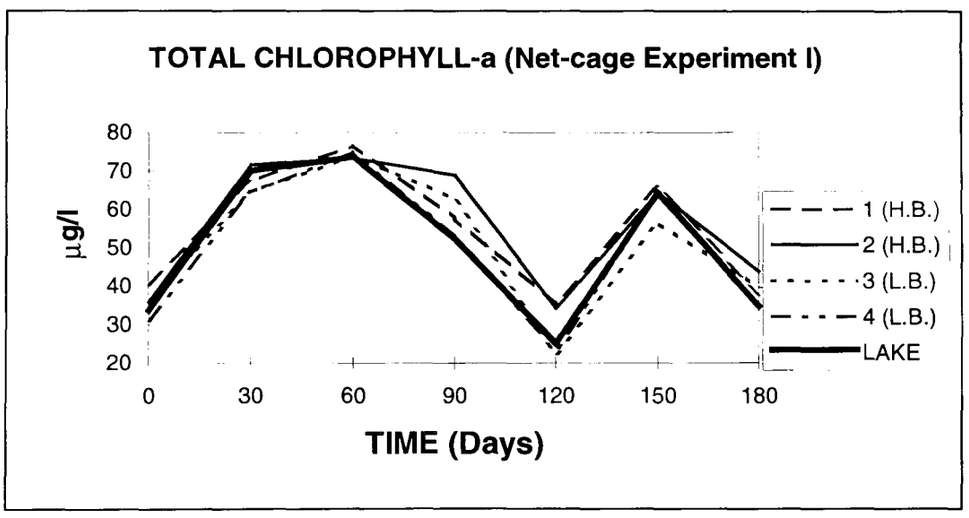
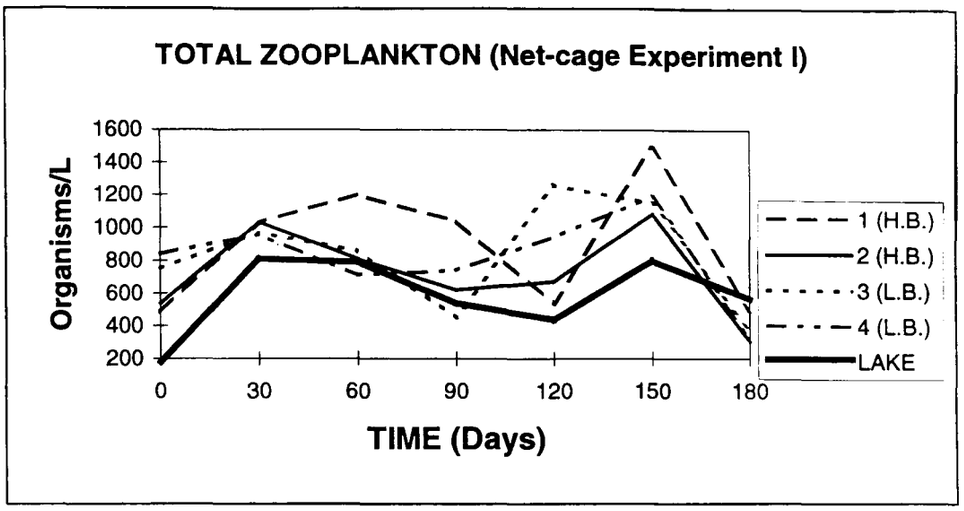


Figure 3.4: (cont.) Evolution of main water quality parameters in cages (H.B.= high biomass, L.B.= low biomass) and adjacent reservoir (LAKE) during experiment I.



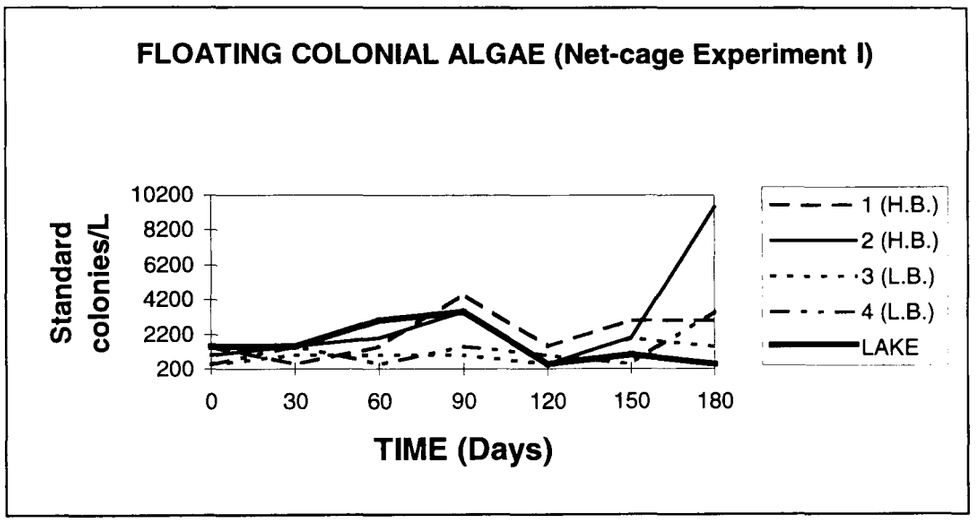
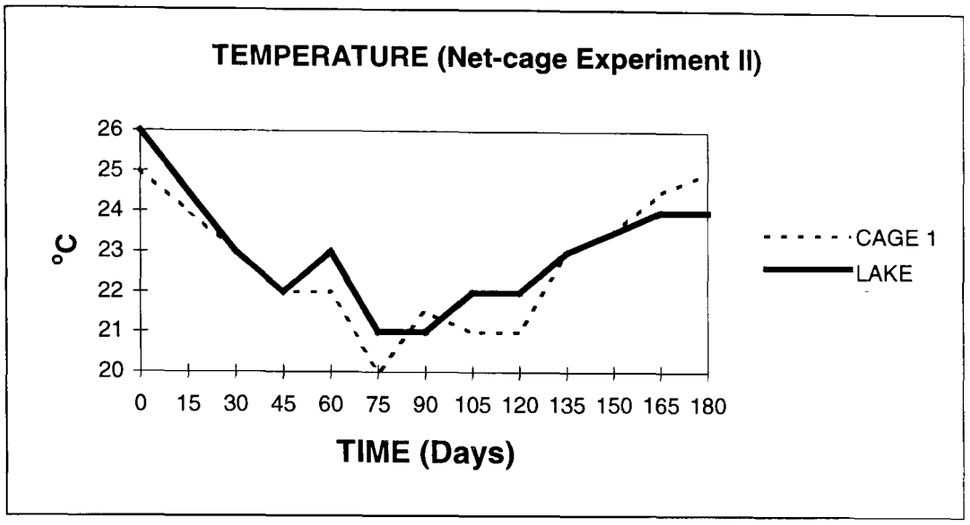
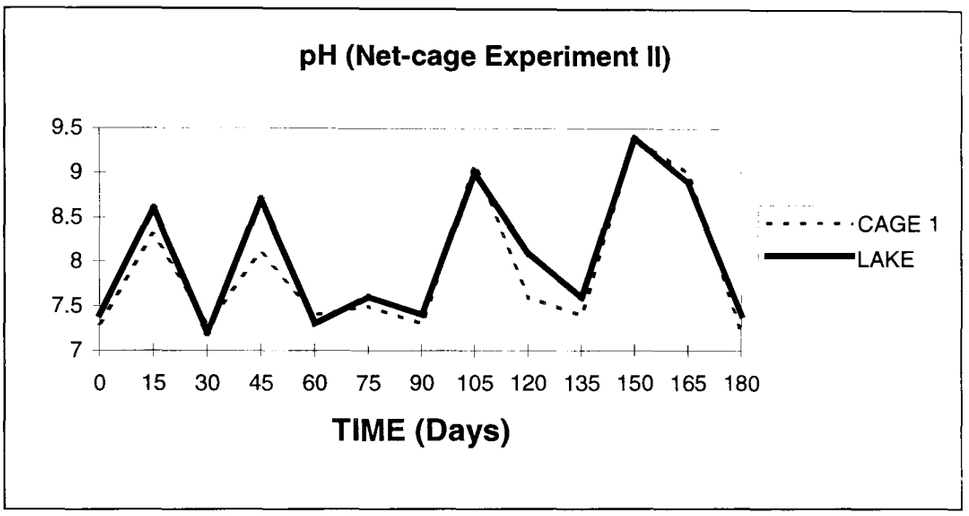


Figure 3.4: (cont.) Evolution of main water quality parameters in cages (H.B.= high biomass, L.B.= low biomass) and adjacent reservoir (LAKE) during experiment I.





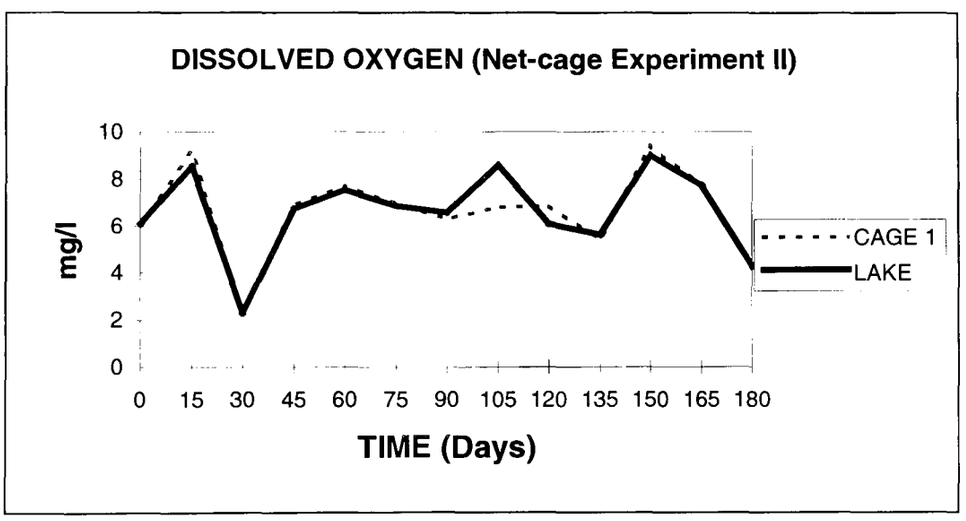
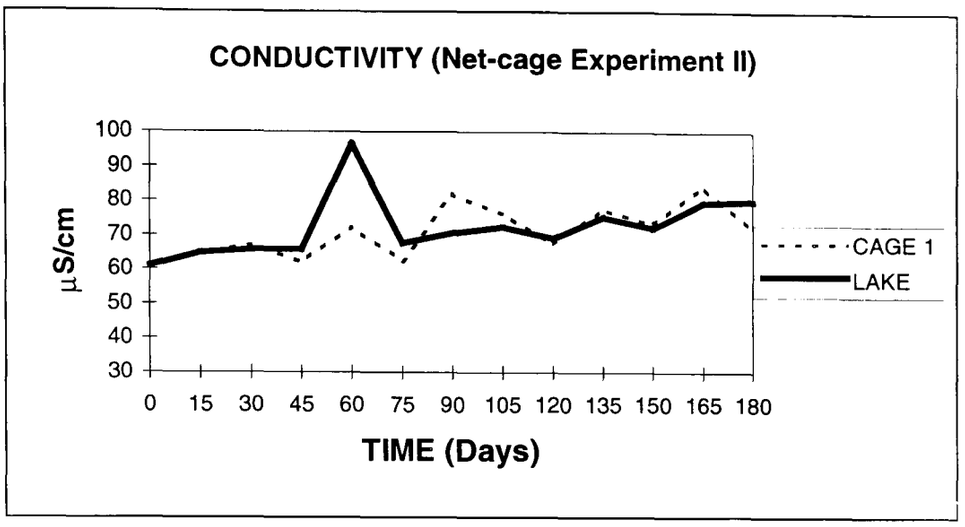
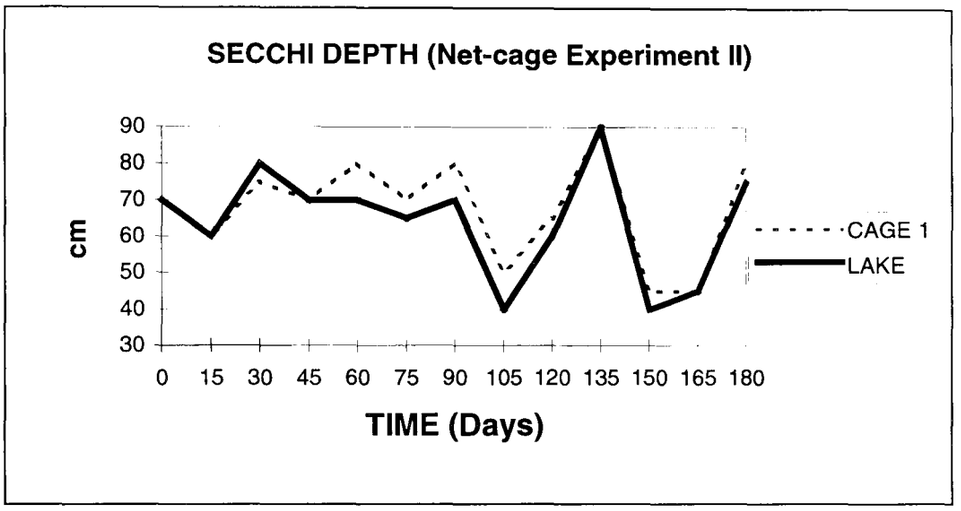


Figure 3.5: Evolution of main water quality parameters in cage1 (H.B.= high biomass, L.B.= low biomass) and adjacent reservoir (LAKE) during experiment II.





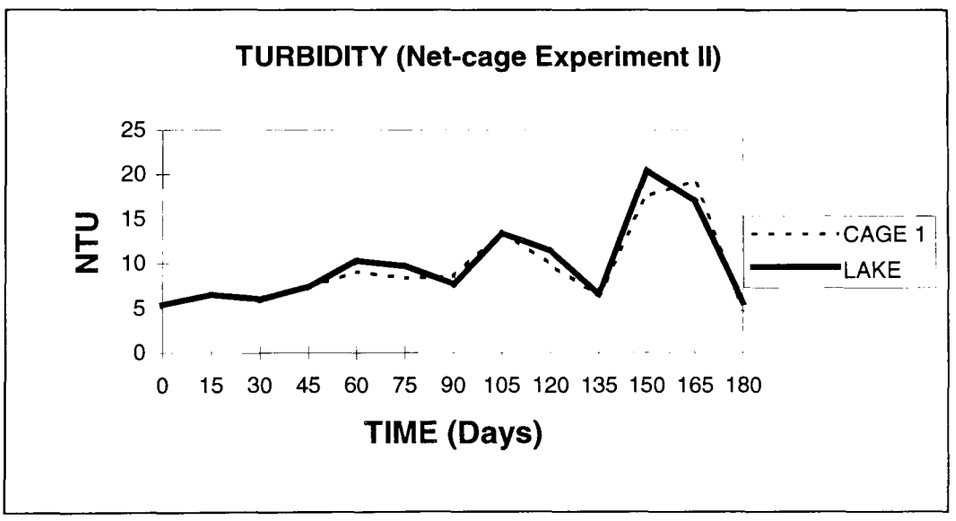
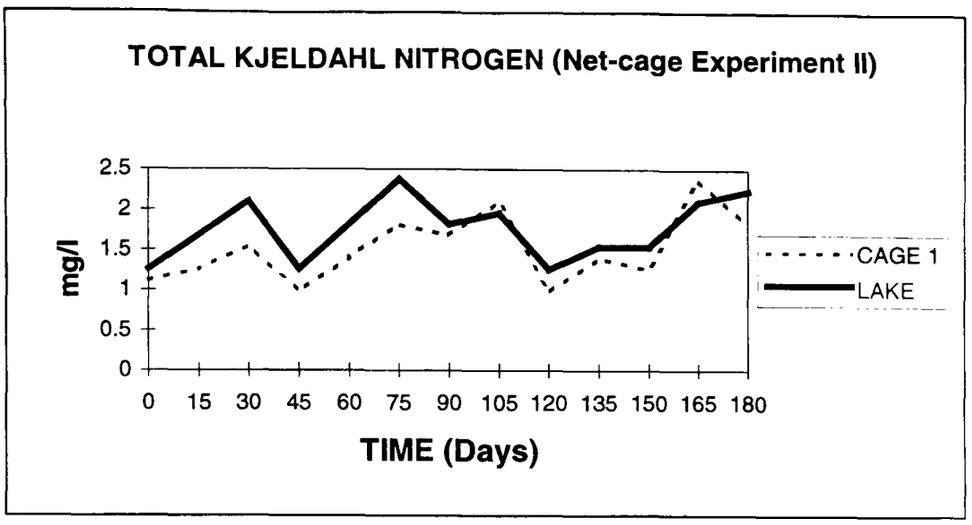
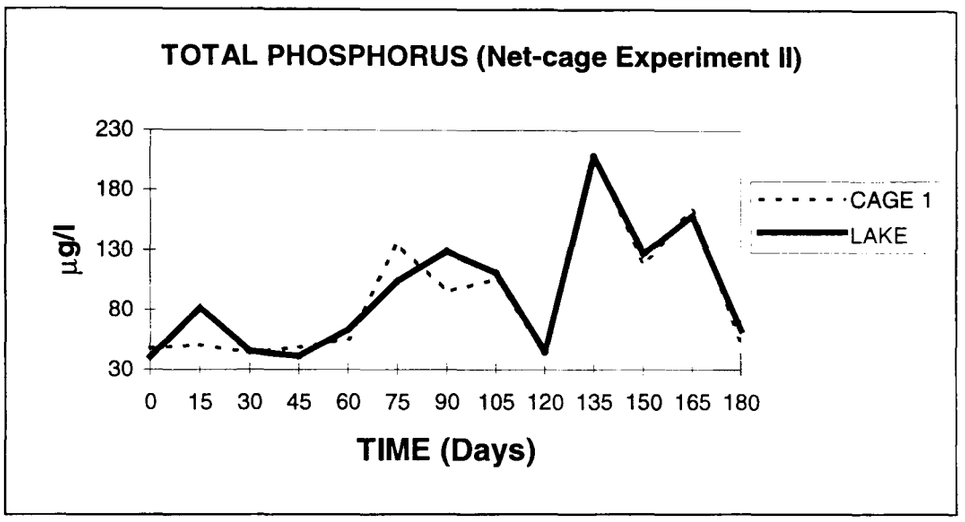


Figure 3.5: (cont.) Evolution of main water quality parameters in cage1 (H.B.= high biomass, L.B.= low biomass) and adjacent reservoir (LAKE) during experiment II.





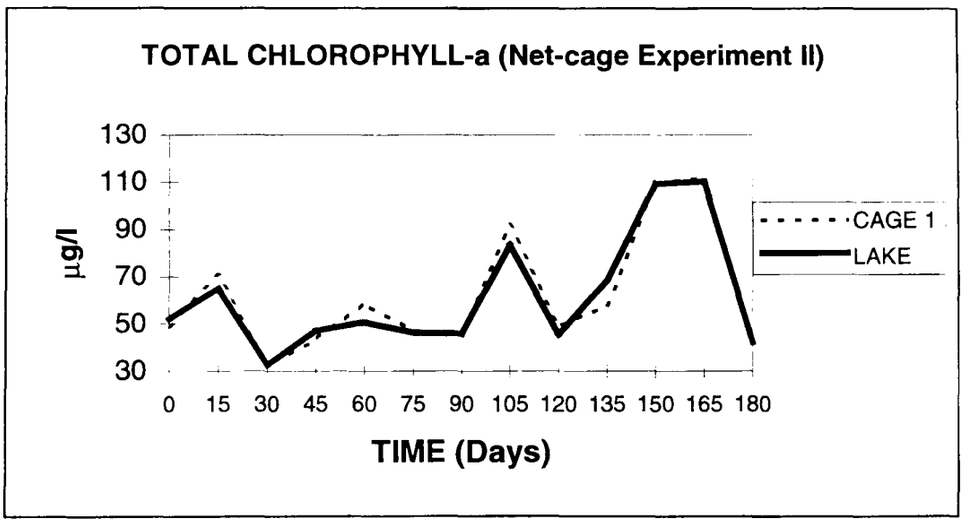
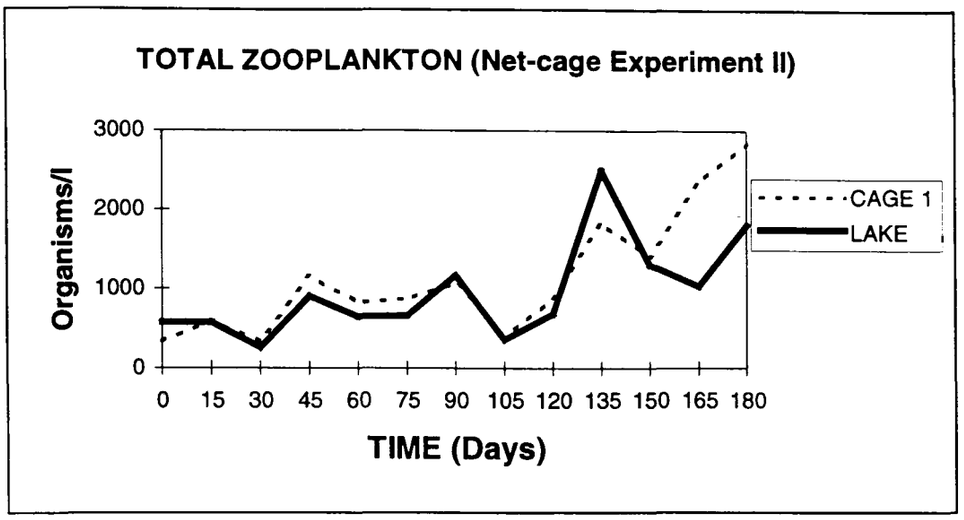


Figure 3.5: (cont.) Evolution of main water quality parameters in cage1 (H.B.= high biomass, L.B.= low biomass) and adjacent reservoir (LAKE) during experiment II.



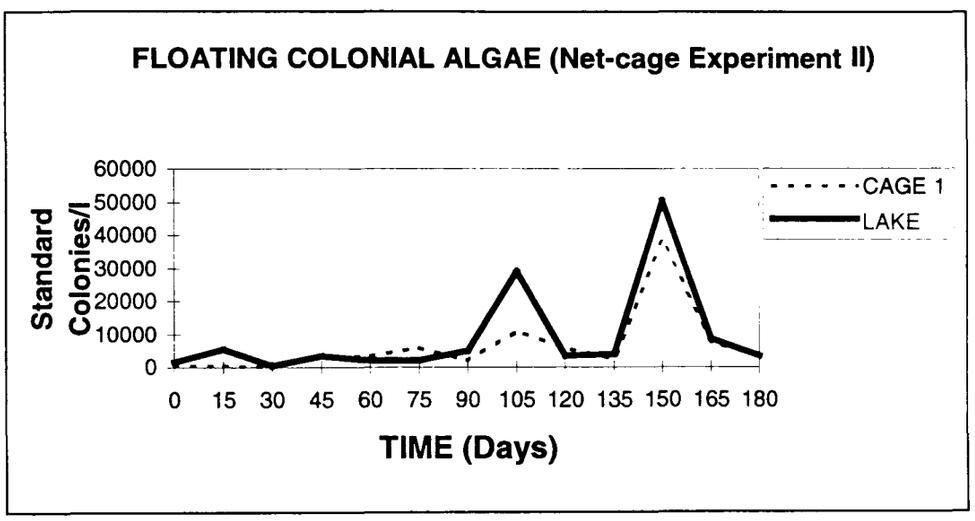
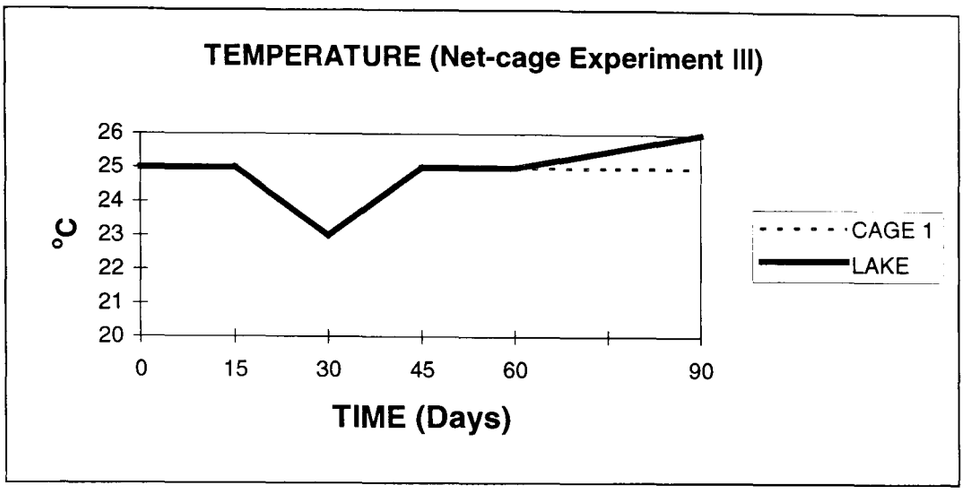
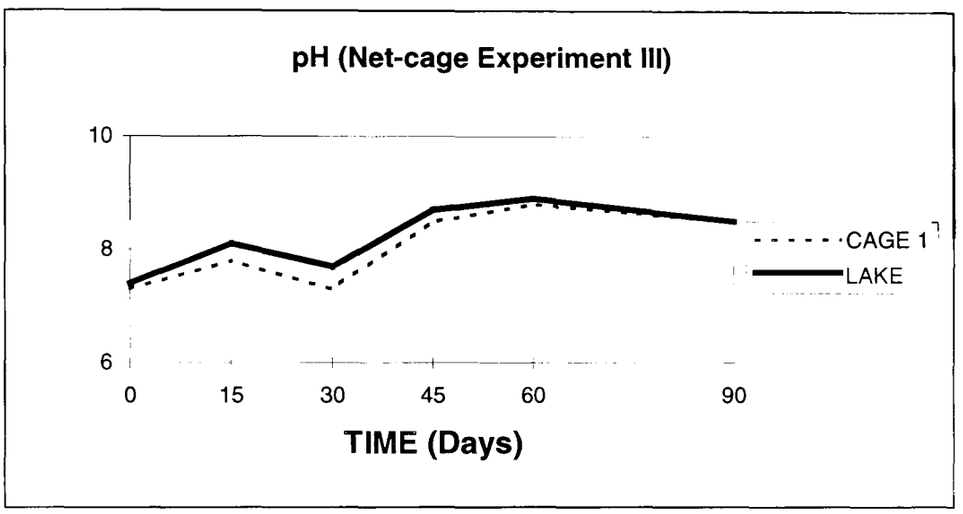


Figure 3.5: (cont.) Evolution of main water quality parameters in cage1 (H.B.= high biomass, L.B.= low biomass) and adjacent reservoir (LAKE) during experiment II.





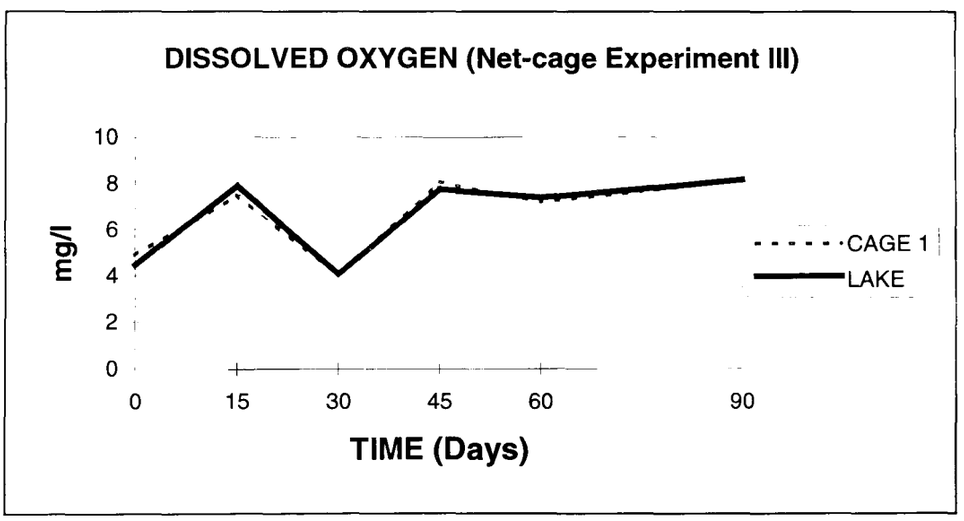
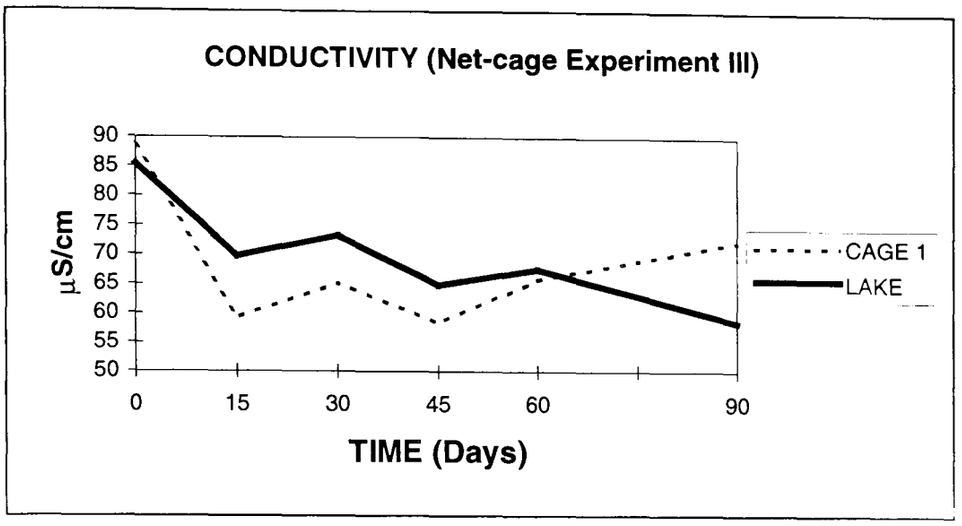
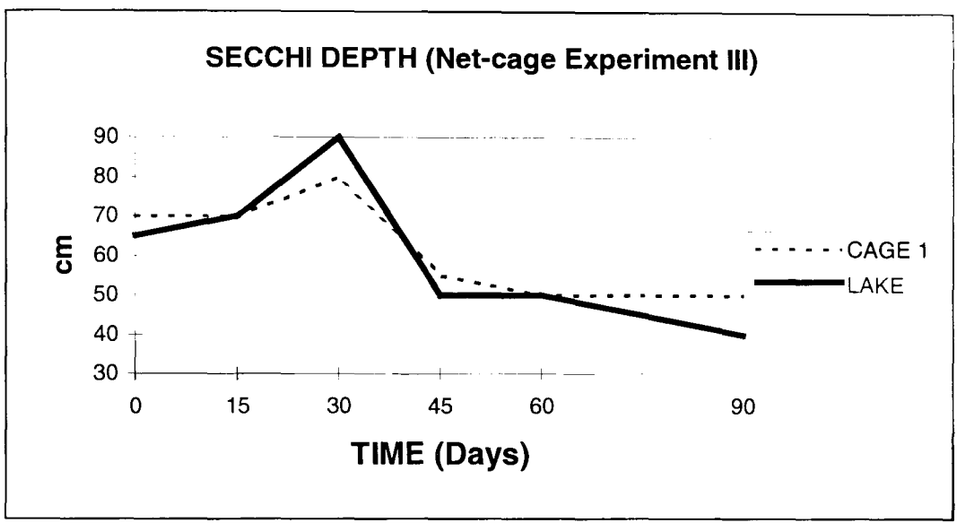


Figure 3.6: Evolution of main water quality parameters in cage1 (H.B.= high biomass, L.B.= low biomass) and adjacent reservoir (LAKE) during experiment III.





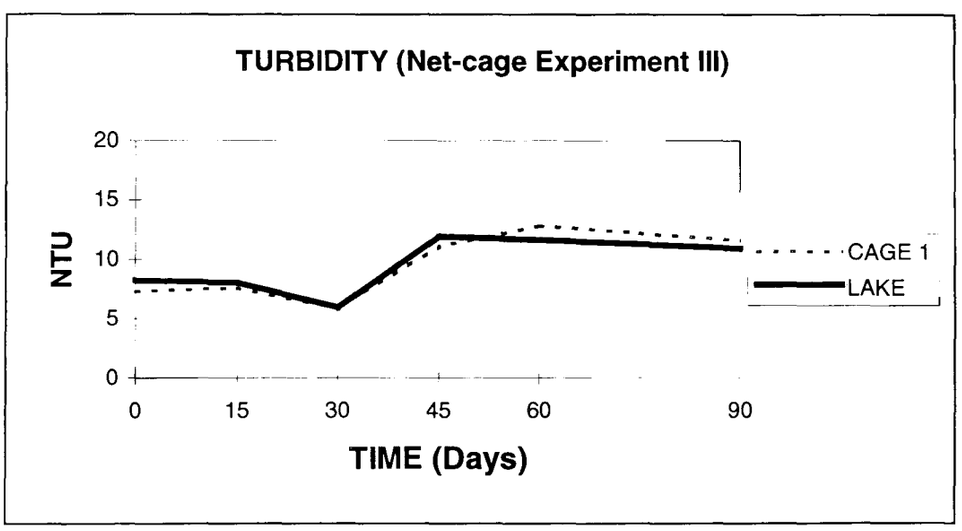
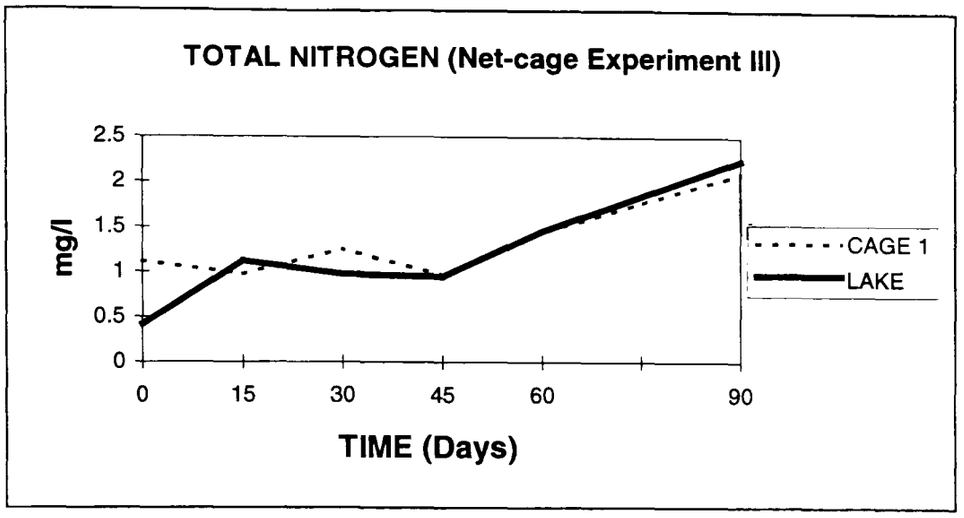
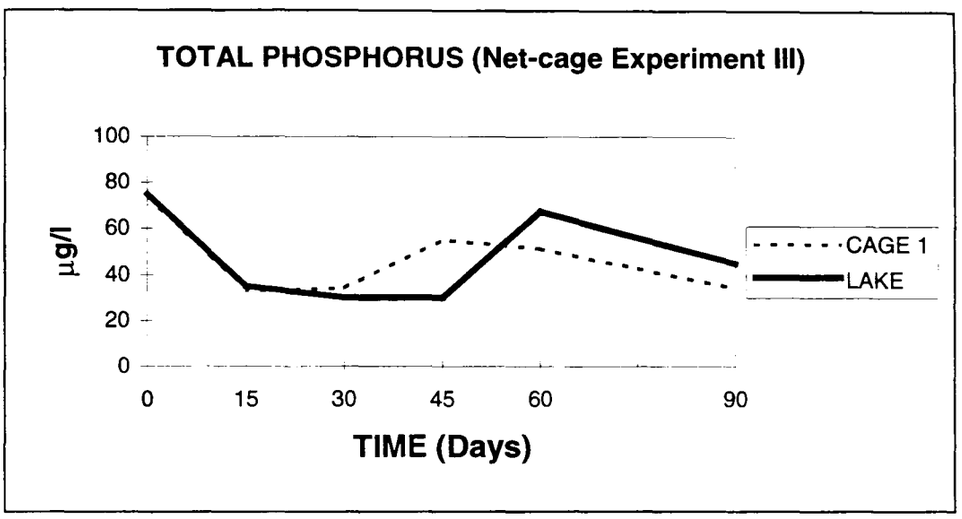


Figure 6: (cont.) Evolution of main water quality parameters in cage1 (H.B.= high biomass, L.B.= low biomass) and adjacent reservoir (LAKE) during experiment III.





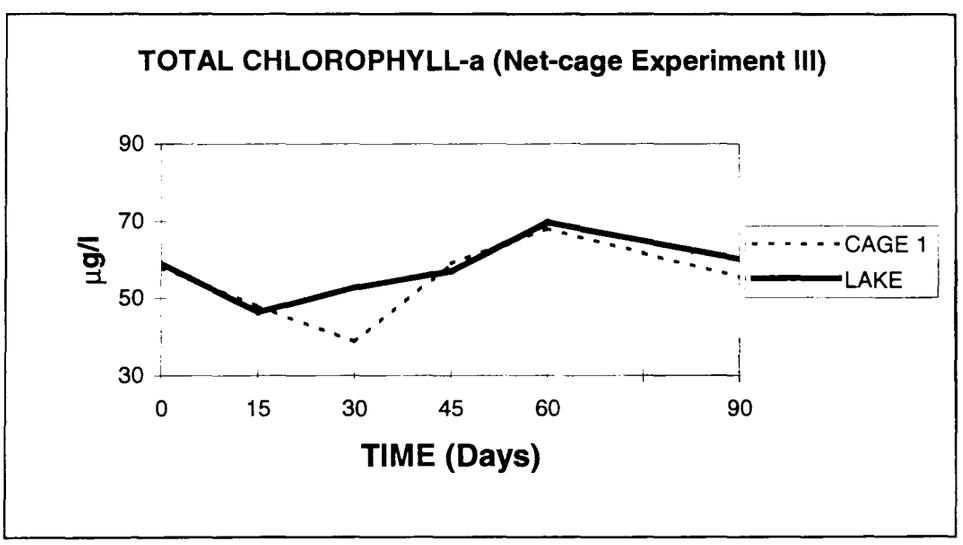
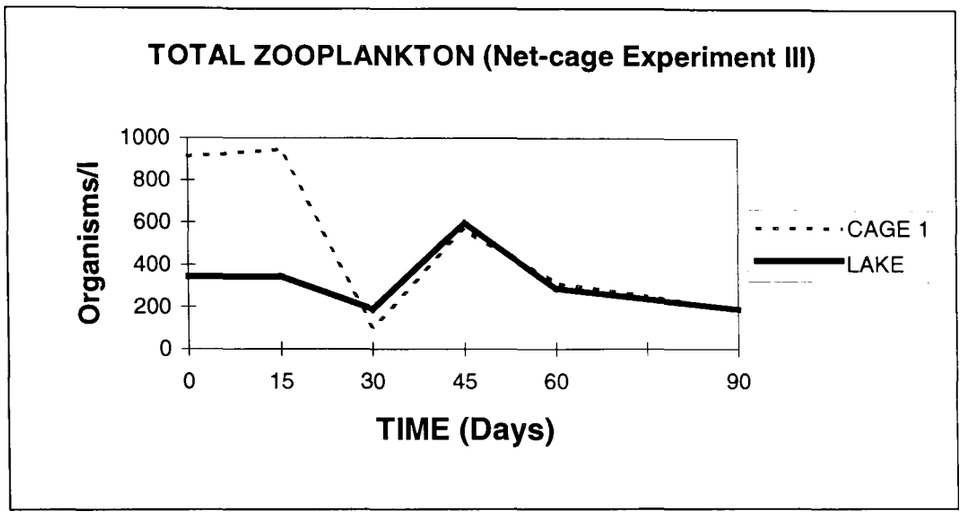


Figure 6: (cont.) Evolution of main water quality parameters in cage1 (H.B.= high biomass, L.B.= low biomass) and adjacent reservoir (LAKE) during experiment III.



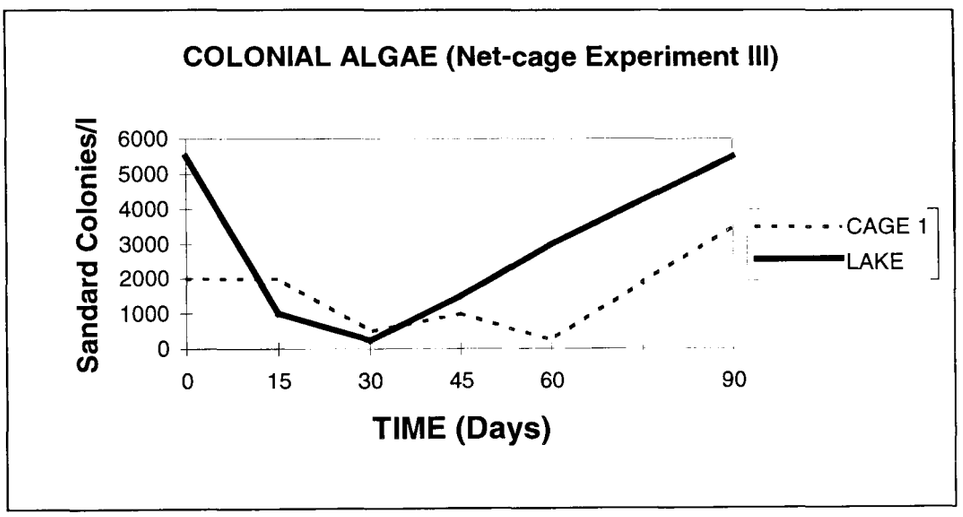
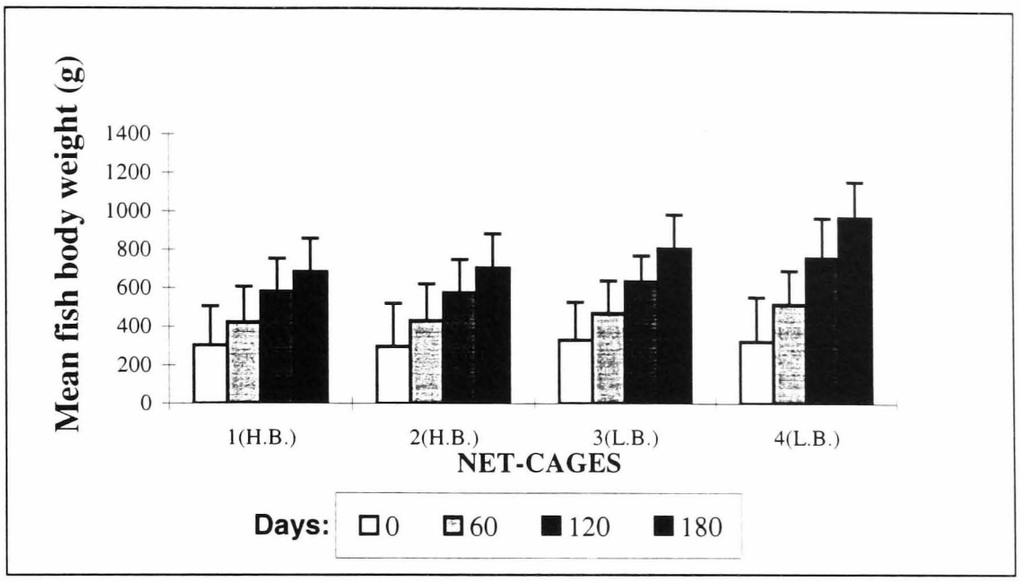


Figure 6: (cont.) Evolution of main water quality parameters in cage1 (H.B.= high biomass, L.B.= low biomass) and adjacent reservoir (LAKE) during experiment III.

Over 180 days, mean daily gains in body weight were 2.11, 2.27, 2.65 and 3.62 g/day, for cages 1, 2, 3, and 4, respectively. This represents a mean increase of 0.45, 0.48, 0.50 and 0.61% in fish initial weight per day for cages 1, 2, 3 and 4, respectively.

During experiment II, all fishes stocked in cages also gained weight. Final average individual body weights were 1.56, 1.47, 1.87 and 1.86 times higher than initial stocking weight for cages 1, 2, 3, and 4, respectively. Figure 3.8 illustrates the evolution of fish mean body weight for each cage and average gain of body weigh per treatment during experiment II. A repeated-measures ANOVA showed that fish growth rate was significantly higher at low fish biomass (P=0.004) compared with high biomass. Over 180 days, mean daily gains in body weight were 3.35, 2.88, 6.01 and 5.35 g/day, for cages 1, 2, 3, and 4, respectively. This represents a mean increase of 0.22, 0.25, 0.35 and 0.35 % in fish initial body weight per day for cages 1, 2, 3 and 4, respectively.

The accidental release of most fishes from all cages near the end of experiment III caused by the sinking of one side of the cage covers, made overall growth rates impossible to determine. However, from the final average weight of the fishes recovered from cage 3 and 4 (10 and 11 fishes, respectively), growth rate could be estimated as 1.22 g/day and 2.39 g/day, or 0.97% and 1.64% of fish initial body weight per day. However, using the conservative approach of taking the largest fish recovered from cages 3 and 4 and considering them to have been the largest at the beginning of the experiment, estimates of growth rates were 2.44 g/day and 4.00 g/day, or 1.03% and 1.15% of fish initial body weight per day, respectively for cages 3 and 4.



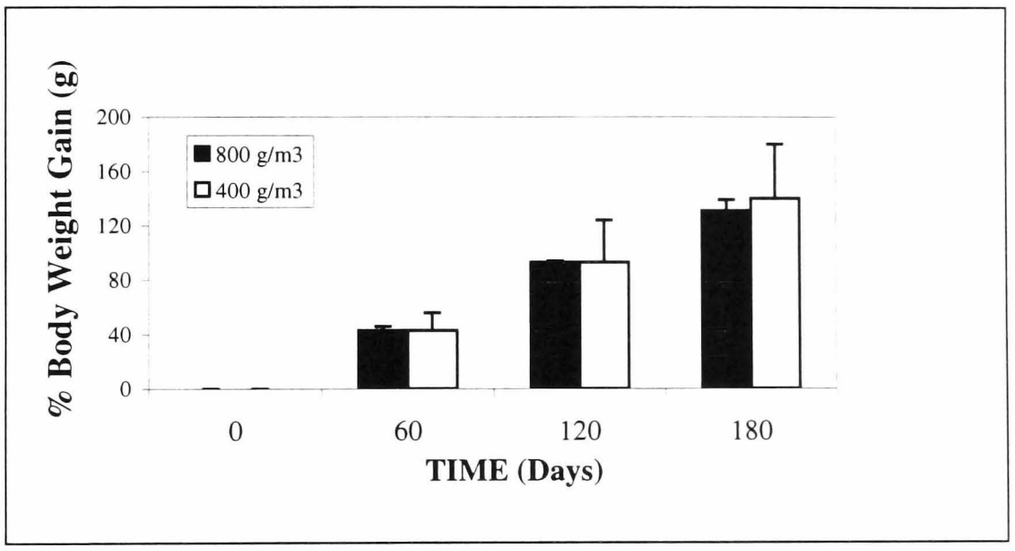
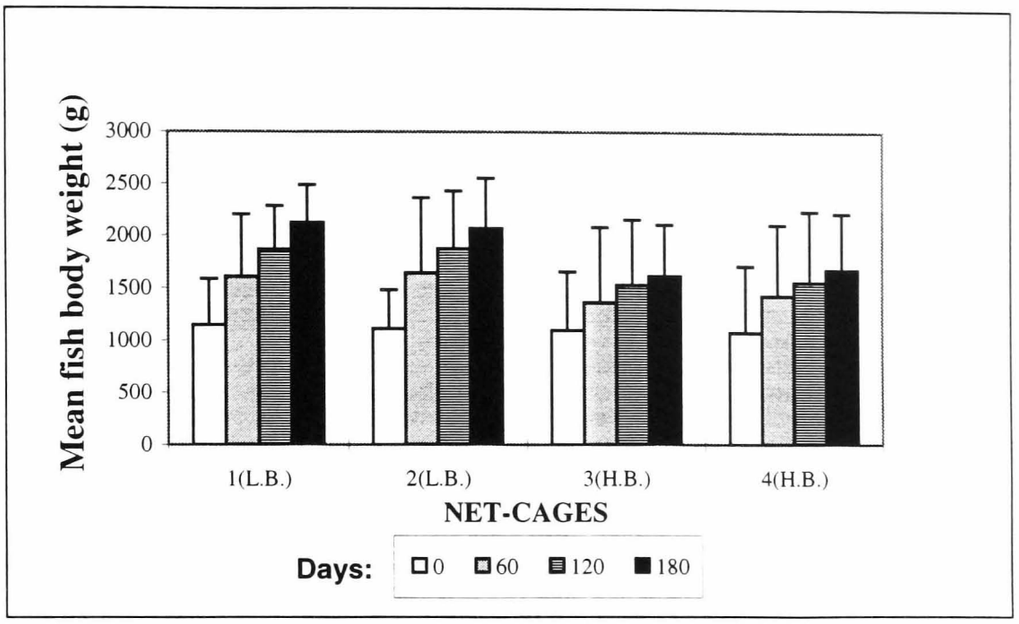


Figure 3.7: Experiment I: mean individual silver carp body weight and percentage of weight gained on days 0, 60, 120 and 180, as compared with initially stocked body weight on day 0. (H.B.= high biomass; L.B.= low biomass).



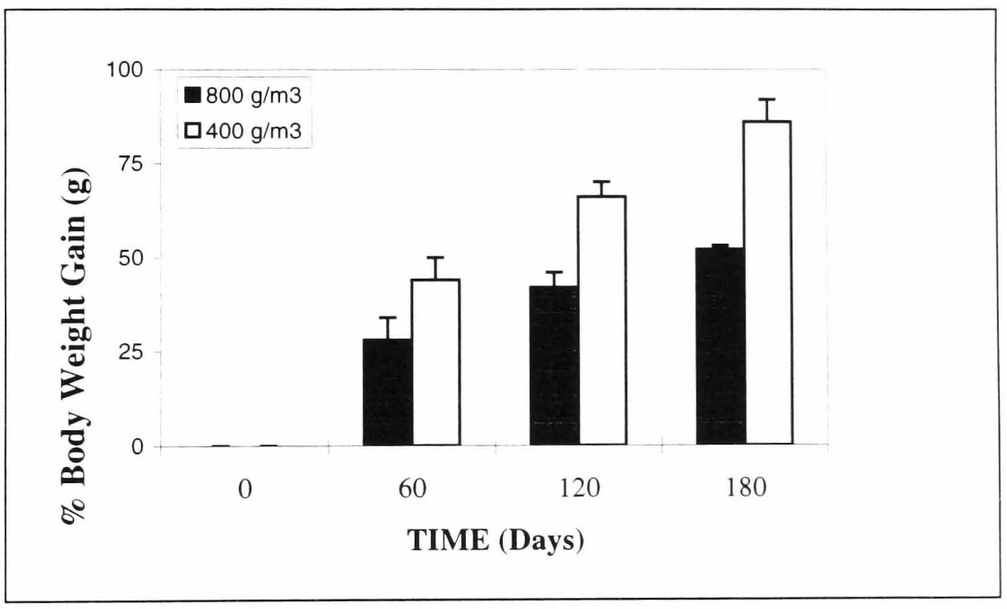


Figure 3.8: Experiment II: mean individual body weight and percentage of weight gained on days 0, 60, 120 and 180, as compared with initially stocked body weight on day 0. (H.B.= high biomass; L.B.= low biomass).

During the course of experiment I, mortality of caged silver carp was very low as only eleven fishes died, of which six were killed by excessive sedation or found dead on days following fish manipulations and were immediately replaced by others of equivalent weight. For this reason, only the remaining five dead fishes (also replaced by living ones) were considered for mortality computations. This represents a mortality of only 6.6%, i.e., a survivorship of 93.4%. Figure 3.9 shows one of the silver carp recovered at the end of experiment I.

Silver carp mortality rate during experiment II was even lower than that of experiment I, as only one fish died and was immediately replaced by one of equivalent body weight. Thus, mortality rate was recorded as 4.2%, and survivorship as 95.8%. The escape of silver carp fingerlings during experiment III prevented estimation of fish mortality.

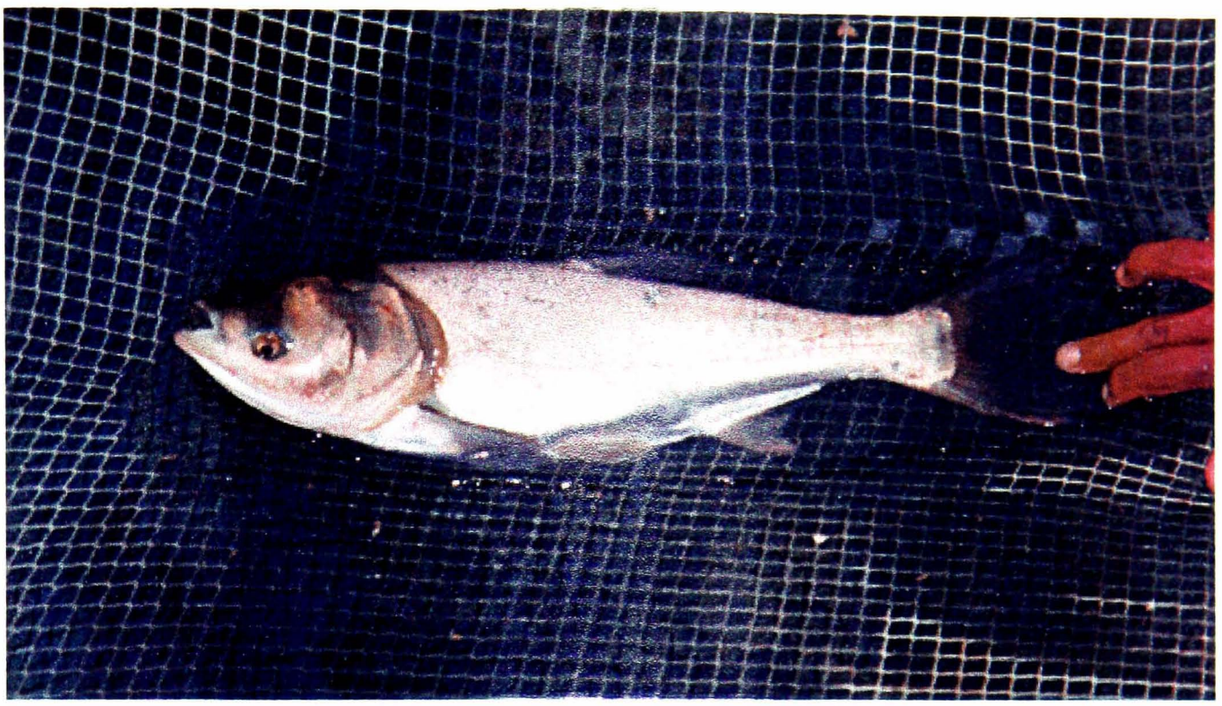
The abundance of nuisance colonial floating algae in the experimental area during the course of experiment I, was relatively low throughout the period (Figure 3.4). Although this is typical of the rainy season (Altafin *et al.*, 1995), the relatively long time interval between samplings would have prevented the detection of any peak in algal abundance in cages and adjacent reservoir during the study period. However, at the end of the dry season in experiment II, *Microcystis* and *Botryococcus* together reached biomass values of up to 50,000 standard colonies/l in both cage and adjacent sampling area (Figure 3.5). During the course of experiment III, the abundance of floating algae remained at the usual low values typical of the rainy season (Figure 3.6).

Despite their relatively low abundance in the environment at the end of each experiment, colonial floating algae was largely consumed by all fishes recaptured from each cage, irrespective of treatment and experiment. Consumption of *Microcystis* and *Botryococcus* was highest for the small fishes in experiment III.

reaching a maximum value of 1.45 x 10^6 standard colonies consumed by a 290-g fish, in comparison with fishes from experiments II and I. The highest number of colonies recorded in a silver carp foregut was 175,000 standard colonies/l for a 1,210-g fish in experiment II, and 82,080 standard colonies/l for a 1,090-g fish in experiment I (Table III.1). Such observed differences in the consumption of colonial algae between experiments cannot be attributed to differences in the availability of food resources, as plankton densities were similar at the end of each experiment, when stomach content analyses were performed.

As illustrated in Figure 3.10, relatively high amounts of other phytoplankton species and detritus were also present in the foreguts of silver carps, although these items were not analyzed here. In addition to algae, zooplankton was consumed in considerable quantities as well. As can be seen in Table III.1, there was a great variation in the number of zooplankton organisms present in the proximate one third of the digestive tract of each silver carp, regardless of fish size.

The total length of the digestive tract of silver carp varied from 3.4 to 8.8 times the total length of fish (Table III.1). A great variation in silver carp intestine length can also be found in the literature, ranging from 3-5 (Wilamovski, 1972), 3.5-7.3 (Cremer & Smitherman, 1980), to 6-10 times the fish total length (Peirond, 1989).



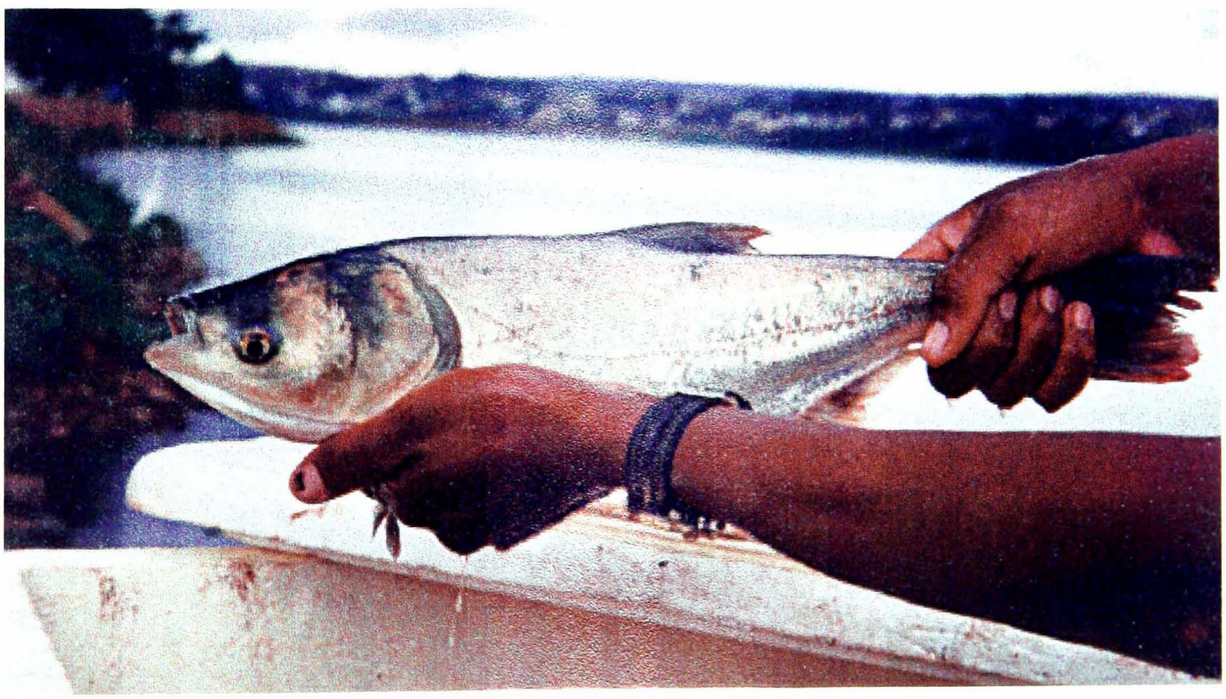


Figure 3.9: Silver carp specimens recovered from net-cages at the end of experiment I.

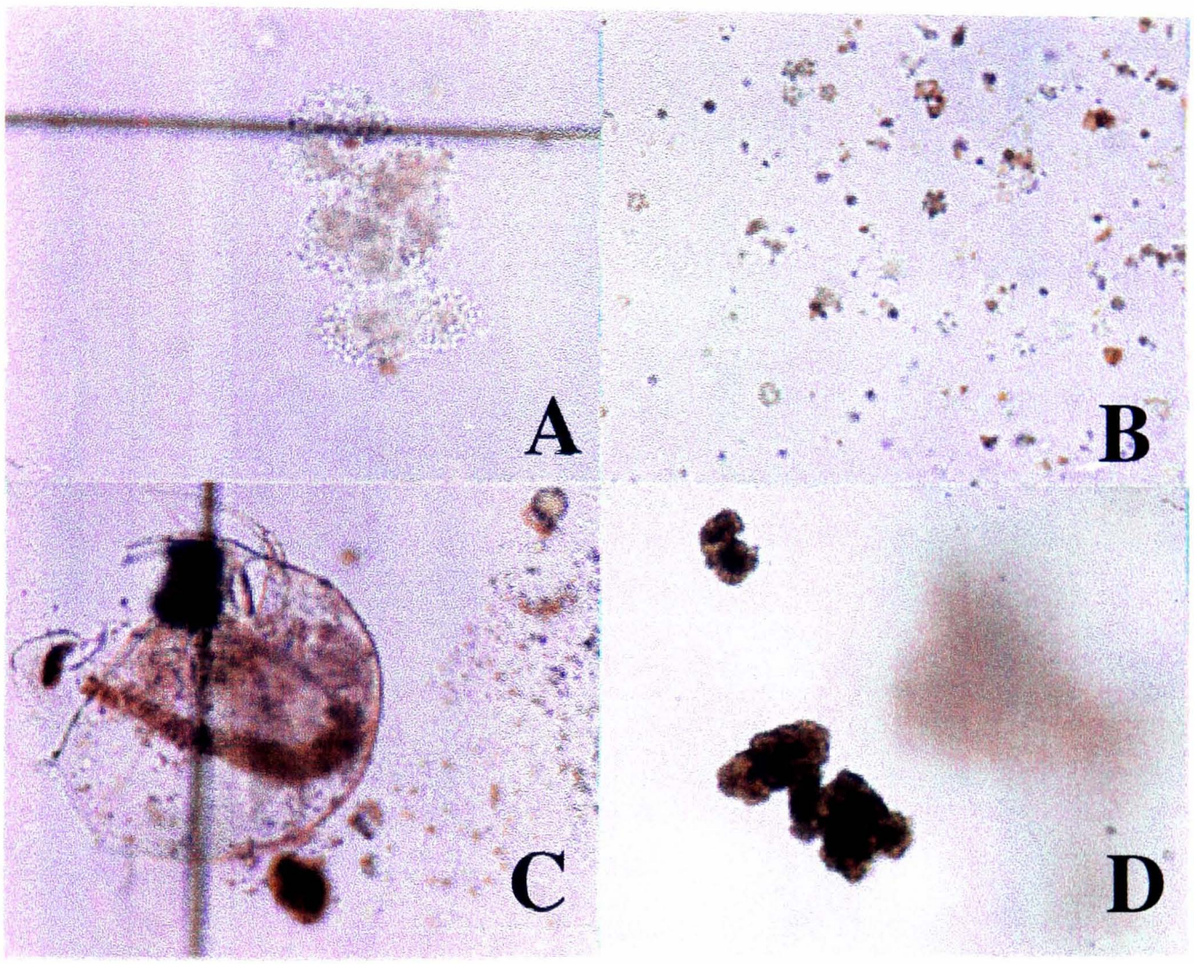


Figure 3.10: Fore intestine content of silver carp captured at the end of the net-cage experiments: A) Colonies of *Microcystis aeruginosa*; B) Desmidiaceae; C) *Bosmina sp.*; D) Colonies of *Botryococcus braunni.*

Table III.1: Total numbers of colonial floating algae and zooplankton recorded in the foregut (proximate one third of the digestive tract) of silver carp individuals recaptured at the end of experiments I, II and III.

Experim. (cage)	Fish	*Botryococcus*	*Microcystis*	Σ Algae	*Keratella*	Nauplius	*Trichocerca*	*Anuraeopsis*	*Bosmina*	Adult Copepoda	Σ Zooplankton
I (1)	(36cm, 785g) intestine=1.9 m	6540	6240	12780	300	0	1260	420	1380	840	4200
I (1)	(33cm, 510g) intestine=1.7 m	7850	2350	10200	350	0	1050	500	1650	550	4150
I (2)	(31cm, 575g) intestine=1.6 m	3500	350	3850	50	0	150	250	0	100	550
I (2)	(29cm, 375g) intestine=1.0 m	1731	208	1939	69.25	0	0	0	69.25	0	138.5
I (3)	(35cm, 715g) intestine=2.1 m	2280	0	2280	0	0	0	60	0	0	60
I (3)	(31cm, 505g) intestine=1.5 m	4950	1150	6100	150	0	1850	200	550	50	2800
I (4)	(35cm, 765g) intestine=1.2 m	2320	116	2436	0	70	140	0	210	0	420
I (4)	(40cm,1090g) intestine=1.4 m	78365	3715	82080	7084.8	0	6220.8	1900.8	259.2	0	15465.6
II (1)	(45cm, 1515g) intestine=2.7 m	2500	62500	65000	450	0	550	8250	50	100	9400
II (2)	(41cm, 1210g) intestine=2.6 m	0	175000	175000	50	50	0	50	0	100	250
II (3)	(36cm, 840g) intestine=1.6 m	700	2900	3600	0	0	0	0	0	50	50
II (4)	(40cm, 1030g) intestine=2.1 m	2200	10100	12300	0	0	0	0	0	0	0
III (3)	(20cm, 165g) intestine=1.3 m	0	1335000	1335000	2300	0	300	7300	300	100	10300
III (3)	20cm, 150g) intestine=1.5 m	5000	1195000	1200000	1100	0	150	450	0	0	1700
III (4)	(25cm, 290g) intestine=2.2 m	0	1446000	1446000	180	0	0	240	0	0	420
III (4)	(16cm, 115g)	0	790000	790000	300	0	0	700	0	0	1000

III.4. Discussion

The area chosen for net-cage experiments represents one of the most eutrophic regions of Lago Paranoá, where blooms of *Microcystis aeruginosa*, oxygen depletion and fish kill are not uncommon events. During the course of these experiments, dissolved oxygen reached critical values (around 2 mg/l), some tilapias were found dead along the reservoir shore adjacent to the cages and nuisance floating algae were present throughout all experimental periods. Despite these unfavorable conditions, silver carp had a very high survivorship and increased body weight when kept in net-cages while feeding exclusively on plankton. The abundant supply of planktonic resources in the nutrient-enriched water of Lago Paranoá explains the observed gain of weight of silver carp as fingerlings, juveniles and adults.

III.4.1. Growth rates of caged silver carp in Lago Paranoá

In order to compare growth rates with other studies, a summary of published works dealing with silver carp aquaculture without supplementary feeding in fish ponds, reservoirs and net cages is presented in Table III.2. In addition to system identification and fish size range, stocking density plus absolute and relative growth rates (as g/day and % body weight/day, respectively) were calculated, whenever possible. Although fish growth rates varied according to the type and location of the ecosystem (as a result of water temperature and food availability), as well as with silver carp stocking density, fish size represented the main factor affecting growth rate. Laws & Weisburd (1994) presented a silver carp growth curve up to the age of 5 years which illustrates a steady growth rate decrease with fish size, in accordance with a von Bertalanffy's growth function.

Silver carp growth rates assessed in the present study, varying from 2.11 to 6.01 g/day, i.e., 0.22 to 1.15 % per day, are within the range of literature values reported in Table III.2 for fish ponds, lakes and reservoirs (from 0.21 to 9.58 g/day, and from 0.06 to 3.34 % per day), as well as for cage aquaculture (from 0.10 to 7.67 g/day and from -0.09 to 2.28 % per day). However, as most studies reviewed in Table III.2 involved the use of smaller fishes whose growth rates are expected to be higher than in the present study, comparisons had to be restricted to only those studies indicated in the table (*).

Average growth rate of juvenile silver carp in experiment I (2.87 g/day or 0.51 % per day) was higher than that reported by Hepher *et al.* (1989) for fish ponds in Israel (0.58 g/day or 0.09 % per day), comparable to that of Bayne *et al.*'s (1991) for net-cages in the United States (4.23 g/day or 0.62% per day), but lower than that of Hamada *et al.*'s (1983) for net-cages in Japan (6.2 g/day or 1.12 % per day).

Mean growth rate of adult silver carp in experiment II (4.39 g/day or 0.29 % per day) was higher than that reported by Adamek & Spittler (1984) for fish ponds in the Czech Republic (1.62 g/day or 0.18 % per day), comparable to that reported by Bialokoz & Krzywosz (1981) for an eutrophic Polish Lake (3.43 g/day or 0.39 % per day), but lower than that obtained by Hamada *et al.* (1983) for net-cages in Japan (6.2 g/day or 1.12 % per day).

In the present study, the highest silver carp growth was obtained for fingerlings in experiment III, as the conservative and underestimated mean growth rate calculated from those recovered fish (3.22 g/day or 1.09 % per day) was comparable to the highest rates obtained by Hepher *et al.* (1989) for fish ponds in Israel (3.03 g/day or 1.45 % per day), by Hamada *et al.* (1983) for net-cages in Japan (3.80 g/day or 1.48 % per day) and by Pradhan & Swar (1987) for net-cages in Nepal (3.84 g/day or 0.98 % per day).

Table III.2: Silver carp growth rates in different aquatic ecosystems without artificial feeding.

Ref.	System Country	weight (g) initial	weight (g) final	time (days)	Fish/m^3 (fish/ha)	g/m^3 (kg/ha)	Growth rate g/day	Growth rate %/day	Selected for comparison
15	Cages (12.5 m^3)	15	320	165	6.0	90	1.85	1.87	
	Nepal	15	285	165	10.0	150	1.63	1.80	
		160	830	165	2.0	384	4.06	1.00	*
		160	760	165	3.0	448	3.63	0.95	*
10	Cages (2.5 m^2)	56	151	44	?	?	2.16	2.28	
	Japan	184	351	44	?	?	3.80	1.48	*
		345	651	50	?	?	6.12	1.28	*
		400	733	53	?	?	6.28	1.15	*
		791	1157	50	?	?	7.32	0.76	*
		1170	1414	53	?	?	4.60	0.36	*
1	Cages	150	360	177	?	?	1.19	0.50	*
	Hungary								
20 (hyb)	Cages (2 m^3)	277	230	202	60.0	16620	-0.37	-0.09	*
	United States	277	1680	202	60.0	16620	6.61	0.90	*
		277	1720	202	60.0	16620	7.04	0.91	*
		277	830	145	60.0	16620	3.64	0.76	*
24	Cages (10 m^3)	145	365	90	4.0	315	2.44	1.03	
	Brazil	200	560	90	8.0	524	4.00	1.15	
		304	685	180	2.6	800	2.11	0.45	
		299	707	180	2.6	800	2.67	0.48	
		332	809	180	1.2	400	2.65	0.50	
		325	977	180	1.2	400	3.62	0.61	
		1096	1615	180	0.8	800	2.88	0.22	
		1073	1676	180	0.8	800	3.35	0.25	
		1143	2125	180	0.4	400	6.01	0.35	
		1111	2074	180	0.4	400	5.35	0.35	
3	Cages (1 m^3)	22	271	159	25.0	543	1.60	1.60	
	Fish Pond	22	439	159	?	?	2.6	1.91	
	United States								
14	Cages (4 m3)	63	488	150	5.0	315	2.84	1.38	
	Bangladesh								
8 (bh)	Cages (120 m^3)	23	2786	360	1.7	38	7.67	1.34	
	Singapore	23	2456	360	2.5	58	6.76	1.30	
		22	1803	360	3.3	74	4.95	1.23	
		23	1109	360	4.2	95	3.02	1.09	
9	Cages (0.57 m^3)	15	189	190	17.0	265	0.92	1.34	
	South Africa								
17	Cages (6.25 m^2)	3	30	257	28.0	96	0.10	0.84	
	India	3	143	252	28.0	84	0.56	1.54	

Ref.	System Country	weight (g) initial	final	time (days)	Fish/m^3 (fish/ha)	g/m^3 (kg/ha)	Growth rate g/day	%/day	Selected for comparison
18	Fish Ponds	53	688	156	(1300)	(70)	4.07	1.66	
	Israel	53	362	156	(2600)	(138)	1.98	1.24	
		364	411	220	(2000)	(728)	0.21	0.06	*
		684	892	220	(1000)	(684)	0.95	0.12	*
11	Fish Ponds Czech Republic	790	1010	136	(2750)	(2173)	1.62	0.18	*
5	Lake Poland	667	1141	138	?	?	3.43	0.39	*
16	Sewage Pond	6	989	306	875	1	3.20	1.66	
	Hong Kong	6	1013	306	875	1	3.29	1.66	
6	Fish Ponds (0.2	22	130	180	(1500)	3	0.60	0.99	
	ha) Poland	20	120	180	(3000)	7	0.54	0.95	
7	Fish Ponds (0.2	65	234	195	(4000)	?	0.87	0.66	
	ha)	65	181	195	(8000)	?	0.59	0.53	
	Poland	65	159	195	(12000)	?	0.48	0.46	
23	Lagoon United States	13	2830	1125	?	?	2.50	0.48	
2	Fish Ponds	10	232	180	(1000)	(10)	1.23	1.76	
	India	10	428	180	(600)	(6)	2.32	2.11	
		17	130	180	(1000)	(10)	0.63	1.14	
13	Fish Ponds	10	600	182	?	?	3.24	2.25	
12	Fish Pond	17	466	281	?	?	1.60	1.19	
	Hawaii	25	458	281	?	?	1.54	1.03	
					?	?			
4	Sewage Pond	41	757	365	?	?	1.96	0.80	
		41	1243	730	?	?	1.65	0.47	
19	Fish Pond Hawaii	98	400	309	(75)	(7)	0.98	0.46	
22	Reservoir Marroco	1	25	90	?	?	0.26	3.34	
21	Fish Pond Israel	54	694	100	?	?	6.4	2.59	
25	Reservoir	500	12000	1200	?	?	9.58	0.27	
	Enclosures	29	68	42	(250)	(73)	0.94	2.07	
	Brazil	36	86	42	(375)	(137)	1.18	2.07	
		44	82	42	(625)	(275)	0.91	1.51	
		40	63	42	(1000)	(398)	0.55	1.10	
		624	662	84	(875)	(550)	0.45	0.07	

LEGEND: **1-** Coche (1979); **2-** Dey *et al.* (1979); **3-** Cremer & Smitherman (1980); **4-** Henderson (1980) *in* Laws & Weisburd (1994); **5-** Bialokoz & Krzywosz (1981); **6-** Opuszynski (1981); **7-** Opuszynski (1981); **8-** Ling (1982); **9-** Gaigher & Krause (1983); **10-** Hamada *et al.* (1983); **11-** Adamek & Spittler (1984); **12-** Costa-Pierce (1984) *in* Laws & Weisburd (1994); **13-** Chien & Tsai (1985) *in* Laws & Weisburd (1994); **14-** Mollah *et al.* (1987); **15-** Pradhan & Swar (1987); **16-** Sin & Chiu (1987); **17-** Sivakami *et al.* (1987); **18-** Hepher *et al.* (1989); **19-** Laws & Weisburd (1990); **20-** Bayne *et al.* (1991); **21-** Milstein *et al.* (1991); **22-** ONEP (1991); **23-** Lieberman (1996); **24-** Present study; **25-** Starling *et al.* (*in prep.*); (bh) = bighead carp; (hyb) = hybrid silver carp x bighead carp.

III.4.2. Production and impacts of silver carp cage culture in Lago Paranoá

The high fish growth rate and survivorship observed in the present study attest the great potential for extensive silver carp cage culture in Lago Paranoá. Average annual silver carp production in this ecosystem is equivalent to 30,000 kg/ha/year based on growth of fish in net-cages experiment I, 15,000 kg/ha/year based on experiment II and 105,000 kg/ha/year based on experiment III. High silver carp yields, ranging from 70,000 to 120,000 kg/ha/year, were also reported by Pradhan & Swar (1987), using similar fish stocking densities and size of cages, when evaluating the potential for silver carp cage culture in a tropical eutrophic reservoir in Nepal.

Annual fish production exceeding 200,000 kg/ha/year has been reported by Ling (1982) in a pilot project rearing bighead carp (*Hypophthalmichthys nobilis*) in floating net-cages in the eutrophic Seletar Reservoir (320 ha), Singapore. Considering the promising results from preliminary experiments, within 8 years of project a total of 26.4 ha of the reservoir were occupied by extensive bighead carp cage aquaculture. As a consequence, as described by Kestemont (1995), " there had been a steady decline in the frequency of algal blooms and plankton biomass corresponding to a decrease in fish production per unit area per unit time". Outputs such as control of nuisance planktonic organisms filtered by fish, nutrient removal by regular harvesting of fish and production of low-cost protein for human consumption make this project in Singapore an illustrative example of the potential use of caged filter-feeding fish for controlling eutrophication.

In the present study, the absence of significant changes in water quality in the vicinity of the cages does not imply that silver carp had a negligible impact on the plankton community. The constant interchange of water between four small cages and the surrounding experimental site in such a large, open area of the reservoir would not allow the detection of any drop in plankton abundance caused by fish feeding.

The high growth rates of silver carp in a restrictive condition of planktophagy imposed by net-cages and the massive presence of planktonic organisms in the digestive tract of fish as recovered from all three experiments make clear that silver carp cropped actively on abundant phytoplankton.

III.4.3. Consumption of cyanobacteria by silver carp filtering activity

Although the impact of silver carp on the plankton community and water quality of Lago Paranoá will be considered further in the following chapters when dealing with large closed systems represented by limnocorrals, it seems important to emphasize the high silver carp consumption of nuisance bloom-forming algae in net-cages.

Silver carp filter-feeding on cyanobacteria is well documented in the literature and can be pointed out as one of the main reasons for its introduction into eutrophic lakes and reservoirs around the world (see review in Costa-Pierce, 1992). Although bloom-forming *Microcystis* has been reported to be one important food item for silver carp in many ecosystems (Hamada *et al.*, 1983; Shapiro, 1985; Miura, 1990), its digestibility and nutritive value have been seriously questioned (Hamada *et al.*, 1983; Ruzicka & Ruzickova, 1988; Bayne *et al.*, 1991). In contrast with such indications of low digestibility and poor nutritional value, it has been determined in the laboratory that silver carp assimilate more than 50% of the free aminoacids and carbohydrate (Itawa *et al.*, 1989) and more than 25% of the dry matter, protein and lipid (Shaolian *et al.*, 1990) from *Microcystis* during intestinal passage. Considering that digestibility of *Microcystis* by silver carp is a controversial matter not yet evaluated in Lago Paranoá, the present discussion will only be restricted to the ingestion of colonial algae (*Microcystis* and *Botryococcus*) by silver carp in net-cages.

The potential for a large-scale silver carp cage culture in Lago Paranoá, in terms of removal of undesirable sestonic particles such as net-phytoplankton species, can be preliminarily evaluated by the following calculations using the data from digestive tract content analyses. Based on the abundance of colonial algae in the environment (net-cages and adjacent reservoir sampling points) and in silver carp foregut at the end of each experiment, it is possible to estimate the consumption rates of such nuisance phytoplankton. Considering the median of the number of standard colonies in silver carp foregut from Table III.1 as equal to 4,975 for experiment I, to 38,650 for experiment II and 1,267,500 for experiment III, then a daily silver carp consumption of colonial algae can be estimated based on food passage time and daily feeding intensity.

Food passage for silver carp is reported to be 3 hours (at 20-21 °C in Adamek *et al.*, 1990 and at 22-24 °C in Herrmann, 1983), 3.3 to 6.3 hours (Moskul, 1976), 4-5 hours (at 28.5 °C in Henebry *et al.*, 1988), 4 hours (at 23 °C in Omarov, 1970), 5-10 hs (at 22-25 °C in Okoniewska & Kruger, 1979) and 10 hours (at 22.6 °C in Bialokoz & Krzywosz, 1981). By taking 3 and 6 hours as the lower and upper limits of gut passage time in silver carp at 25 °C, fish would be filling their intestine with food from 2 to 4 times during 12 hours of feeding intensity. To be conservative, it is assumed that silver carp feed for only 12 hours a day, although it has been shown that silver carp essentially cease feeding for only 5 hours (from 23:00 hs to 4:00 hs) in a 24-hours basis (Wang *et al.*, 1989).

On the basis of the microscope enumerations, the abundance of standard colonies can be converted into biomass by taking the volume of the sphere inscribed inside the 90 μm x 90 μm square of the Kellner eye-piece and assuming a specific gravity of 1 mg mm^{-3}. Using this assumptions, the volume and weight of 1 standard colony equals 381,703 μm^3 and 381.7 x 10^{-9} g, respectively.

Then, lower (2 fillings per 12 hours) and upper (4 fillings per 12 hours) daily consumption of colonial algae by silver carp can be calculated for each experiment, as shown in Table III.3.

Table III.3: Estimates of daily consumption of colonial algae by silver carp during net-cage experiments.

1) Experiment I:	
Lower estimate	4,975 x 3 x 2 = 29,850 st.col./fish/day or 45 st.col./g ww/day or 11.4 mg/fish/day or 0.017 mg/g ww/day
Upper estimate	4,975 x 3 x 4 = 59,700 st. col./fish/day or 90 st.col./g ww/day or 22.8 mg/fish/day or 0.034 mg/g ww/day
2) Experiment II:	
Lower estimate	38,650 x 3 x 2 = 231,900 St.col./fish/day or 202 St.col./g ww/day or 88.5 mg/fish/day or 0.077 mg/g ww/day
Upper estimate	38,650 x 3 x 4 = 463,800 St. col./fish/day or 404 St.col./g ww/day or 177 mg/fish/day or 0.154 mg/g ww/day
3) Experiment III:	
Lower estimate	1,267,500 x 3 x 2 = 7,605,000 St.col./fish/day or 42,250 St.col./g ww/day or 2,903 mg/fish/day or 16.13 mg/g ww/day
Upper estimate	1,267,500 x 3 x 4 = 15,210,000 St.col./fish/d or 84,500 St.col./g ww/day or 5,806 mg/fish/day or 32.26 mg/g ww/day

Lower and upper estimates of daily consumptions of *Microcystis* and *Botryococcus* were respectively 0.0017% and 0.0034% for experiment I, 0.0077% and 0.0154% for experiment II and 1.61% and 3.2% for experiment III. The values obtained for experiments I and II are far below the daily consumption estimates of Cyanobacteria by silver carp in the literature. Herodek *et al.* (1989) measured the daily consumption of *Anabaena* by silver carp in the laboratory as equals to 1.43, 4.15 and 7.01 % of wet body weight respectively for algal biomass of 15, 40 and 65

mg/l. The values obtained for experiment III, for a colonial algae biomass equivalent to 1.7 mg/l, are more consistent with the literature.

Moreover, it is important to emphasize that colonial algae were not the only food item ingested and many other phytoplankton species, as well as zooplankton organisms and detritus were also present in the gut contents (Figure 3.10). Although most were not quantified, some of such food items might have been of greater importance, on a volumetric basis, than colonial algae. Silver carp is well known to have a high daily food intake, reaching up to 20% of fish body weight per day (Borutskiy, 1973). Daily consumption rates per body weight of fish equal to 2.7-7.2 % (Leventer, 1979), 5.7-11.7 % (Bialokoz & Krzywosz, 1981), 12.2 % (Chen *et al.*, 1985), 15 % (Shei *et al.*, 1993), 16.5 % (Wang *et al.*, 1989), 17 % (Rothbard, 1981), 19 % (Omarov, 1970) and 20.9 % (Moskul, 1977) have been reported for silver carp in the literature.

From the above considerations, it seems clear that only consumption rates from experiment III might have been sufficiently high to generate an impact on the phytoplankton community. As phytoplankton generation time can be shorter than a day, a relatively high proportion of the total colonial algae abundance would have to be filtered every day for the fish to have a significant effect on algal biomass. A removal of 50% of total colonial algal biomass per day was assumed to impact population size.

Thus, consider a given area of 1 ha and 2.5 m mean in depth (25,000 m^3) where a colonial algae biomass of 4,500 standard colonies/l (i.e., total of 11.25 x 10^{10} standard colonies or 43 kg of colonial algae) is established. The caged silver carp biomass required to be stocked in this area to remove half of the colonial algae standing crop per day can be calculated based on the lower and upper estimation of consumption rates from experiment III. Taking the minimum estimated consumption rate of 16.13

mg/g/day, or 0.16% of its own body weight/day, a total of 1,333 kg of silver carp stocked in cages would be required to remove 50% of the algae (21.5 kg) per day. Based on the upper estimation of consumption rate of 32.26 mg/g/day, or 0.32% of its own body weight/day, half of this amount of fish, i.e., 667 kg of silver carp stocked in the area would be capable of removing 50% of the algae (21.5 kg) per day.

Considering a silver carp filtering rate of 0.3 l/ g ww/ hour (Starling, unpublished data), which falls within the limits of 0.26 l/ g ww/hour (Herodek *et al.*, 1989) and 0.55 l/ g ww/hour (Smith, 1989) reported from literature, it is possible to predict how long it would take for the stocking population in cages to filter the whole volume of water from this area. The total fish biomass of 1,333 kg, would potentially filter a total volume of 4,800 m^3 per day and would require a minimum of 5 days for filtering the overall water volume. Where only 666 kg of fish were stocked, the total daily volume of 2,400 m^3 would be filtered and it would take at least 10-11 days to have all water passed through the silver carp filtration apparatus.

This scenario showing a relatively higher consumption of colonial algae than expected from passive filtration of the water alone can be explained by silver carp selective feeding on the surface area where colonial floating algae is more concentrated. As demonstrated by Dong & Li (1994), silver carp has the ability to select a feeding areas, seeking and staying in areas in which high plankton concentrations or preferred food items are present.

Using the stocking density of 800 g/m^3 from experiment III, held in large net-cages of 6 x 5 x 2 m (30 m^2 and 60 m^3), a minimum of 14 and a maximum of 28 cages holding 48 kg of fish each would be required. A total area of 420 m^2 or 840 m^2 would be occupied by net-cages, which represents 4.2% and 8.4% of the total area under consideration (1ha). An annual fish production of 315 kg per cage (5.25

kg/m^3/year) or a total of 4.4 to 8.8 tons of fish would be expected to be produced every year at very low cost as no artificial food would have to be given to fish.

This final figure of a large-scale extensive cage culture capable of producing water quality improvements is very similar to the system successfully implemented in Singapore (Ling, 1982), where 8.25% of the surface area of Seletar Reservoir were occupied by 220 large net-cages (120 m^3) holding 82,000 bighead carp fingerlings, and producing over 80,000 kg of fish per year (5.6 kg/m^3/year).

Nevertheless, despite water quality improvements of filtering out large amounts of sestonic particles and partially incorporating them as fish biomass for human consumption, the adoption of such large-scage cage culture in Lago Paranoá may conflict with other important multiple uses of this ecosystem. As a matter of discussion in the final chapter (Final Remarks and Recommendations), all potential restrictions and interferences on landscaping, aquatic sports and navigation must be weighed against such ecological, social and economic benefits in order to determine the scale of silver carp cage culture that might be implemented in Lago Paranoá.

III.5. Conclusions

- Silver carp showed an excellent adaptation to the hypertrophic conditions of Lago Paranoá, as demonstrated by low mortality rates in net-cages (always below 10 %) throughout all climatic seasons.
- Silver carp growth rates were comparatively higher at low stocking rates, despite the fact that all cages were equally supplied with abundant planktonic food supply.

- Growth rates and production of caged silver carp in Lago Paranoá without supplementary food were both satisfactory and comparable to the best examples in the literature.
- Nuisance bloom-forming algae (*Microcystis aeruginosa* and *Botryococcus braunii*) were largely consumed by silver carp as evidenced by the high number of these algae in the fore intestine of individual fishes (from 1,939 to 1,446,000 st. colonies).
- Although no impacts of silver carp cage culture on limnological parameters were detected in the vicinity of the experimental area due to the constant flow of water through the cages, the substantial removal of sestonic particles (especially nuisance phytoplankton) represents a potentially positive effect on water quality. In addition, cage silver carp will enable the removal of a considerable fraction of high nutrient internal load incorporated as plankton biomass by harvesting the fish.
- The potential detrimental effects of cage culture (particularly nutrient enrichment of bottom sediment) have to be evaluated prior to establishing the appropriate scale of the extensive cage culture unit area in the selected region of Lago Paranoá.

CHAPTER IV
EFFECTS OF SILVER CARP BIOMASS ON WATER QUALITY

IV.1. Introduction

It is only since the late 1960's and early 1970's that filter-feeding fish, and particularly silver carp, have received special attention as possible agents to increase fish production by utilizing a relatively unexplored food resource, the phytoplankton (Januszko, 1972). Based on the assumption that silver carp is an obligate phytoplanktivorous fish, its potential use to counteract algal blooms had frequently being recommended and tested mainly in fish ponds under temperate climatic conditions (see reviews in Lazzaro, 1987; Northcote, 1988; Smith, 1988; and Costa-Pierce, 1992).

In the majority of such studies however, silver carp did not successfully control algal biomass as fish indirectly promoted the excessive growth of small nanno-phytoplankton by consuming the net-phytoplankton, its competitors, and the zooplankton, its grazers (Opuszynski, 1979; Milstein *et al.*, 1985a,b, 1988; Burke *et al.*, 1986). Despite some recent field evidences of the effective control of nuisance cyanobacteria in lakes and reservoirs outside temperate regions (Buch, 1977; Carruthers, 1986; Leventer & Teltsch, 1990; Miura, 1990; Teltsch *et al.*, 1991; Sagi, 1992; Bangxi *et al.*, 1993), the usefulness of silver carp as a biomanipulation agent to improve water quality is still very controversial (see Costa-Pierce, 1992, comments from Smith, 1994 and reply from Laws & Weisburd, 1994).

Differences in the size-structure of phytoplankton community (Laws & Weisburd, 1990), strength of the zooplankton grazing pressure on dominant algae (Arcifa *et al.*, 1995), and above all the stocking density and biomass of silver carp (Spataru & Gophen, 1985; Milstein, 1992; Starling, 1993a) represent key factors which may contribute to silver carp failures in some occasions.

The role of planktivorous fish biomass in the top-down control of aquatic food chain started receiving a special attention when the nutrients release by fish were shown to substantially contribute to internal nutrient loadings and thus to eutrophication process (McQueen *et al.*, 1986; Threlkeld, 1987; Northcote, 1988; Brabrand *et al.*, 1990).

Using an experimental mesocosm setup, Lazzaro *et al.*(1992) quantified the effects of increasing planktivores biomass (0-75 g/m^3) of filter-feeding gizzard shad and visual-feeding bluegill sunfish and concluded that fish biomass is more important than planktivore type as a regulator of plankton communities and water quality. However, the most important issue, i.e. whether the shape of the planktivore biomass : phytoplankton biomass relationship is linear (McQueen *et al.*, 1986) or sigmoid (Lazzaro *et al.*, 1992), is still under debate.

A recent tank-mesocosm study (Drenner *et al.*, 1996) in which five biomass levels of filter-feeding gizzard shad were cross-classified with two levels of lake trophic state (oligotrophic and eutrophic conditions), shows that filter-feeding omnivorous fish interact synergistically with trophic state. Indeed, the biomass-dependent increase in chlorophyll by filter-feeding omnivorous fish was more pronounced in eutrophic rather than in oligotrophic conditions.

Earlier experimental works in eutrophic Lago Paranoá have demonstrated (1) how efficient the consumption of dominant cyanobacteria by silver carp in aquaria is by comparison with that of other filter-feeding fishes (Starling & Rocha, 1990) and (2)

the significant reduction of both total phytoplankton density and biomass in mesocosms in the presence of a moderate biomass level of silver carp (Starling, 1993a).

However, the desirable range of silver carp biomass at which the control of excessive phytoplankton development would be achieved in Lago Paranoá remains unknown. Although silver carp grazing pressure increases with fish biomass, it will be counteracted by (1) a concomitant increase in phosphorus supply to algae due to fish excretion and (2) a decrease in zooplankton grazing pressure resulting from increased fish predation.

Therefore, our working hypotheses are:

i) Below a certain stocking biomass threshold, although fish side effects of supplying nutrient to phytoplankton by excretion and reducing zooplankton grazing pressure by predation would be minimized, silver carp grazing may not be sufficiently strong enough to control the development of nuisance algae population.

ii) Above this stocking biomass threshold, such fish side effects will be more important than the direct beneficial fish grazing effect on algae, and silver carp will fail in controlling undesirable cyanobacteria.

The present study aims at determining, in a situation as close as possible to reservoir conditions, created within large replicated littoral limnocorrals, the "optimal" range of silver carp stocking biomass at which fish grazing on net-phytoplankton will result in water quality improvements, i.e., a reduction in phytoplankton, particularly cyanobacteria. Predictions of silver carp population dynamic evolution in Lago Paranoá will also be made based on fish growth rate data obtained in enclosures (Chapters IV and V) and net-cages maintained in this ecosystem (Chapter III).

IV.2. Materials and Methods

During the rainy season of 1996, ten large littoral limnocorrals (5 m x 16 m, 1.8 m mean depth, 80 m^2 surface area, and 50 m^3 volume) were constructed within a small bay located in the experimental site, nearby the Northern Sewage Treatment Plan (Figure 4.1). On October 1-9, limnocorrals were formed by erecting four walls made of PVC- reinforced sheets, which completely isolated the water mass from the adjacent lake (see details of limnocorrals construction in Chapter II). All limnocorrals were set in a linear array extending from a depth of about 1.5 m to the lake shore (Figure 4.2). Limnocorrals were completely opened to sediment, but isolated from lake shoreline.

During the process of construction, one side of each limnocorral was left opened but covered with a fish net (12 mm mesh) to allow water but prevent fish exchange between the limnocorral and the adjacent reservoir. On October 24-30, all fishes confined inside each recently erected limnocorrals were captured by seining. The same fishing effort, i.e., seining 5 times in each limnocorral with a 5 mm mesh seine, was applied until limnocorrals were considered fishless. The total number of fishes captured in this pre-experimental condition is shown in Table IV.1. The fish community was widely dominated by tilapia (including *Tilapia rendalli* and *Oreochromis niloticus*), which in average represented over 80% of the total fish density (excluding fish larvae, which were not identified to the species level). On November 01, after verifying that all limnocorrals had reached similar limnological conditions, they were all totally closed from the adjacent reservoir by raising their open side.

Figure 4.1: Location of experimental site in the vicinity of the Northern Sewage Plant.

Figure 4.2: Overview of Experimental Site showing the set of 10 limnocorrals.

Based on analyses of nutrient content of sediment performed on November 04, limnocorrals were divided into two groups (blocks), limnocorrals 1-5 having nutrient richer sediment (P around 2,000 mg/kg of wet sediment), and limnocorrals 6-10 having nutrient poorer sediment (P <1,000 mg/kg of wet sediment). Five silver carp biomass treatments ranging from 0 to 60 g/m^3 were randomly assigned to limnocorrals within each block, as shown in Table IV.2. Based on the average tilapia abundance captured in limnocorrals (Table IV.1), an equivalent tilapia biomass of 25 g/m^3 was re-introduced into eachl limnocorral as adult fishes (~100 g) collected in the neighborhood using cast-nets. A ratio of 4:1 between *Tilapia rendalli* and *Oreochromis niloticus* was used as typical of this reservoir area. As a certain amount of fish larvae was assumed to have escaped from initial seining and adult tilapia would certainly spawn in limnocorrals, the fish community from the lake was considered to be well represented by adult tilapia plus fish larvae. Numbers and weights of tilapias stocked in each limnocorral are also shown in Table IV.2. An increasing biomass of small free-roaming silver carp (~ 40 g each) was used according to treatments. Numbers and weights of silver carp stocked in each limnocorral are also shown in Table IV.2. The procedure of stocking fish into all limnocorrals took 2 days (November 4-5). The experiment begun on November 5 by monitoring limnological characteristics in each limnocorral as well as at two adjacent sampling stations in the reservoir at weekly intervals.

Using a small boat, a 10-litre surface composite sample was collected from each limnocorral, by pooling samples from 3 stations within the limnocorrals for temperature, pH, turbidity, total alkalinity, dissolved oxygen, conductivity, total phosphorus (TP), orthophosphate (PO_4P), ammonia, nitrate, total Kjeldahl nitrogen,

Table IV.1: Fish biomass removed from limnocorrals before starting the experiment.

	Limnocorral / Volume (m^3)										
	1	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**Mean**
	48.0	**56.0**	**61.6**	**66.4**	**71.2**	**60.8**	**58.4**	**47.2**	**52.0**	**47.2**	**56.9**
Fish larvae (g)	3000	2995	2780	1840	2255	1475	2810	1900	1695	2135	2289
***Tilapia* weight (g)**	975	815	1110	995	110	4585	165	1815	1205	845	1262
***Tilapia* number**	22	19	16	14	2	78	3	22	33	19	23
***Tilapia* mean weight (g)**	44.3	42.9	69.4	71.1	55.0	58.8	55.0	82.5	36.5	44.5	56.0
Other fish species (g)	440	535	65	1175	0	495	15	0	165	80	297
Overall weight (g)	4415	4345	3955	4010	2365	6555	2990	3715	3065	3060	3848
***% tilapia* weight ***	68.9	60.4	94.5	45.9	100	90.3	91.7	100	88.0	91.4	83.1
Total Biomass (kg/ha)	552	543	494	507	296	819	374	464	383	541	497
Total Biomass (g/m^3)	92.0	77.6	64.2	60.4	33.2	107.8	51.2	78.7	58.9	64.8	68.9

Legend: * share of tilapia in total fish abundance, excluding larvae not identified at species level.

total nitrogen (TN), total chlorophyll-a, chlorophyll-a fraction <15 µm, phytoplankton primary productivity (dark and light bottles incubated during 2 hours around noon), phytoplankton and zooplankton counts (APHA, 1985). A sub-sample (500 ml) of this surface water from each limnocorral was concentrated to 30 ml by sedimentation over 24 hours in standard glass cones (2000 ml) and preserved in 0.5% Lugol's iodine solution to obtain a size range spectrum of particles distribution (APHA, 1985). A general counting procedure with a Coulter Multisizer® using an orifice tube size of 70 µm was conducted as described in Rahmatullah (1992). Secchi depth and fish mortality were monitored weekly.

Table IV.2 : Experimental design, initial nutrient content of sediment and fish biomass stocked to each limnocorral.

	Treatments / Limnocorral											
	SC10 1	**SC60 2**	**SC40 3**	**CTL 4**	**SC20 5**	**CTL 6**	**SC60 7**	**SC40 8**	**SC10 9**	**SC20 10**	**Lake 11**	**Lake 12**
TN in sediment (mg/kg)	512	561	465	621	458	502	528	357	276	388	342	754
TP in sediment (mg/kg)	1916	1922	2264	1974	1951	1177	641	240	200	170	168	1848
Number of *tilapia* re-stocked	13	13	13	13	13	13	13	13	13	13	-	-
Weight of *tilapia* re-stocked (g)	1210	1405	1580	1335	1420	1330	1225	1270	1375	1275	-	-
Tilapia biomass (kg/ha)	151	176	198	167	178	166	153	159	172	159	-	-
Tilapia biomass (g/m^3)	25.2	25.1	25.6	20.1	19.9	21.9	21.0	26.9	26.4	27.0	-	-
Number of silver carp stocked	20	80	50	0	30	0	80	50	30	20	-	-
Weight of silver carp stocked (g)	580	3065	2410	0	1240	0	3300	1985	575	945	-	-
Silver carp biomass (kg/ha)	73	383	301	0	155	0	413	248	72	118	-	-
Silver carp biomass (g/m^3)	12.0	54.7	39.1	0	17.4	0	56.5	42.1	11.1	20.0	-	-
Total fish Biomass (kg/ha)	224	559	499	167	333	166	566	407	244	277	-	-
Total fish Biomass (g/m^3)	37.2	79.8	64.7	20.1	37.3	21.9	77.5	69.0	37.5	47.0	-	-

Legend: CTL=control without silver carp; SC10=silver carp biomass of 10 g/m^3; SC20= silver carp biomass of 20 g/m^3; SC40= silver carp biomass of 40 g/m^3; SC60= silver carp biomass of 60 g/m^3; LAKE= surrounding lake sampling point.

Verification of the complete isolation of each limnocorral from the adjacent reservoir was performed by scuba divers twice a month.

The experiment lasted 35 days, after which all fishes were recovered from each limnocorral using a combination of rotenone (1 mg/l) and seining (Timmons *et al.*,

1979). Individual weights of silver carp recaptured from limnocorrals were recorded. Tilapia (2-3 per limnocorral) were individually tagged using fine, pliant, colored wire inserted at the base of the anterior three hard rays of the dorsal fin in order to evaluate growth rates throughout the experiment. Nutrient content of sediment from each limnocorral and the adjacent lake was analyzed at the end of the period.

Repeated-measures ANOVAs with randomized block design was applied to log-transformed data to detect differences between treatments for each limnological variable. Contrasts between significantly affected treatments were made *a posteriori* using Tukey Multiple Comparison Test. Due to low replication and statistical power, a probability level of alpha < 0.10 was chosen to reduce the chance of making the type II error of failing to reject a false null hypothesis. Initial differences between limnocorrals for all limnological variables as well as for nutrient content of sediment were tested using One-way ANOVA. All computations were performed using SYSTAT (Wilkinson, 1989).

IV.3. Results

All limnocorrals (excluding the 2 adjacent lake sampling stations) had similar initial values for all limnological variables (Table IV.3). The only exception was nanno-phytoplankton chlorophyll-a, which was higher in limnocorrals stocked with 20 g/m^3 of silver carp (designated SC 20) than in limnocorrals stocked with 10 g/m^3 of silver carp (designated SC 10).

Table IV.3: Probability values from one-way ANOVAs performed on initial values (day 0) between all treatments.

Limnological Variables	Probability Value (P)
Total Zooplankton	0.117
Primary Productivity	0.876
Nanno-phytoplankton Chlorophyll-a	0.065*
% Net-phytoplankton Chl-a fraction	0.223
Total Chlorophyll-a	0.394
Filamentous algae abundance	0.950
Colonial algae abundance	0.419
Total Alkalinity	0.949
Total Kjeldahl Nitrogen (TKN)	0.616
Nitrate	0.987
Orthophosphate	0.809
Total phosphorus	0.620
Conductivity	0.534
Turbidity	0.614
Secchi depth	0.803
Total Nitrogen in sediment	0.334
Total Phosphorus in sediment	0.988

Legend: *significant, at α=0.10 level.

Table IV.4 shows the results from repeated-measures ANOVAs performed on data from all treatment levels during the experimental period (days 7, 14, 21, 28, 35). There were significant silver carp biomass effects on total zooplankton (P=0.021), percentage of net-phytoplankton chlorophyll-a (P=0.023), and colonial algae abundance (P=0.093). For these variables, stocking silver carp was associated with a reduction in large-sized plankton filtered by fish. The magnitude of such effect increased with silver carp biomass, as net-phytoplankton and zooplankton abundance levels were lower at higher fish stocking rates.

Table 4: Probability values from Repeated-measures ANOVAs performed on data for all treatments during the 35 days of the experimental period (days 7, 14, 21, 28 and 34).

Limnological Variables	Silver Carp Biomass Effect : P (Treatment)
Total Zooplankton	0.021*
Primary Productivity	0.785
Nanno-phytoplankton Chlorophyll-a	0.128
% Net-phytoplankton Fraction	0.023*
Total Chlorophyll-a	0.638
Filamentous Algae Abundance	0.816
Colonial Algae Abundance	0.093*
Total Alkalinity	0.934
Total Kjeldahl Nitrogen	0.579
Nitrate	0.873
Orthophosphate	0.999
Total Phosphorus	0.806
Conductivity	0.904
Dissolved Oxygen	0.257
Turbidity	0.568
PH	0.520
Secchi Depth	0.934

Legend: *significative, at the alpha level used (0.10).

The evolution and overall mean values per treatment level for these planktonic variables significantly affected by silver carp biomass are shown in Figures 4.2-4.4. In general, the introduction of lower silver carp biomass (10 and 20 g/m^3) caused only a significant reduction in colonial algae abundance by 30% and 54% respectively, with no effect on the percentage of net-phytoplankton chlorophyll-a and zooplankton density. By increasing silver carp biomass up to 40 and 60 g/m^3, the removal of colonial algae was intensified up to 48% and 81%, respectively. However, at these higher fish biomass levels, although the percentage of net-phytoplankton chlorophyll-a was also reduced by 30% and 39%, zooplankton was suppressed by 74% and 77%, respectively.

The impact of increasing silver carp biomass on planktonic community is well illustrated by the gradual shift towards small particles in size distribution of particles

between 2 and 20 µm in diameter (Figure 4.5). Although this figure represents the overlay of the different average size distribution curves through time obtained for each treatment, which have not been statistically tested, it clearly illustrates not only the tendency of reducing particles size in the water as a consequence of silver carp filtering activity but also its direct relationship with fish stocking rate.

However, it has to be emphasized that the Coulter Multisizer® calculates the bio-volume of particles using particles sizes estimated by converting all particles dimensions, irrespective of size and shape, to idealized sphere (Rahmatullah, 1992). By expressing particle size in terms of equivalent spherical diameter (ESD), the real linear dimensions of the majority of phytoplankton species are not respected, particularly for those which are elongated in shape. For instance, a particle considered by the Coulter Multisizer as having 4 µm in diameter may be an ellipsoid-shaped algae with 6 µm in largest dimension. As far as particle retention by silver carp filtrating apparatus is concerned, despite the lack of precision on the real largest dimension of the particles counted by the Coulter Multisizer, the ranges of particle diameters suppressed and enhanced by fish presence are perfectly consistent with the fish gill rakers mesh size of 11-26 µm (Hampl *et at.*, 1983) or 8-14 µm (Starling, 1989), as well as with laboratory feeding experiments showing no filtration for spherical particles smaller than 10 µm in diameter by silver carp (Herodek *et al.*, 1989; Smith,1989).

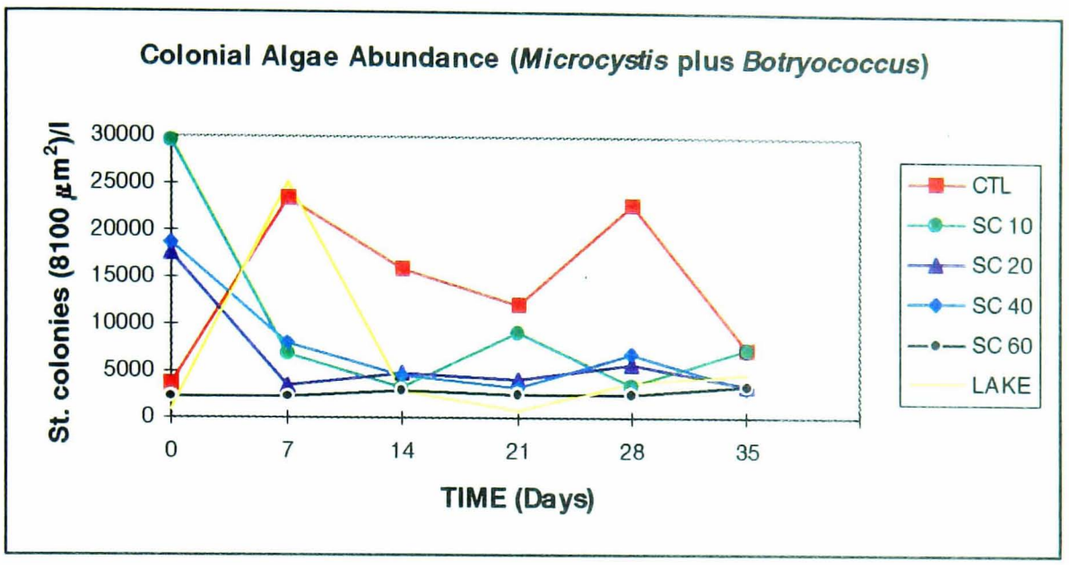
Total fish biomasses recovered from limnocorrals at the end of the experiment are shown in Table IV.5. Despite the reduction in number and biomass of adult tilapia initially stocked into limnocorrals, probably as a result of predation by abundant aquatic birds, the original tilapia biomass stocked into all limnocorrals increased considerably as a consequence of fish spawning. The amount of fish larvae was on

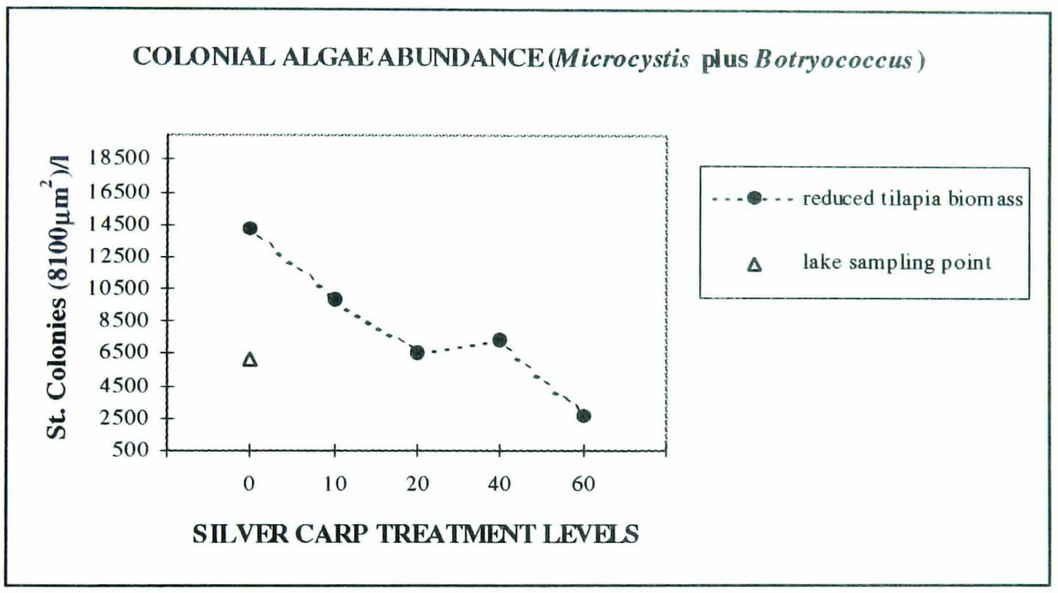
average 3 times higher in the absence of silver carp (CTL vs. SC limnocorrals), which suggests the existence of a possible negative influence of silver carp on tilapia population, as will be discussed later. The scarcity of fish other than tilapia in all limnocorrals at the end of the experimental period, representing less than 8% of the total fish biomass, confirms that the pre-experimental removal of reservoir fishes was very successful. The general increase in the overall reservoir fish biomass during the experiment, irrespective of silver carp presence and stocking density, illustrates the high availability of food resources in all limnocorrals, evidence that the carrying capacity of the systems had not been reached.

By recovering some of the tagged tilapia stocked into each limnocorral, it was possible to estimate the growth rate of fish from the reservoir. At the end of the experiment, 24 out of 30 marked tilapias were recovered and had their total initial and final weights recorded as 2,670 g and 2,870 g, respectively (111.25 g and 119.58 g, initial and final mean individual body weights). Based on the overall gain of weight of 200 g or 4.76 g/d, the individual growth rate can be calculated as 0.2 g/day or 0.17 % of body weight/day. Such low growth rates can be explained by the fact that they were obtained from adult fishes during a spawning period, when energy is preferentially allocated to reproduction rather than to growth.

Since silver carp gained weight in all limnocorrals, irrespective of treatment levels, there was an overall increase in fish biomass of around 100% during the experiment (Table IV.6). Nevertheless, the original contrast between treatments was maintained in all situations when taking initial, final or average fish biomass into account.

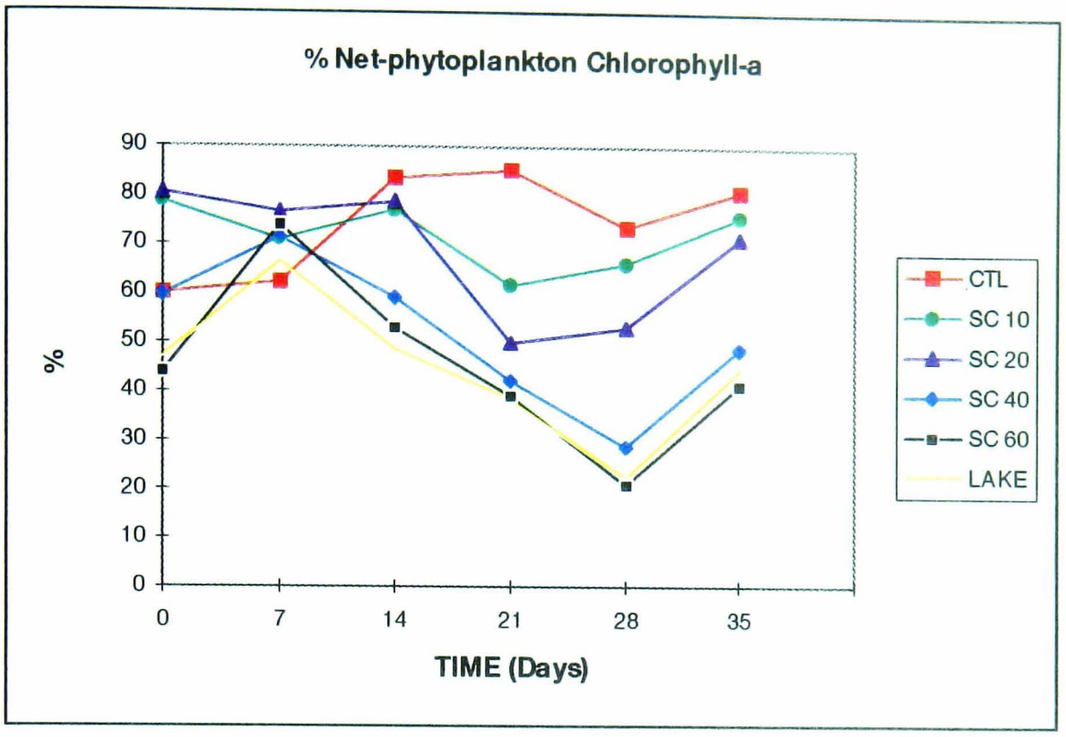
Almost no silver carp mortality was recorded during the experiment, as only 9 fishes out of 360 were not recovered at the end, making survivorship rate higher than 95% (Table IV.7). Silver carp growth rates were generally inversely proportional to fish stocking rates, as final fish weight increased by 134.4%, 139.5%, 88.4% and

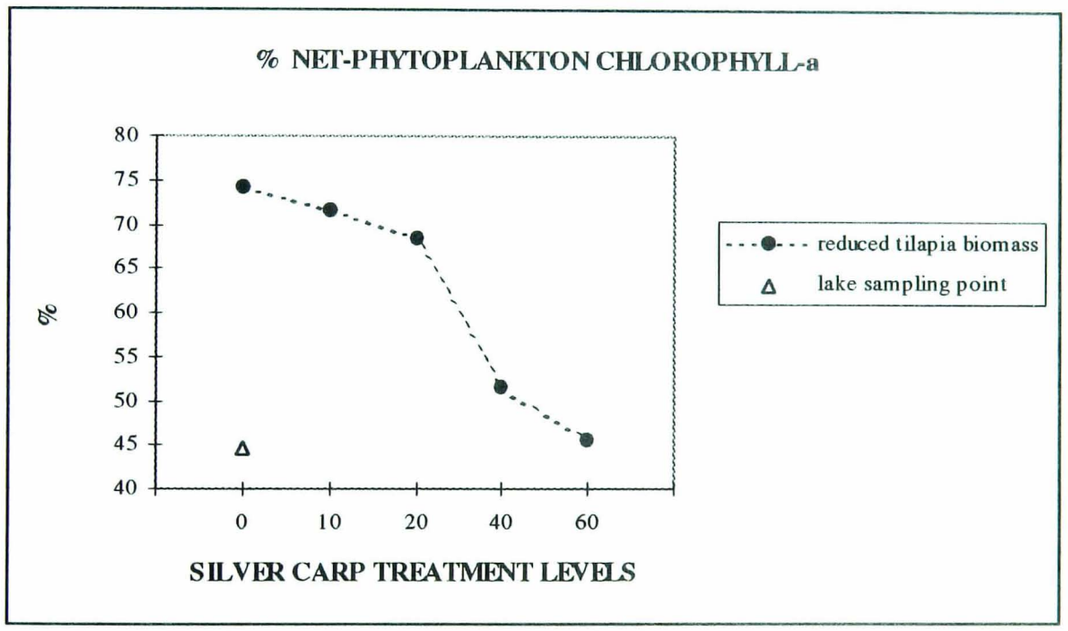




Treatments:		% Reduction
CTL	(control or reduced tilapia biomass without silver carp)	0
SC 10	(control plus 10 g/m^3 silver carp biomass)	30.6
SC 20	(control plus 20 g/m^3 silver carp biomass)	54.2
SC 40	(control plus 40 g/m^3 silver carp biomass)	47.8
SC 60	(control plus 60 g/m^3 silver carp biomass)	81.2

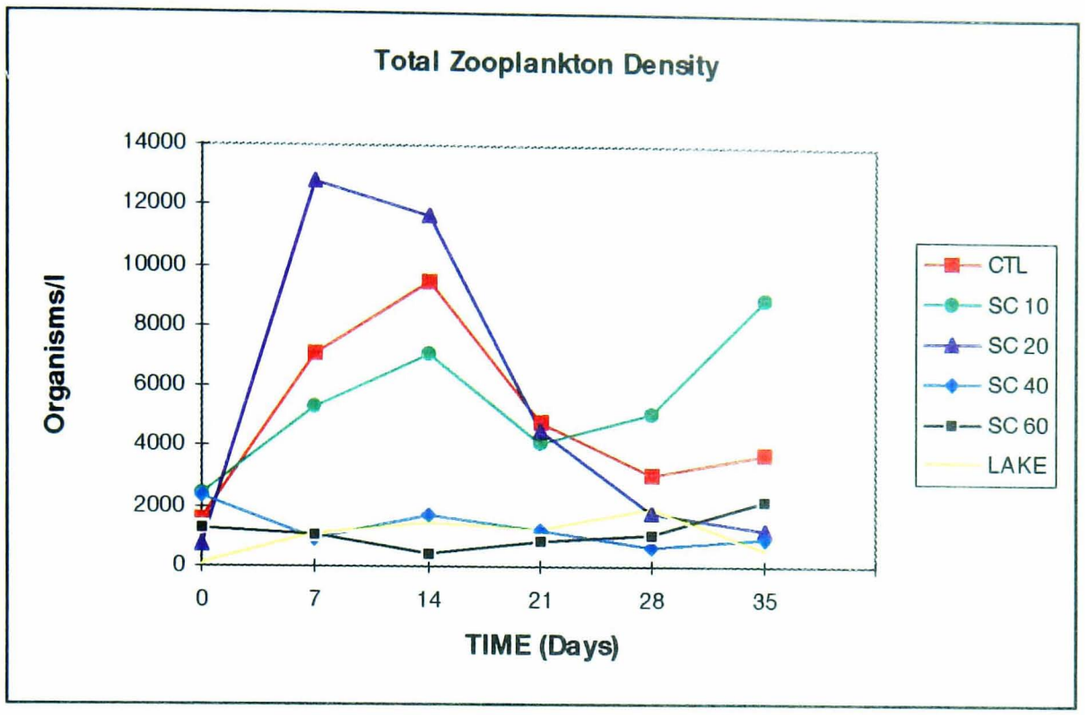
Figure 4.3: Evolution and mean response of colonial algae abundance over the experimental period for each silver carp biomass treatment level.

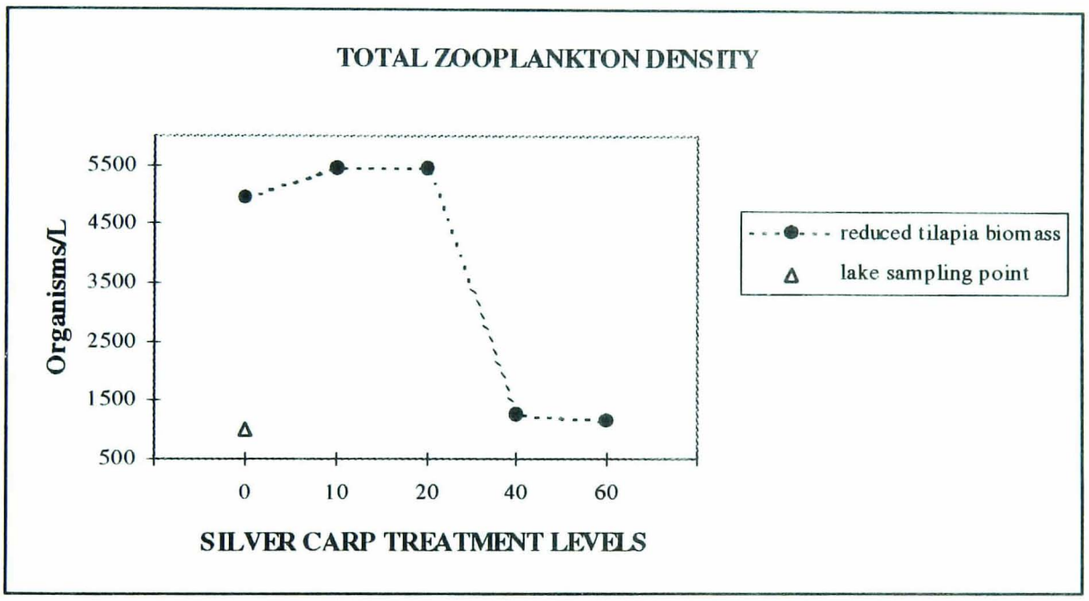




Treatments:		% Reduction
CTL	(control or reduced tilapia biomass without silver carp)	0
SC 10	(control plus 10 g/m^3 silver carp biomass)	3.4
SC 20	(control plus 20 g/m^3 silver carp biomass)	7.9
SC 40	(control plus 40 g/m^3 silver carp biomass)	30.5
SC 60	(control plus 60 g/m^3 silver carp biomass)	38.8

Figure 4.4: Evolution and mean response of % net-phytoplankton chlorophyll-a over the experimental period for each silver carp biomass treatment levels.





Treatments:		%Reduction
CTL	(control or reduced tilapia biomass without silver carp)	0
SC 10	(control plus 10 g/m^3 silver carp biomass)	0
SC 20	(control plus 20 g/m^3 silver carp biomass)	0
SC 40	(control plus 40 g/m^3 silver carp biomass)	74.2
SC 60	(control plus 60 g/m^3 silver carp biomass)	76.7

Figure 4.5: Evolution and mean response of total zooplankton abundance over the experimental period for each silver carp biomass treatment.

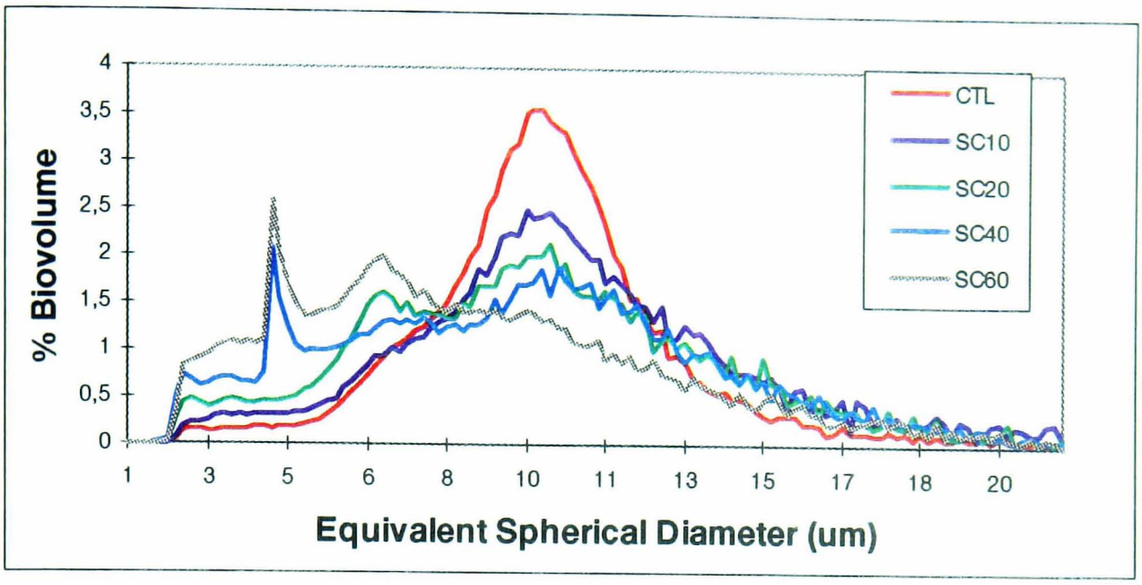


Figure 4.6: Overlay of average particle volume distribution (in %) between silver carp biomass treatment levels over the limnocorral experiment (days 7 to 35) for the range of particle diameter of 2 to 20 µm)

Table IV.5 : Fish recovered (body weight, density and biomass) from each limnocorral at the end of the experiment.

	Treatments / Limnocorral										
	CTL 4	**CTL 6**	**SC10 1**	**SC10 9**	**SC20 5**	**SC20 10**	**SC40 3**	**SC40 8**	**SC60 2**	**SC60 7**	**Overall Mean**
Weight /number adult tilapia (g)	620 (4)	1295 (12)	520 (6)	1315 (12)	1045 (8)	890 (9)	1105 (9)	760 (7)	525 (5)	190 (2)	827 (7)
Biomass adult tilapia (g/m^3)	9.3	21.3	10.8	25.3	14.7	18.9	17.9	16.1	11.1	3.3	14.9
Weight/number small tilapia (g)	420 (14)	1035 (25)	30 (1)	335 (10)	35 (1)	310 (7)	340 (13)	0	350 (7)	95 (3)	292 (8)
Weight Other fish (g)	0	250	0	140	0	380	325	520	530	0	215
Weight Fish larvae (g)	2770	3400	1260	255	1100	640	790	1540	720	1200	1368
Total weight fish from lake(g)	3810	5980	1810	2045	2180	2220	2560	2820	2125	1485	2707
Total lake fish biomass (g/m^3)	57.4	98.4	37.7	39.5	30.6	47.0	41.6	37.4	37.9	25.4	45.3
Total lake fish biomass (kg/ha)	476	748	226	257	273	278	320	221	266	186	325

Table IV.6: Overall fish biomass changes during the course of limnocorrals experiment.

	Treatments/Limnocorrals					
	CTL 4 & 6	SC 10 1 & 9	SC 20 5 & 10	SC 40 3 & 8	SS 60 2 & 7	Mean
Total lake fish stocked biomass (g/m^3)	21.0	25.8	23.5	26.3	23.1	23.9
Total final lake fish biomass (g/m^3)	77.9	65.8	38.8	39.5	31.7	50.7
Total average lake fish biomass (g/m^3)	49.5	45.8	31.2	32.9	27.4	37.9
Total silver carp stocked biomass (g/m^3)	0	11.6	18.7	40.6	55.6	25.3
Total final silver carp biomass (g/m^3)	0	27.2	44.8	76.5	88.1	47.3
Total average silver carp biomass (g/m^3)	0	19.4	31.8	58.6	71.9	36.3
Overall fish stocked biomass (g/m^3)	21.0	37.4	42.2	66.9	78.7	49.2
Overall final fish biomass (g/m^3)	77.9	93.0	83.6	116.0	119.8	98.1
Overall fish average biomass (g/m^3)	49.5	65.2	63.0	91.5	99.3	73.7

58.4% respectively for stocking rates of 10, 20, 40 and 60 g/m^3. The silver carp growth rates obtained can be considered very high when taking into consideration that limnocorrals were closed systems with no water exchange and additional plankton supply from external lake water.

Table IV.7 : Silver carp growth rates during the course of limnocorral experiment.

	Treatments/Limnocorral								
	SC 10 1	SC 10 9	SC 20 5	SC 20 10	SC 40 3	SC40 8	SC 60 2	SC 60 7	Mean
Total gain of weight (g)	660	915	1515	1455	1930	1915	1815	1905	1514
Total gain of weight (g/d)	15.4	21.3	35.2	33.8	44.9	44.5	42.2	44.3	35.2
Fish mean initial weight (g)	29.0	28.8	41.3	31.5	48.2	39.7	38.3	41.3	37.3
Growth rate (g/d/fish)	0.77	1.06	1.17	1.13	0.90	0.89	0.53	0.55	0.88
Growth rate (% w.w/fish/d)	1.83	2.29	1.92	2.24	1.41	1.62	1.11	1.09	1.44
Mortality rate (%)	5	5	6.7	6.7	0	2	1.3	1.3	3.5

IV.4. Discussion

The potential use of filter-feeding fish to control nuisance net-phytoplankton directly by grazing, mainly in tropics and subtropics, has frequently been suggested in the literature (Nilssen, 1984; Drenner *et al.*, 1987; Lazzaro, 1987; Gophen, 1990a; Crisman & Beaver, 1990; Starling & Rocha, 1990). This recommendation is based on both (a) laboratory and field evidences showing the effective consumption and reduction of net-phytoplankton by filter-feeding fish, and (b) common observation that plankton communities of tropical and subtropical lakes are dominated by large-sized algae (mainly cyanobacteria), small-sized poorly efficient microzooplankton grazers (mainly rotifers), and lacked large-sized daphinids (Nilssen, *op.cit.*). In such extreme situation of weak zooplankton:phytoplankton trophic link, where detritivory dominates over grazing in the food chain, the control of undesirable abundant net-phytoplankton could not be achieved by the "classical" biomanipulation approach based on the enhancement of zooplankton grazing pressure on phytoplankton.

Despite the potential for an alternative approach in biomanipulation, the success of controlling phytoplankton biomass with filter-feeding fish is not so commonly reported in the literature. By reviewing 15 studies on the effects of filter-feeding fish on phytoplankton biomass (9 of them involving silver carp, i.e. 60%), Smith (1988) found only 4 studies (i.e., 27%), among which 2 used silver carp, in which algal biomass was significantly reduced and 7 (47%) in which the abundance of algae significantly increased (5 of them involving silver carp). No significant effect or insufficient replication was found in the remaining 4 studies.

Drenner *et al.* (1996) reached the same general conclusion in a review of 29 studies covering 48 experiments on the impacts of different omnivorous filter-feeding

fishes. Phytoplankton abundance, biomass and/or productivity were altered by fish presence in 43 experiments (90%). Omnivorous fish impacts were undoubtedly detrimental (water quality deterioration towards eutrophication) in 30 experiments (63%), and beneficial (net-phytoplankton reduction without overall algal biomass increase) in only 8 of them (17%). However, it must be noticed that 21 of the considered experiments (44%) tested some bottom-feeding fish species, such as common carp, bream, and roach, well known for their frequent water quality deterioration via sediment resuspension, and only 7 experiments (15%) involved silver carp. This may give a biased picture of omnivorous fishes, as highly specialized open-water filter-feeding silver carp, bottom-feeding fishes, even bighead carp and gizzard shad cannot be expected to have similar impacts. Indeed, in contrast with these less efficient filter-feeders equipped with more widely spaced gill rakers, silver carp combines a powerful suction ability with a highly developed filtering apparatus for collecting particles larger than 10 μm (Herodek *et al,* 1989; Smith, 1989).

A contrasting picture emerges when reviewing the literature concerning attempts to use silver carp as a biomanipulation tool (Table IV.8). In 14 of 26 studies (54%) involving silver carp addition to enclosures, fish ponds, lakes and reservoirs, silver carp presence was associated with water quality improvements, i.e., net-phytoplankton reduction without a simultaneous increase in overall algal biomass. Among these 14 successful studies, 9 studies (64%) were performed in the Tropics or Subtropics, 11 studies (79%) referred to situations of net-phytoplankton dominance or cyanobacteria blooms, and 10 studies (71%) involved the addition of low to moderate silver carp biomass (< 1,200 fish/ha; < 70 g/m^3 or 900 kg/ha). Only 5 out of these 26 studies (19%), which involved the addition of a high silver carp stocking density (2000-12000 fish/ha) to temperate fish ponds, reported overall detrimental impacts.

Other 7 studies (27%) showed contradictory results, or a combination of beneficial and detrimental effects.

As also frequently observed in other tropical and subtropical ecosystems (Nilssen, 1984), Lago Paranoá has reached a high eutrophic condition where phytoplankton is largely dominated by nuisance bloom-forming large-sized cyanobacteria species not subdued to a considerable grazing pressure from abundant microzooplankton (Pinto-Coelho, 1983). In such situation, stocking silver carp at low to moderate biomass would improve water quality by controlling nuisance net-phytoplankton abundance without enhancing total algal biomass. The well known side effect of increasing nanno-phytoplankton abundance as a result of the concomitant removal of its competitors (net-phytoplankton) and grazers (zooplankton) by silver carp feeding (Milstein, 1992) is therefore of limited relevance in the case of Lago Paranoá.

Convincing beneficial effects of silver carp have also been recently obtained in a similar experimental study conducted in China (Bangxi *et al.*, 1993). The impacts of six silver carp biomass levels (0, 10, 15, 20, 25 and 40 g/m^3) cross-classified with five common carp biomass levels (0, 30, 40 50 and 75 g/m^3) were evaluated in thirty 14.3 m^3 enclosures (incomplete design of 15 treatment combinations, with duplicate replication) monitored over 34 days in a fish farming reservoir. Total chlorophyll-a, total phytoplankton density and biomass and zooplankton density and biomass were substantially reduced by silver carp at all stocking densities (except at 10 and 20 g/m^3 without common carp addition) when compared with the control limnocorrals.

Overall values for total chlorophyll-a, total phytoplankton densities and biomass and zooplankton numbers were reduced in average by 39%, 46%, 41% and 40% respectively, at contrasting silver carp biomasses and regardless of common carp biomass level. Comparatively higher suppression of planktonic organisms (an average reduction of 50%, 65%, 60% and 43% respectively for chlorophyll-a, total algal

Table IV.8: Silver carp impacts in different aquatic ecosystems reported by experimental and field studies.

Ref.	System Country	Limnological characteristics	Silver carp numbers/ha and biomass (kg/ha) [g/m^3]	Chemical variables	Effects on water quality Phytoplankton biomass and production	Zooplankto abundanc
1	Enclosures (9 m^3) Poland	Eutrophic (⇧ *Microcystis)*	14400, 43200 (450, 1350) [31, 94]	?	↓ biomass (4.5 x, 4.5 x)	↓ biomass (4.5 x, 16 x)
2	Fish ponds Germany	?	3000-12000 (?) [?]	?	↑ biomass ↓ % cyanobacteria	?
3	Carp ponds Poland	?	4000-12000 (?) [?]	?	↑ biomass and net-production	?
4	Carp ponds Poland	?	4000-12000 (895-1811) [?]	↑ nutrient recycling	↑ biomass and production	↓ abundance
5	Reservoir Israel	(⇧ Peridinium*)*	400 (?) [?]	?	↓ net-phytoplankton biomass	
6	Carp ponds (0.2ha) Poland	Fertilized	1500-12000 (> 750-6000) [> 95-750]	↑ oxygen levels	↑ biomass and net-production	?
7	Reservoir (341 ha) Czech Republic	Eutrophic (algal bloom)	91 (75) [1.7]	?	↓ algal bloom (*Microcystis*)	?
8	Polyculture ponds Israel	Fertilized	1300-2600 (70-140) [?]	?	↑ biomass ↓ algal size	↓ abundance
9	Tank (1m^3), divided USA	Heavily fertilized	110000 (>1400) [>110]	?	no effect	↓ abundance
			220000 (>2800) [220] *[1]	?	↓ biomass (99%)	↑ *Daphnia*
10	Lake Kinneret Israel	Blooms of *Peridinium*	total = 13 x 10^6 fish (?) [?]	?	↓*Peridinium* and ↑ nanoplankton	↓ abundance
11	Catfish ponds USA	?	3000-12000 (?) [?]	↑ NO_3-N and NH_4-N	↑ biomass	↓ abundance
12	Lake New Zealand	Eutrophic (*Microcystis*)	3000 (100-2000)	↑ secchi depth and O_2	↓ algal bloom (*Microcystis*)	?
13	Polyculture ponds Israel	?	400-600 (360-520)	?	↓net-phytoplankton ↑ nano-plankton	?

Table IV.8: Cont.

Ref.	System Country	Limnological characteristics	Silver carp numbers/ha and biomass (kg/ha) [g/m^3]	Effects on water quality: Chemical variables	Phytoplankton biomass and production	Zooplankton abundance
14	Carp ponds (0.25ha) Germany	?	2000, 4000, 6000 (912, 1644, 2496) [91, 164, 250]	?	↑ biomass and net-production	↓ abundance
15	Tanks (5.5 m^3) USSR	Fertilized pond water	≈25000-35000 (2500, 4000, 7000) [250, 400, 700]	?	↓biomass, ↑production,nanoplankton	↓ abundance
16	Fish ponds Hawaii	Fertilized	75 (7.4) [0.7] 3750 (1125) [113]	↑ particulate C and , N	↑ biomass ↓net-phytoplankton biom.	? ?
17	Reservoirs (20 ha) Israel	Blooms of *Peridinium*	235-4350 (421-12000) [94-300]	↓organic matter	↓biomass and algal bloom	↓ abundance
18	Lake (1200 ha) China	Eutrophic (*Microcystis*)	1100 (?) [?]	?	↓biomass, algal bloom ↑ primary production	?
19	Enclosures (2 m^3) Brazil	Eutrophic reservoir (4000 ha)	127300 (3000) [117]	no effects	no effects	↓ abundance
20	Enclosures (50 m^3) Israel	Reservoir (325 ha)	1600 (?) [?]	↓ clogging capacity (75%)	↓ biomass	?
21	Enclosures (50m^3) Israel	Irrigation reservoir algal blooms	5200 (hyb.) (811) [41] 2000-4000 (304-608) [25-50]	?	↓ particles (20-600 µm), algal size, net-plankton	↓ abundance
22	Enclosures (14.3 m^3) China	Fish ponds common carp	7000-28000 (262-980) [0, 10, 15, 20, 25, 40]	↓ total phosphorus	↓ biomass and density	↓ biomass
23	Enclosures (6.5 m^3) Brazil	Eutrophic reservoir (4000 ha)	16000 (850) [40]	no effects	↓ numbers (35%), biomass (20%), production (80%)	↓ microzoopl. Abundance
24	Fish ponds China	?	? (≈ 240, 880, 1260) [12, 44, 63] *[2]	?	↑ nanno-picoplankton, ↓ biomass, production	↓ abundance
25	Pond (0.8 ha) USA	Eutrophic (algal blooms)	1875 / 1250 (141 / 42) [≈14 / 4] *[3]	↑ turbidity, TP, and TN	↑ biomass, nanoplankton ↓ net-plankton., blooms	↓ abundance
26	Limnocorral (50 m^3) Brazil	Eutrophic reservoir (4000 ha)	0-10000 (75, 150, 300, 400) [10, 20, 40, 60]	no significant effects	↓ biomass net-plankton, *Microcystis* abundance	↓ abundance

Legend: **1**-Kajak *et al.* (1975); **2**-Barthelmes (1977); **3**-Januszko (1978); **4**-Opuszynski (1979); **5**-Leventer (1981); **6**-Piotrowska-Opuszynsza (1984); **7**-Hochman *et al.* (1985); **8**-Milstein et al. (1985a,b); **9**-Smith (1985); **10**-Spataru & Gophen (1985); **11**-Burke *et al.* (1986); **12**-Carruthers (1986); **13**-Milstein *et al.* (1988); **14**-Barthelmes (1989); **15**-Vyrbonov (1989); **16**-Laws & Weisburd (1990); **17**-Leventer & Teltsch (1990); **18**-Miura (1990); **19**-Starling & Rocha (1990); **20**-Teltsch *et al.* (1991); **21**-Sagi (1992); **22**-Bangxi *et al.* (1993); **23**-Starling (1993a), **24**-Yu-Bao (1993); **25**-Lieberman (1996); **26**-Present study; *[1]fish confined in half of the tank; *[2] bighead+silver carp ; *[3] bighead+silver carp/silver carp.

densities and biomass and zooplankton densities) occurred at the highest silver carp biomass (40 g/m^3). The best results in terms of water quality improvements were generally obtained at low to moderated silver carp stocking rate (20 to 40 g/m^3).

The desirable range of silver carp biomass to be maintained in a given ecosystem to successfully control net-phytoplankton will depend fundamentally on the structure of the planktonic community, as emphasized here. However, it is of common sense in polyculture that silver carp abundance in fish ponds should not exceed 1,000 fish/ha, threshold above which there is a risk of ichthyoeutrophication (Opuszynski, 1981; Milstein, 1992). Milstein *et al.* (1988) performed a series of experiments in 1983, 1984 and 1985, respectively involving the addition of decreasing silver carp stocking densities of 1,300-2,600, 600 and 400 fish/ha, but increasing biomasses of 70-140, 360 and 520 kg/ha (by using progressively larger fishes). As the intensity of fish effects decreased from 1983 to 1985, despite the increase in the fish biomass, they suggested that the influence of silver carp seems to depend more on the number than on the body size of stocked fish, probably for allometric reasons.

Although the relative importance of silver carp stocking number versus biomass was not investigated in the present study, as the biomass (in terms of g/m^3) was considered the most correct and realistic way to express fish abundance, only positive effects of silver carp were obtained at relatively high fish densities ranging from 2,500 fish/ha (in SC10 limnocorrals) and 10,000 fish/ha (in SC60 limnocorrals). The same picture emerges from the enclosures experiment recently conducted in China (Bangxi *et al.*, 1993), where only water quality improvements were observed at silver carp stocking densities ranging from 6,993 to 27,972 fish/ha.

Based on the results from the present study and taking into account some of the successful examples in the literature (Table IV.8), it may be concluded that the silver carp stocking rate to be maintained in Lago Paranoá should not be less than 20 g/m^3

(or equivalent to 200-300 kg/ha on an areal basis) and should not exceed 70 g/m^3 (or 700-900 kg/ha) to maximize the control of nuisance net-phytoplankton.

Silver carp do not reproduce naturally in lacustrine ecosystems (Costa-Pierce, 1992). However, because of the high growth rate and survivorship of the small silver carps (40 g) in the present study, a very rapid increase in silver carp biomass is expected to occur in Lago Paranoá following its introduction.

Growth of silver carp population in a hypertrophic branch of Lago Paranoá, can be estimated using the fish growth rates obtained for different size classes from the net-cage experiments (Chapter IV.3), plus results from the present study and some unpublished data on Lago Paranoá (Table IV.9).

Table IV.9: Data on silver carp growth rates in Lago Paranoá used for estimates of population increase.

Size range of silver carp (g)	Growth rate (% of weight/d)	Source of reference
100-300	1.66	Present limnocorral experiment
300-1000	0.79	Net-cages experiment I (Chapter III)
1000-2000	0.65	Net-cages experiment II (Chapter III)
2000-12000	0.22	Unpublished data from Lago Paranoá

Future increase of silver carp biomass in Lago Paranoá can be estimated as follows:

Area of Bananal Branch = 973 ha

Target silver carp biomass = 50 g/m^3 or 625 kg/ha or 608,125 kg in the branch

Silver carp weight at stocking = 100 g

Initial stocking density of 70,000 fishes:

0 days (70,000; 100 g each) = 7,000 kg or 7,2 kg/ha; mortality= 10%

67 days (63,000; 300 g each) = 18,900 kg or 19,4 kg/ha; mortality= 10%

220 days (56,700; 1,000 g each) = 56,700 kg or 58,3 kg/ha; mortality= 5%

327 days (53,865; 2,000 g each)=107,730 kg or 110,7 kg/ha; mortality= 5%

1147 days (51,172; 12,000 g each)= 614,061 or 631 kg/ha

Therefore, starting at a very low silver carp biomass of only 1.15% of the target biomass (~ 600 kg/ha), silver carp population would reach 20% of this biomass by the end of the first year and the final target biomass within 38 months (i.e. 3.2 years). According to the present results, by the end of the first year after stocking, the silver carp population (100 kg/ha) would be significantly suppressing nuisance bloom-forming cyanobacteria. Only after 3 to 4 years of stocking, fishing campaigns aimed at removing part of the older fish, plus some periodic re-stocking of fingerlings, would be necessary.

IV.5. Conclusions

- Silver carp grazing activity promoted a shift in the size spectrum of particle distribution in the water towards small edible forms, the intensity of this shift was directly proportional to fish stocking density.
- Silver carp significantly suppressed colonial floating algae biomass (mainly *Microcystis aeruginosa*) irrespective of fish biomass level stocked into limnocorrals.
- The share of net-phytoplankton within the total algal biomass, and the total zooplankton densities were only significantly reduced at moderate silver carp biomass levels of 40-60 g/m^3 (c.a. 300-400 kg/ha).
- While present results suggest that there is no way to control nuisance algae without simultaneously suppressing zooplankton biomass, this is of little relevance to Lago Paranoá because the most abundant cyanobacteria are outside the grazing spectrum of dominant zooplankton (rotifers).

- A literature review showed that most succesful uses of silver carp as a biomanipulation agent (10 out of 14 studies, or 70%) were conducted in the tropics and subtropics, referred to situations of net-phytoplankton dominance or cyanobacteria blooms and involved the addition of low to moderate silver carp biomasses (< 1200 fish/ha, < 70 g/m^3 or < 900 kg/ha).
- In agreement with these studies, the present work suggests that the desirable silver carp biomass to be maintained in Lago Paranoá should not be lower than 20 g/m^3, but not exceed 70 g/m^3 to maximize the control of nuisance phytoplankton.
- Despite silver carp is unable to reproduce naturally in lacustrine ecosystems, high growth rate of silver carp population can be expected in Lago Paranoá based on data from net-cages and limnocorrals experiments; this indicates the need to implement a long-term fishing strategy to maintain silver carp abundance at desirable levels.

V - TILAPIA Vs. SILVER CARP MANIPULATION STRATEGIES

V.1. Introduction

It is well accepted that visual feeding planktivores indirectly enhance algal biomass and that such negative effect on water quality is fish-biomass dependent (McQueen *et al.*, 1986; Northcote, 1988; Lazzaro *et al.*, 1992). In contrast, the impacts of filter-feeding planktivores may vary according to a number of factors including fish species (Smith, 1988), type of ecosystem (Spataru & Gophen, 1985), fish biomass (Milstein, 1992; Arcifa *et al.*, 1995), and lake trophic state (Drenner *et al.*, 1996). Although the majority of filter-feeding planktivores are classified as omnivores, the size and amount of plankton material collected from the water and thus the final impacts of fish on water quality differ from species to species as a function of filtration ability and success. By comparing chinese carps, for instance, bighead carp is generally considered as zooplanktivorous and silver carp as phytoplanktivorous because of its highly developed and fine-meshed filtering which enable a more efficient algae consumption. Despite their ability to filter-feed on phytoplankton (Northcott *et al.*, 1991) and their high potential to digest plant material (Bowen, 1981), tilapia may shift between feeding modes and strategies during growth and/or depending on the relative amount of food resources in the environment. Thus, the same fish species may affect water quality in different manners (Drenner *et al.*, 1987; Vinyard *et al.*, 1988; Lazzaro, 1991; Beveridge & Baird, *in prep.*).

Therefore, from the literature concerning the impacts of tilapias on water quality, the same confounding picture emerges as from the studies on silver carp. From a total of 5 experimental studies using tilapias reviewed by Smith (1988), one work reported

algal increase, one found no effects and three referred to reductions in phytoplankton biomass by the fish. In a recent review of 48 experimental works on the impact of omnivorous filter-feeding fish, Drenner *et al.* (1996) included 10 trials involving tilapias. Likewise, only 2 resulted in clearly detrimental effects of fish (increase in phytoplankton biomass), a further 2 did not detect any fish effect and 6 recorded water quality improvements with tilapia presence.

As already discussed for silver carp in the previous chapter, fish biomass represents a key factor to explain such successes and failures as fish grazing effect may be offset by the direct supply of nutrient to algal growth via fish excretion. This detrimental fish-biomass mediated effect is especially important for tilapias as they represent a very prolific fish species that normally attains extremely high biomasses in lakes and reservoirs (Kolding, 1993).

Since the first experimental study of the impacts of planktivores on the plankton community and water quality of Lago Paranoá (Starling & Rocha, 1990), there were indications of both, the detrimental influence of high reservoir fish biomass (mainly tilapia) and the possible use of silver carp to directly control algae. From laboratory feeding experiments, it was shown that *Tilapia rendalli* was unable to substantially reduce the abundance of the dominant phytoplankton species, *Cylindrospermopsis raciborskii*, while silver carp efficiently suppressed this filamentous cyanobacteria in aquaria filled up with plankton from the reservoir. Consequently, during field experiment in small bag-type enclosures (2.5 m^3) stocked with a very high fish biomass (3,000 kg/ha), an enhancement of phytoplankton biomass in presence of tilapia but not of silver carp was observed in enclosures.

Based on such contrasting impacts from these two filter-feeding planktivores, Starling (1993b) suggested the progressive replacement of tilapia by silver carp as a management strategy for Lago Paranoá. Further experimental work has confirmed

both, the suitability of stocking silver carp at low to moderate densities (Starling, 1993a; Chapter IV) and the need to control tilapia over-population in the reservoir (Starling & Lazzaro, 1997; Chapter II). However, as these two promising biomanipulation strategies have not yet been simultaneously tested and compared under the same experimental conditions, it is still not known whether one can influence the other positively (synergism) or negatively (antagonism). Furthermore, there is no information on possible seasonal variations in their effectiveness, as well as on the potential direct and indirect influence of silver carp upon tilapia population dynamics.

Our working hypotheses are:

i) When stocked at low to moderate biomass, silver carp improves water quality by directly grazing on dominant net-phytoplankton (mainly colonial cyanobacteria).

ii) The control of tilapia biomass in the reservoir results in water quality improvements by reducing the internal nutrient recycling rates and availability to phytoplankton growth.

iii) Each biomanipulation strategy (tilapia removal and silver carp stocking) interferes with the other in a positive manner (synergism) and do not act independently from one another.

iv) Seasonal variations in the effects of both strategies may occur, following fluctuations in the abundance of nuisance bloom-forming net-phytoplankton.

In order to test the above hypotheses, two 3-month limnocorrals experiments were performed: experiment I during the rainy season and experiment II during the dry season.

V.2. Materials and Methods

A set of ten large littoral limnocorrals (5 m x 16 m, 1.8 m mean depth, 80 m^2 surface area, and 100 m^3 volume) were constructed within a small bay located in the vicinity of the Northern Sewage Treatment Plan, one of the most eutrophic areas of Lago Paranoá. On October 1-9 1995 (wet season), limnocorrals were formed by erecting four walls made of PVC- reinforced sheets, which completely isolated the water mass from the adjacent reservoir (see details of limnocorrals construction in Chapter II). All limnocorrals were set in a linear array extending between 2 m and 0.5 m in depth along the shore. Limnocorrals were completely opened to sediment but isolated from the reservoir shoreline.

During the process of construction, one side of each limnocorral was left open but covered only with a fish net (12 mm) to allow water but prevent fish exchange between limnocorral and adjacent reservoir. On October 10-12 and November 1-7, all fishes confined inside each of the recently erected limnocorrals were captured using seines. The same fishing effort, i.e., seining 5 times in each limnocorral with a 5 mm seine, was applied until limnocorrals were considered fishless. Total fish biomass captured in this pre-experimental condition is shown in Table V.1.

As fishes confined inside limnocorrals represented only a percentage of the overall original biomass that had not escaped during the process of erecting the barriers, it was found necessary to estimate the actual fish biomass in an equivalent surrounding area. On October 24, a marginal area (10 m x 8 m, 80 m^2 surface area, and 68 m^3 volume) was quickly isolated by netting (8 mm mesh) and rotenone (1 mg/l, mixed with the tracer substance "rhodamine stain") was applied to estimate the fish biomass characteristic from this littoral area of the reservoir. Potassium

permanganate (2 mg/l) was applied outside the area to neutralize rotenone and prevent reservoir fish from being harmed (Marking, 1992). All fishes captured inside the area were identified, measured and weighed. Results from this estimate are presented in Table V.2. On November 08, after verifying similar limnological conditions inside each limnocorral, they were all totally closed to adjacent reservoir by lifting the open side.

Based on nutrient analysis of sediment performed on October 16, limnocorrals were divided into two groups (blocks): limnocorrals 1-5 having nutrient-rich sediment (P concentration > 2,000 mg/kg of wet sediment) and limnocorrals 6-10 having nutrient- poor sediment (P concentration < 1,500 mg/kg of wet sediment). Results from the analysis are shown in Table V.3.

Experiment 1- Four treatments involving 2 levels of tilapia and silver carp biomass were randomly assigned to limnocorrals within each block following a 2 x 2 factorial design (Table V.4). An additional treatment involving caged silver carp was applied to limnocorrals 5 and 8, but will not be considered in this dissertation.

Determination of a typical and a reduced tilapia biomass from the reservoir to be re-introduced into limnocorrals took into account the total amount of tilapias captured by rotenone in the adjacent reservoir and the average reservoir fish biomass present inside limnocorrals before the experiment, respectively. Thus, a typical (120 g/m^3) or reduced tilapia biomass (40 g/m^3) was re-introduced into limnocorrals at a ratio of 4:1 between *Tilapia rendalli* and *Oreochromis niloticus* using adult fishes (~100 g) collected by cast-nets in the neighborhood. Adult tilapia represented over 70% of the total fish biomass both as estimated in the littoral adjacent area and in recently constructed limnocorrals; they were expected to spawn later on in limnocorrals where a certain amount of fish larvae was assumed to have escaped from initial seining. Thus, the fish community from the reservoir was assumed to be well represented by

adult tilapia plus some small tilapias which might have escaped from initial catches and larvae which were likely to be produced during the course of the experiment.

Table V.1: Fish biomass removed from limnocorrals before starting the experiment I.

	Limnocorral / Volume (m^3)										
	1	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**Mean**
	94.4	**101.6**	**107.2**	**112.0**	**118.4**	**108.0**	**105.6**	**98.4**	**103.2**	**94.4**	**104.3**
Fish larvae (g)	840	185	1005	1175	315	220	1405	550	750	1085	753
***Tilapia* weight (g)**	2420	7570	590	3975	6090	2185	980	2195	1065	495	2756
***Tilapia* number**	48	141	36	112	182	81	38	98	92	29	85.7
***Tilapia* mean weight (g)**	50.4	53.7	16.4	35.5	33.5	27	25.8	22.4	11.6	17.1	29.3
Other fish species (g)	1785	1654	475	810	2165	1350	330	785	255	275	987.5
Overall weight (g)	5045	9400	2070	5960	8570	3755	2715	3530	2070	1855	4497
***% tilapia* weight ***	58	82	55	83	74	62	75	74	81	64	70.8
Total Biomass (kg/ha)	631	1175	259	745	1071	469	339	441	259	232	562.1
Total Biomass (g/m^3)	53.4	92.5	19.3	53.2	72.4	34.8	25.7	35.9	20.1	19.7	42.7

Legend: * share of tilapia in total fish abundance, excluding larvae not identified at species level.

Table V.2: Results from total fish biomass estimates in an adjacent reservoir area by rotenone addition.

Species	Number	Weight (g)	Mean weight (g)	% number	% weight	Biomass (g/m^3)	Biomass (kg/ha)
T.rendalli	223	10115	45.4	79.4	84.7	148.8	1264
O.niloticus	23	380	16.5	8.2	3.4	5.6	47.5
L.macrochira	7	160	22.9	2.5	1.3	2.4	20
A.portalegrensis	15	415	27.7	5.3	3.5	6.0	51.9
C.carpio	1	95	95	0.4	0.8	1.4	11.9
H.malabaricus	6	470	78.3	2.1	3.9	6.9	58.8
G.carapo	1	185	185	0.4	1.5	2.7	23.1
Rhamdia sp.	1	75	75	0.4	0.6	1.1	9.4
A.bimaculatus	4	40	10	1.3	0.3	0.6	5
TOTAL	281	11935	555.8	100	100	175.5	1491.5

A low to moderate biomass of adult silver carp was used according to treatments as free-roaming fishes only. Numbers and weights of tilapia and. silver carp stocked in each limnocorral are shown in Table V.3. The procedure of stocking fish in all limnocorrals took 3 days (November 9-11). Experiment started on November 9 by monitoring limnological characteristics in each limnocorral as well as in two adjacent reservoir sampling areas at 7 days intervals

Using a small boat, a 10-litre surface composite sample was collected from each limnocorral, by pooling 3 samples for temperature, pH, turbidity, total alkalinity, dissolved oxygen, conductivity, total phosphorus (TP), orthophosphate (PO_4P), ammonia, nitrate, total Kjeldahl nitrogen, total nitrogen (TN), total chlorophyll-a, chlorophyll-a fraction < 15 µm, phytoplankton primary productivity (dark and light bottles incubated for 2 hours around noon), phytoplankton and zooplankton counts (APHA, 1985). A sub-sample (500 ml) of the surface water from each limnocorral was concentrated down to 30 ml by sedimentation for 24 hours in standard glass cones (2000 ml) and preserved in 0.5% Lugol's iodine to obtain a size range spectrum of particles distribution (APHA, *op. cit.*). Counting and sizing was done by Coulter Multisizer® fitted with tube of orifice size 70 µm was conducted as described in Rahmatullah (1992). Water transparency (Secchi disk) and fish mortality were also monitored weekly. Verification of the complete isolation of each area from the adjacent reservoir was done by scuba divers twice a month.

The experiment lasted 84 days, after which all fishes were recovered from each limnocorral using a combination of rotenone (1mg/l) and seine nets. Individual weights of silver carp recaptured from limnocorrals were recorded. Nutrient content of sediment from each limnocorral and adjacent lake were also analyzed at the end of the period.

Table V.3: Nutrients in the sediment and fish biomass stocked to limnocorrals in experiment I (see Table V.4 for codes).

	Treatments / Limnocorrals											
	TR 1	TT+FC 2	TR+FC 3	TT 4	5	TT+FC 6	TR 7	8	TR+FC 9	TT 10	LAKE 11	LAKE 12
TN in sediment (mg/kg)	695	490	502	704	389	465	70	139	165	122	449	-
TP in sediment (mg/kg)	2186	2805	2581	2264	2401	1486	585	279	224	200	1006	-
Number of *tilapia* re-stocked	39	117	44	124		105	41		41	120	-	-
Weight of *tilapia* re-stocked (g)	4340	12465	4010	12035		12650	4050		4050	13014	-	-
Tilapia biomass (kg/ha)	543	1558	501	1504		1581	506		506	1627		
Tilapia biomass (g/m^3)	46	123	37	107		117	38		39	138		
Number of silver carp stocked	0	7	7	0		7	0		7	0		
Weight of silver carp stocked (g)	0	4205	4265	0		4585	0		4180	0		
Silver carp Biomass (kg/ha)	0	536	533	0		573	0		523	0		
Silver carp Biomass (g/m^3)	0	41	40	0		42	0		40	0		
Total fish Biomass (kg/ha)	543	2084	1034	1504		2154	506		1029	1627		
Total fish Biomass (g/m^3)	46	164	77	107		159	38		79	138		

Table V.4: Experimental design of the first limnocorral experiment (wet season).

		Tilapia Treatment	
		Typical Biomass (120 g/m^3)	Reduced Biomass (40 g/m^3)
Silver Carp Treatment	Absent (0 g/m^3) and Free (40 g/m^3)	TT (limnocorrals 2 and 6) 120 g/m^3	TR (limnocorrals 1 and 7) 40 g/m^3
		TT+FC (limnocorrals 5 and 8) 160 g/m^3	TR+FC (limnocorrals 3 and 9) 80 g/m^3

Legend: TT = typical tilapia (or control);TR = tilapia reduction; FC = free carp

Repeated-measures ANOVA with randomized block design was applied to detect differences between treatments for each limnological variable at α=0.10, using log-transformed data. Initial differences between limnocorrals for all limnological parameters, as well as for nutrient content of sediment, were tested using One-way ANOVAs. All statistical analyses were performed using SYSTAT (Wilkinson, 1989).

Experiment II- The same set of ten large littoral limnocorrals from the previous experiment was reconstructed and used during the dry season trial. On May 15 1996, after having been left in the reservoir bottom for three months, the PVC-reinforced sheets were again lifted to form the limnocorrals walls. As in the previous study, one side of each limnocorral was left open and covered only with a fish net (12 mm) to allow water but prevent fish exchange between limnocorral and the adjacent reservoir. On May 28-29 and 11-12 June 1996, all fishes confined inside each recently erected limnocorral were captured by seine netting. Fishing effort was increased relative to the previous study by seining 7 times in each limnocorral (5 mm mesh seine) until they were considered fishless. Total fishes captured in this pre-experimental condition are shown in Table V.5.

Based on the results obtained in experiment I and taking into account the availability of two additional limnocorrals for running experiment II, it was decided to triplicate the number of limnocorrals for the treatments of typical and reduced biomass of tilapia in the absence of silver carp. The total fish biomass stocked into the limnocorrals was also slightly modified as a result of treatment intensification. However, a factorial design similar to that of previous experiment was followed (Table V.6).

Table V.5: Fish biomass removed from limnocorrals before starting the experiment II.

	Limnocorral / Volume (m^3)										
	1	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**Mean**
	98.4	**105.6**	**111.2**	**116.0**	**122.4**	**112.0**	**109.6**	**102.4**	**107.2**	**98.4**	**108.3**
Fish larvae (g)	3180	2140	825	270	345	495	305	255	425	2190	1043
***Tilapia* weight (g)**	545	1560	1540	460	545	1360	1550	200	1625	490	988
***Tilapia* number**	17	48	73	25	34	66	27	10	55	23	38
***Tilapia* mean weight (g)**	32.0	32.5	21.1	18.4	16.0	20.6	57.4	20.0	29.5	21.3	26.9
Other fish species (g)	295	600	815	10	600	595	1645	190	675	890	631.5
Overall weight (g)	4020	4300	3180	740	1490	2450	3500	645	2725	3570	2662
***% tilapia* weight ***	64.9	72.2	65.4	97.9	47.6	69.6	48.5	51.3	70.7	35.5	62.4
Total Biomass (kg/ha)	502.5	537.5	397.5	92.5	186.3	306.3	437.5	80.6	340.6	446.3	332.8
Total Biomass (g/m^3)	40.9	40.7	28.6	6.4	12.2	21.9	31.9	6.3	25.4	36.3	25.1

Legend: * share of tilapia in total fish abundance, excluding larvae not identified at species level.

By performing additional analysis on nutrient content of sediment inside each limnocorral on June 17, the persistence of the enrichment gradient in the two blocks from experiment I was verified for experiment II: block I with limnocorrals 1-5 having nutrient-rich sediment (P > 1,000 mg/Kg of wet sediment), and block II with

limnocorrals 6-10 having nutrient-poor sediment (P < 1,000 mg/Kg of wet sediment). Results from this analysis of sediment as well as number and weight of tilapia and silver carp stocked to limnocorrals are shown in Table V.7. The stocking of fish into all limnocorrals took 5 days (June 13-17).

Table V.6: Experimental design of the second limnocorral experiment (dry season).

		Tilapia Treatment	
		Typical Biomass ($100\ g/m^3$)	Reduced Biomass ($25\ g/m^3$)
Silver Carp Treatment	Absent ($0\ g/m^3$) and Free ($50\ g/m^3$)	TT (limnocorrals 3, 5 and 7) $100\ g/m^3$	TR (limnocorrals 4, 6 and 10) $25\ g/m^3$
		TT+FC (limnocorrals 2 and 9) $150\ g/m^3$	TR+FC (limnocorrals 1 and 8) $75\ g/m^3$

Legend: TT = typical tilapia (or control); TR = tilapia reduction; FC = free carp

The experiment started on June 18 by monitoring limnological characteristics from each limnocorral and from two adjacent reservoir sampling areas at 7-day intervals over a 56-day period. At the end of this period, all fishes were recovered from each limnocorral using a combination of rotenone (1mg/l) and seine nets. Total weight of silver carp recovered from limnocorrals was recorded. Repeated-measures ANOVA with randomized block design was applied to detect differences between treatments for each limnological variable at $\propto$=0.10, using log-transformed data. Initial differences between limnocorrals for all limnological parameters, as well as for nutrient content of sediment, were tested using one-way ANOVAs. All statistical analyses were performed using SYSTAT (Wilkinson, 1989).

Table V.7 : Nutrient content of sediment and fish biomass stocked to limnocorrals in experiment II.

	Treatments / Limnocorrals											
	TR+FC 1	TT+FC 2	TT 3	TR 4	TT 5	TR 6	TT 7	TR+FC 8	TT+FC 9	TR 10	LAKE 11	LAKE 12
TN in sediment (mg/kg)	585	597	840	801	915	489	317	259	248	226	114	668
TP in sediment (mg/kg)	1309	1712	1837	1725	1681	474	354	157	227	192	215	1439
Number of *tilapia* re-stocked	30	117	115	30	118	30	111	30	108	30	-	-
Weight of *tilapia* re-stocked (g)	2645	10890	10760	2725	11290	2655	10490	2685	10520	2655	-	-
Tilapia biomass (kg/ha)	331	1361	1345	341	1411	332	1311	336	1315	332	-	-
Tilapia biomass (g/m^3)	27	103	97	23	92	24	96	26	98	27	-	-
Number of silver carp stocked	8	8	0	0	0	0	0	8	8	0	-	-
Weight of silver carp stocked (g)	5110	5350	0	0	0	0	0	5235	5290	0	-	-
Silver carp biomass (kg/ha)	639	669	0	0	0	0	0	654	661	0	-	-
Silver carp biomass (g/m^3)	52	51	0	0	0	0	0	51	49	0	-	-
Total fish biomass (kg/ha)	970	2030	1345	341	1411	332	1311	990	1976	332	-	-
Total fish biomass (g/m^3)	79	154	97	23	92	24	96	77	147	27	-	-

V.3. Results

Experiment I - During the wet season, all limnocorrals (excluding the 2 adjacent lake sampling areas) had similar initial values for temperature (P=0.737), pH (P=0.567), turbidity (P=0.657), dissolved oxygen (P=0.611), conductivity (P=0.868), total phosphorus (P=0.535), orthophosphate (P=0.121), total chlorophyll-a (P=0.739), chlorophyll-a fraction < 15 μm (P=0.200), percentage of nanno- and picoplankton (P=0.348), phytoplankton primary productivity (P=0.370), *Microcystis* abundance (P=0.677), total zooplankton counts (P=0.658), nitrogen content of sediment (P=0.919) and phosphorus content of sediment (P=0.835). The only significant initial differences between limnocorrals were detected for Secchi depth (P=0.039), nitrate (P=0.086), total Kjeldahl nitrogen (P=0.015), and total nitrogen (P=0.020). Secchi depth was higher at typical tilapia biomass (TT) if compared with typical tilapia biomass plus silver carp (TT+FC). Nitrate was lower at typical tilapia biomass (TT) relatively to reduced tilapia biomass plus silver carp combination treatment (TR+FC). TKN was lower in combination treatment (TR+FC) than in the typical tilapia biomass plus silver carp (TT+FC). TKN and total N were lower at reduced tilapia biomass (TR) when compared to combination treatment (TR+FC) and typical tylapia biomass (TT).

As shown in Table V.8, the only significant treatment effect occurred for *Microcystis* abundance (P=0.054), which declined in the presence of silver carp but was not affected by tilapia reduction. As can be seen from the evolution of *Microcystis* abundance for each treatment during the experiment (Figure 5.1), there was a bloom in limnocorrals containing a typical tilapia over-population from the reservoir. This is well illustrated by the general pattern in the size distribution of 2-25

μm diameter particles generated by the Coulter Multisizer® (Figure 5.2). Although this figure represents the overlay of the different average size distribution curves obtained for each treatment (differences not statistically tested), it is clear that large-sized particles accumulated in the control treatment is associated with the bloom of *Microcystis*. Over 60% reduction in the abundance of such nuisance algae was achieved by stocking silver carp (Figure 5.1). The concomitant control of tilapia over-population did not bring any additional benefit in terms of *Microcystis* suppression as no interaction effect between silver carp addition and tilapia reduction was detected.

Table V.8: Probability values from repeated-measures ANOVA performed on data for all treatments (except caged-silver carp) during experiment I (84 days, 12 sampling dates).

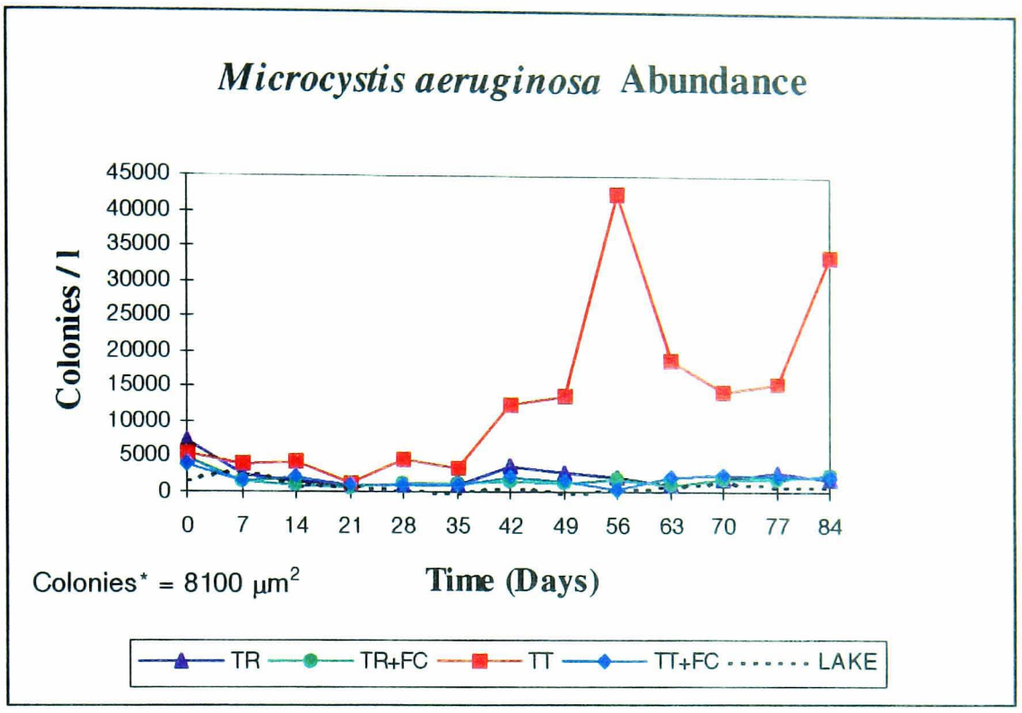
Limnological Parameters	Treatments main effects		
	P(Carp)	P(Tilapia)	P(Interaction)
Total Zooplankton	0.433	0.761	0.579
Primary Productivity	0.561	0.508	0.275
Nanoplankton Chlorophyll-a	0.475	0.109	0.495
% Nanoplankton Fraction	0.983	0.686	0.829
Total Chlorophyll-a	0.804	0.392	0.948
Microcystis Abundance	0.054*	0.102	0.148
Total Nitrogen	0.768	0.494	0.964
Total Kjeldahl Nitrogen	0.772	0.418	0.987
Nitrate	0.857	0.863	0.678
Orthophosphate	0.845	0.735	0.326
Total Phosphorus	0.834	0.998	0.473
Conductivity	0.893	0.516	0.183
Dissolved Oxygen	0.654	0.772	0.458
Turbidity	0.620	0.342	0.918
pH	0.636	0.434	0.233
Secchi Depth	0.557	0.133	0.660

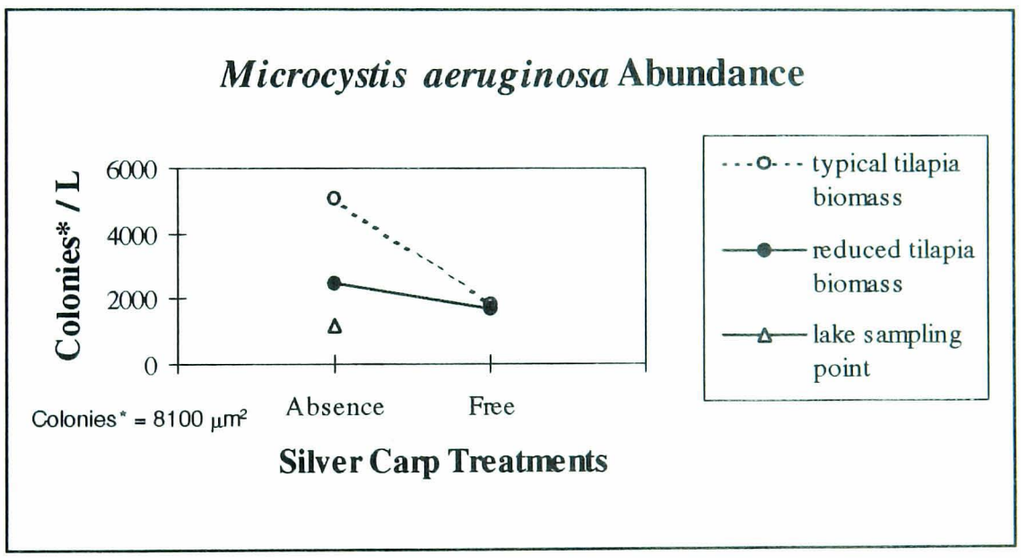
Legend: P(Til.)=probability value for tilapia main effect; P(Carp)=probability value for silver carp main effect; P(Inter.)=probability value for tilapia x silver carp interaction effect; *significant, $\propto = 0.10$.

Total fish biomass recovered from limnocorrals at the end of the experiment is shown in Table V.9. The original fish biomass stocked in the limnocorrals suffered some changes as a result of fish spawning, predation by aquatic birds, plus some problems with the pre-experimental fish removal. From the final total reservoir fish biomass captured in all limnocorrals, 25% were fish larvae* (< 5 cm TL) which reflects fish spawning, and 6% were fish from species other than tilapia, indicating that the initial removal of fishes was very efficient. In general, total fish biomass increased in the treatments involving tilapia reduction and decreased in control treatment with higher tilapia biomass.

Table V.9 : Final reservoir fish capture (in terms of weight, number and biomass) in each limnocorral a the end of experiment I.

	Treatments / Limnocorral								
	TR 1	**TR 7**	**TR+FC 3**	**TR+FC 9**	**TT 4**	**TT 10**	**TT+FC 2**	**TT+FC 6**	**Mean**
Weight /number tilapia (g)	4805 (64)	5100 (54)	2865 (28)	3295 (50)	7635 (73)	4170 (46)	8350 (138)	4065 (55)	5036 (64)
Weight/number small tilapia (g)	330 (35)	395 (46)	1045 (81)	45 (5)	565 (51)	185 (18)	235 (19)	150 (12)	369 (33)
Weight /number other fish (g)	790 (9)	95 (7)	240 (1)	0 (0)	1650 (3)	450 (3)	615 (6)	135 (4)	497 (4)
Weight fish larvae (g)	1385	3365	1765	1725	1990	3270	1355	1010	1983
Total weight fish from lake(g)	7310	8955	5915	5065	11840	8075	10555	5360	7884
Total lake fish biomass (g/m^3)	77.4	84.8	55.2	49.1	105.7	85.5	103.9	49.6	61.1
Total lake fish biomass (kg/ha)	914	1119	739	633	1480	1009	1319	670	985
% of initial lake fish biomass	168	217	149	126	99	62	84	42	118





Treatments:		% Reduction
TT	(typical tilapia biomass from lake or control)	0
TR	(reduction of tilapia overpopulation)	51.3
TT+FC	(silver carp addition on control situation)	64.5
RT+FC	(combination of tilapia reduction and silver carp addition)	66.5

Figure 5.1: Evolution of *microcystis* abundance and mean responses for each treatment during the course of the experiment I (wet season).

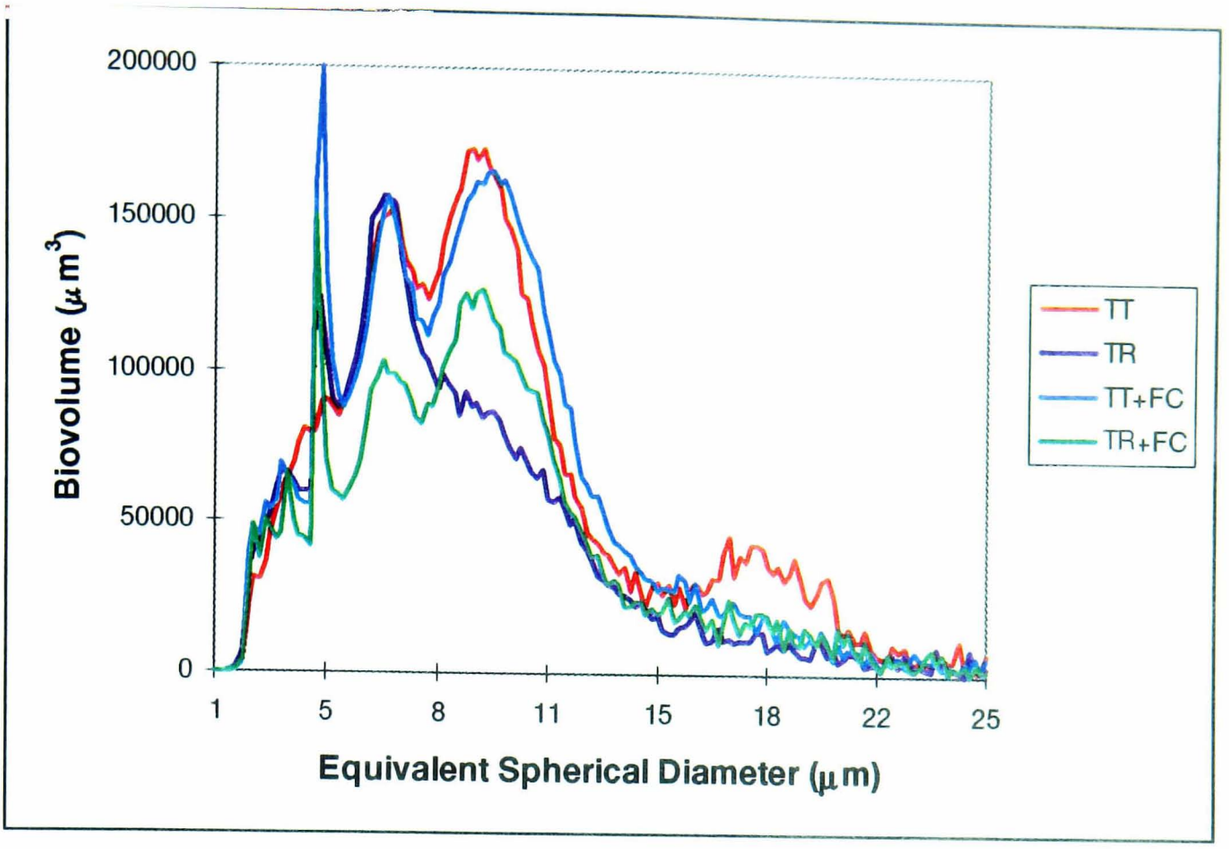


Figure 5.2: Total seston biovolume accounted for by average particle spherical diameter (1-25 µm) classes on 13 sampling dates for different treatments.

Consequently, the initial contrast between typical and reduced tilapia biomass was attenuated from 66% to 16% at the end (Table V.10). This indicates that the carrying capacity for tilapia biomass in limnocorrals is somewhere between the reduced and normal biomass used. Despite such changes, the basis of the original experimental design in terms of treatments definition was maintained, as can be seen in Table V.10.

Table V.10: Overall fish biomass changes during the course of limnocorral experiment I.

	Treatments			
	TR	**TR+FC**	**TT**	**TT+FC**
Total stocked fish biomass (g/m^3)	42	78	122.5	161.5
Total recaptured fish biomass (g/m^3)	77.5	88	92.5	119
Total average fish biomass (g/m^3)	59.8	83	107.5	140.3

All silver carps stocked into limnocorrals were recovered at the end of the experiment. Initial and final weights of individual fish, as well as fish growth rate for the 84-day duration of experiment I are shown in Table V.11. Except for limnocorral 9 (TR+FC), silver carp gained weight in all limnocorrals, although growth rates were much lower than those obtained for smaller fishes in limnocorrals (Chapter IV) and in cages incubated in an adjacent open area of Lago Paranoá (Chapter III).

Table V.11 : Silver carp growth rates during the course of limnocorral experiment I.

	Treatments/Limnocorral				
	TR+FC 3	**TR+FC 9**	**TT+FC 2**	**TT+FC 6**	**Mean**
Total gain of weight (g)	265	-5	225	200	171
Total gain of weight (g/d)	3.15	0	2.68	2.38	2.05
Fish mean initial weight (g)	609.3	597.1	600.7	655.0	615.5
Growth rate (g/d/fish)	0.45	0	0.38	0.34	0.29
Growth rate (% w.w/fish/d)	0.072	0	0.062	0.051	0.046
Mortality rate (%)	0	0	0	0	0

Experiment II - At the start of experiment II, all limnocorrals (excluding the 2 adjacent lake sampling areas) had similar initial values for water temperature (P=0.684), Secchi depth (P=0.231), alkalinity (P=0.884), pH (P=0.276), turbidity (P=0.245), dissolved oxygen (P=0.430), conductivity (P=0.662), total phosphorus (P=0.648), orthophosphate (P=0.194), nitrate (P=0.251), ammonia (P=0.285), total Kjeldahl nitrogen (P=0.959), total nitrogen (P=0.990), total chlorophyll-a (P=0.380), chlorophyll-a fraction < 15 μm (P=0.459), percentage of nanno- and picoplankton (P=0.263), colonial algae abundance (P=0.413), *Botryococcus* abundance (P=0.611), total zooplankton counts (P=0.620), nitrogen content of sediment (P=0.483) and

phosphorus content of sediment (P=0.867). The only significant initial difference between treatments was detected for phytoplankton primary productivity (P=0.024), which was lower in limnocorrals stocked with typical tilapia biomass plus free silver carp (TT+FC) comparatively to all other treatments.

Table V.12 shows the results from repeated-measures ANOVA performed on data from all treatments for the whole period. There were significant tilapia main effects for Secchi depth (P=0.055), turbidity (P=0.003), total phosphorus (P=0.079), total chlorophyll-a (P=0.001) and colonial algae abundance (P=0.063). For these variables, the tilapia reduction improved water quality when compared with the typical tilapia biomass. Total chlorophyll-a values were also significantly reduced in the presence of silver carp (P=0.027) and represented the only main effect of this fish species. No synergism between tilapia and carp was observed except for a significant interaction effect for pH (P=0.077).

The evolution and overall mean values for water quality variables significantly affected by the biomass manipulation of the two omnivorous fishes are shown in Figures 5.3 to 5.7. In general, the reduction of tilapia was associated with an increase in Secchi depth of 33%, a reduction in turbidity of 46%, and decreases in total phosphorus, total chlorophyll-a and abundance of colonial algae of 31%, 38% and 70%, respectively, compared with the typical tilapia biomass situation. The only significant effect of silver carp concerned the most important parameter, total chlorophyll-a, which was reduced by 22% at typical tilapia biomass plus silver carp (TT+FC) treatment and by 60% at reduced tilapia biomass with silver carp addition treatment.

The general pattern of particle biovolume distribution for each treatment during the experiment II illustrates the progressive improvement in water quality brought about by the application of biomanipulation approaches (Figure 5.8). The

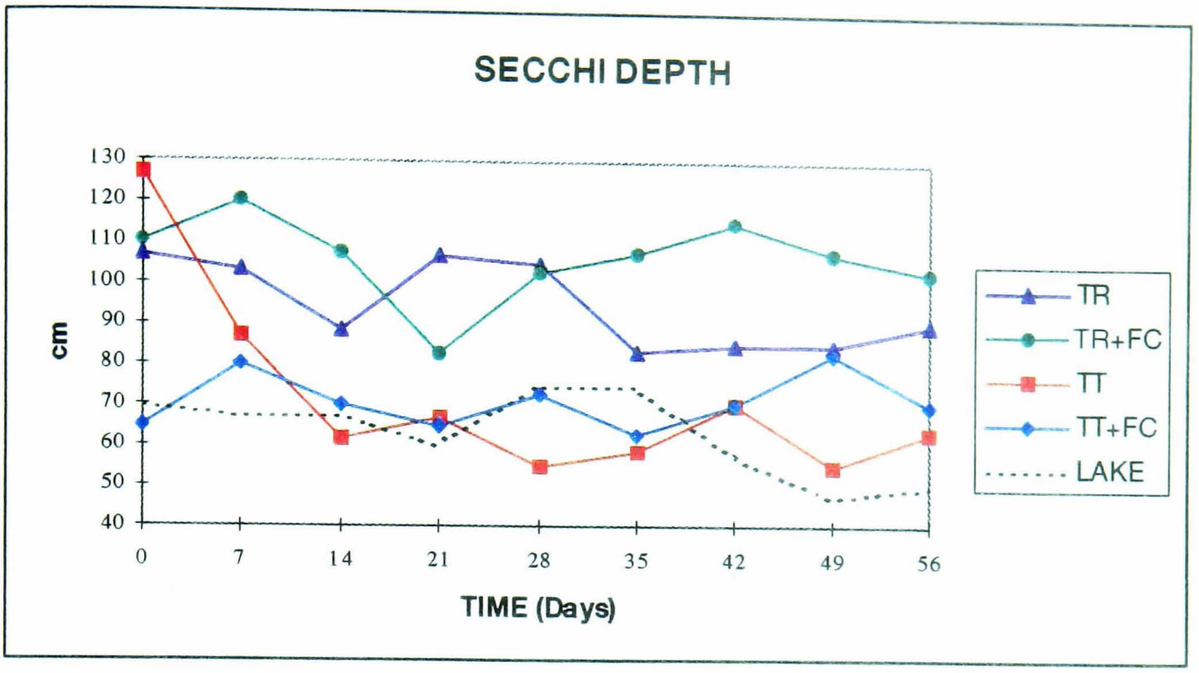
combination of stocking silver carp with reduction of tilapia biomass was found to be the most successful treatment as it resulted in the lowest amount of particles in the water. This also represents an additional indirect evidence of the higher transparency and lower phosphorus, turbidity, chlorophyll-a and colonial algae values associated with all treatments when compared with the typical tilapia biomass situation (Figures 5.3 to 5.7).

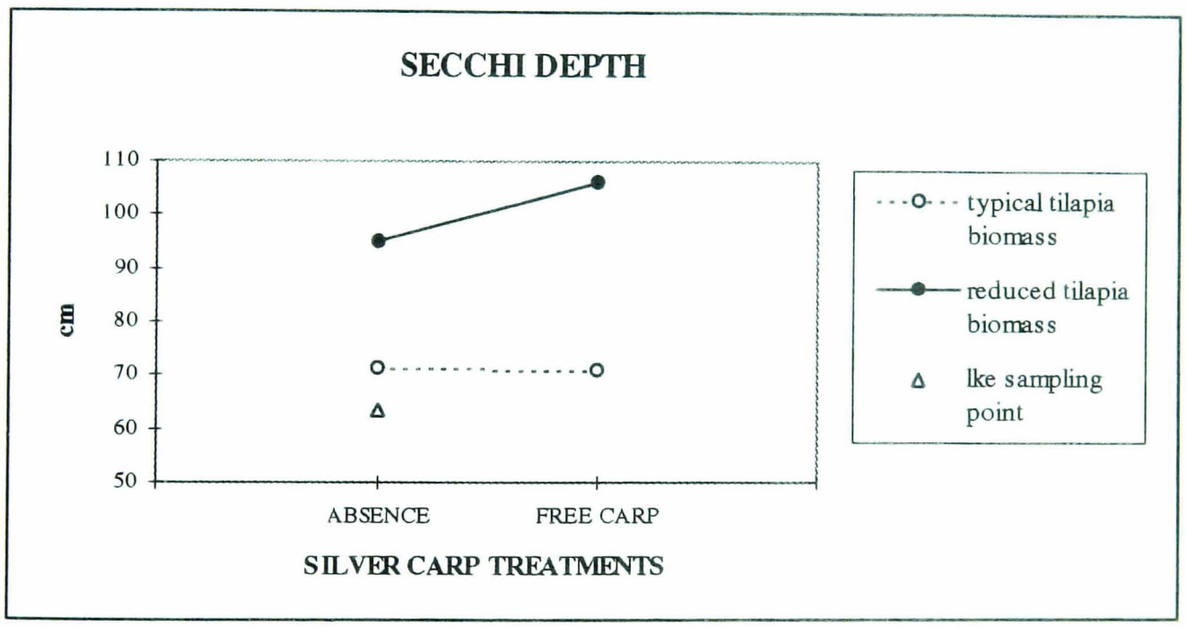
Table V.12: Probability values from repeated-measures ANOVA performed on original data for all treatments during experimental II (56 days, 8 sampling dates).

Limnological Parameters	Treatments main effects		
	P(Carp)	P(Tilapia)	P(Interaction)
Total Zooplankton	0.125	0.897	0.454
Primary Productivity	0.864	0.452	0.878
Nanoplankton Chlorophyll-a	0.336	0.353	0.818
% Nanoplankton Fraction	0.813	0.155	0.833
Total Chlorophyll-a	0.027*	0.001*	0.254
Microcystis Abundance	0.837	0.063*	0.913
Total Nitrogen	0.858	0.692	0.901
Total Kjeldahl Nitrogen	0.862	0.841	0.986
Nitrate	0.615	0.972	0.951
Orthophosphate	0.998	0.564	0.503
Total Phosphorus	0.285	0.079*	0.782
Conductivity	0.596	0.836	0.520
Dissolved Oxygen	0.171	0.193	0.391
Turbidity	0.102	0.003*	0.920
PH	0.397	0.316	0.077*
Secchi Depth	0.616	0.055*	0.693

Legend: P(Til.)=probability value for tilapia main effect; P(Carp)=probability value for silver carp main effect; P(Inter.)=probability value for tilapia x silver carp interaction effect; *significant, $\propto = 0.10$.

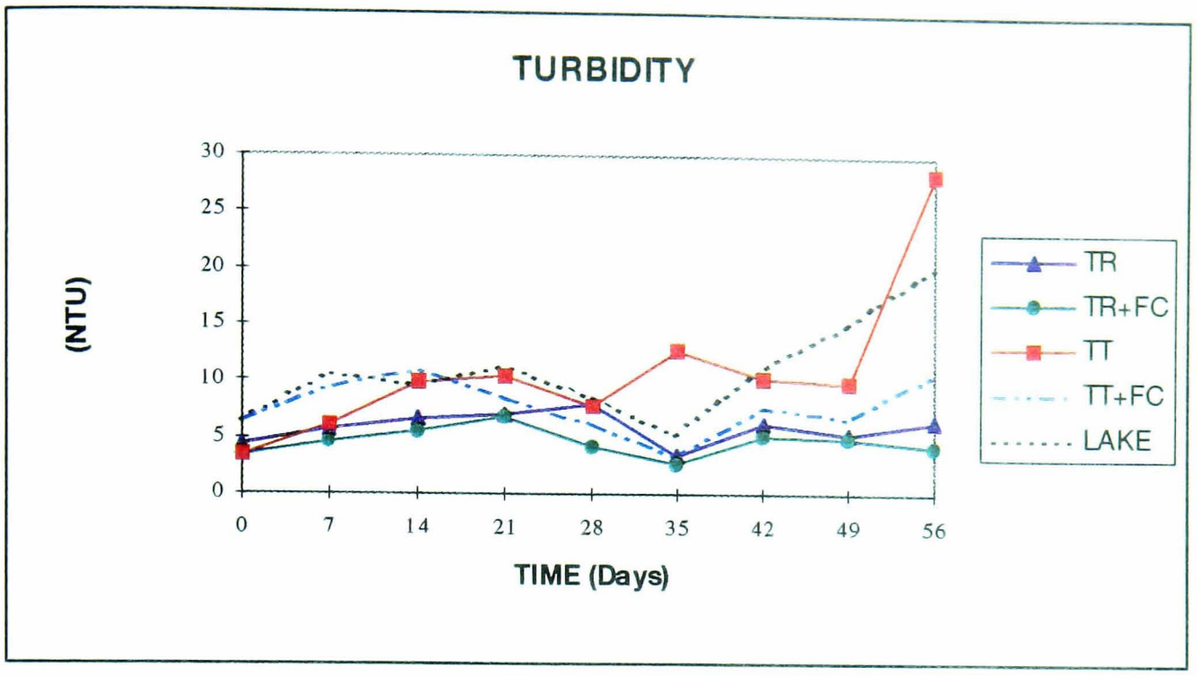
Total fish biomass recovered from limnocorrals at the end of the experiment is shown in Table V.13. As in the previous experiment, the original fish biomass stocked in limnocorrals suffered some changes as a result of fish spawning, predation by aquatic birds and incomplete pre-experimental fish removal. Although total fish biomass remained almost the same in the treatments involving tilapia reduction, it had decreased in treatment with typical tilapia biomass.

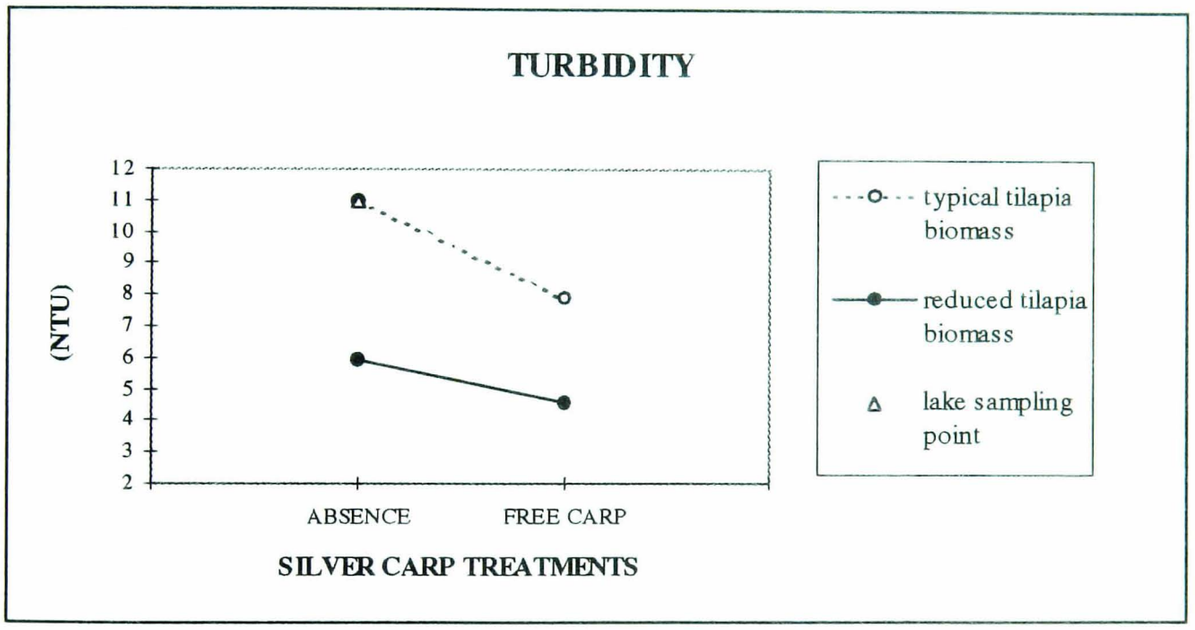




Treatments:		% Increase
TT	(typical tilapia biomass from lake or control)	0
TT+FC	(typical tilapia biomass plus silver carp addition)	0
TR	(reduction of tilapia overpopulation)	32.6
TR+FC	(combination of tilapia reduction with silver carp)	48.4

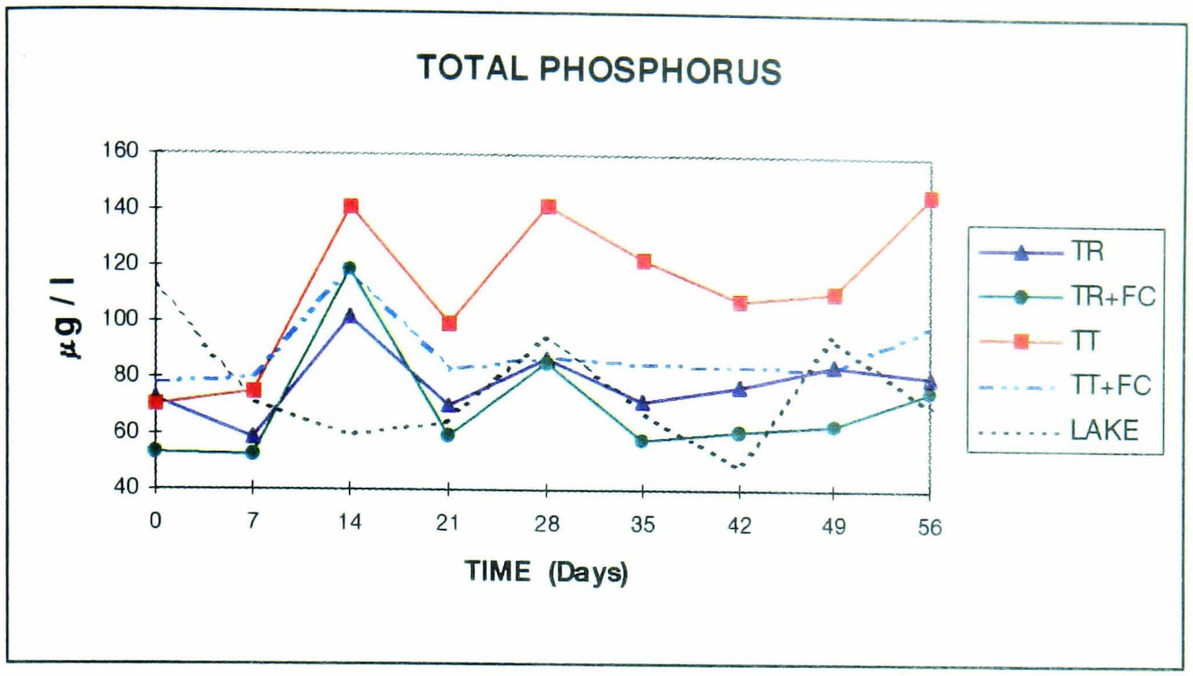
Figure 5.3: Evolution of Secchi depth and its overall mean values for each treatment during the course of the experiment II.

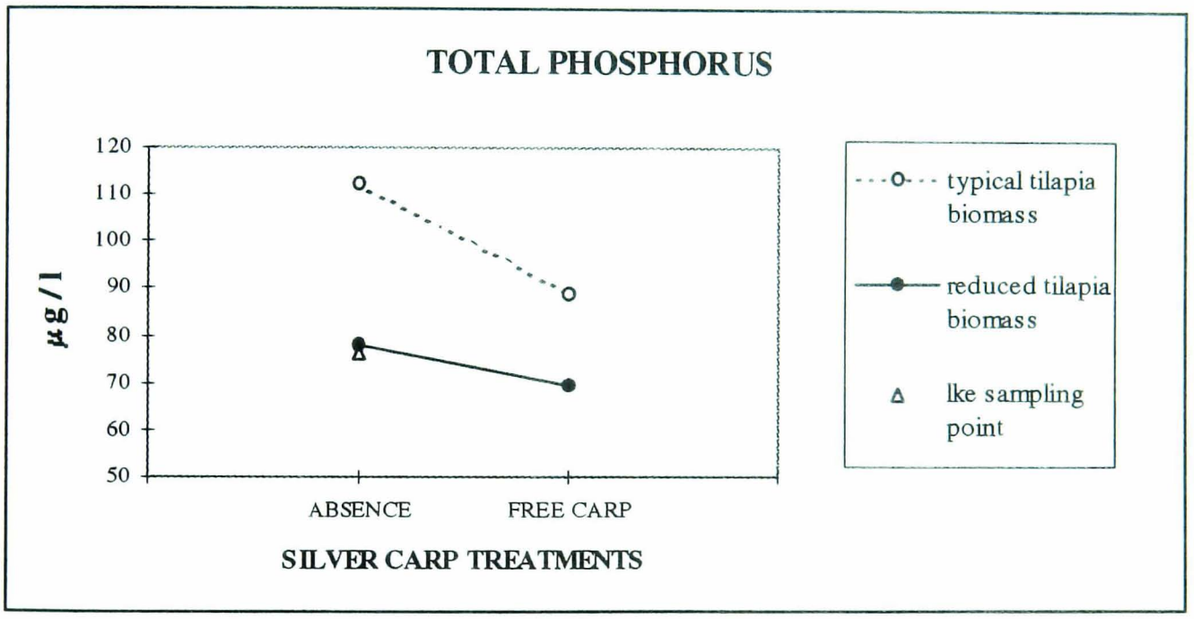




Treatments:		% Increase
TT	(typical tilapia biomass from lake or control)	0
TT+FC	(typical tilapia biomass plus silver carp addition)	28.3
TR	(reduction of tilapia overpopulation)	46.1
TR+FC	(combination of tilapia reduction with silver carp)	58.1

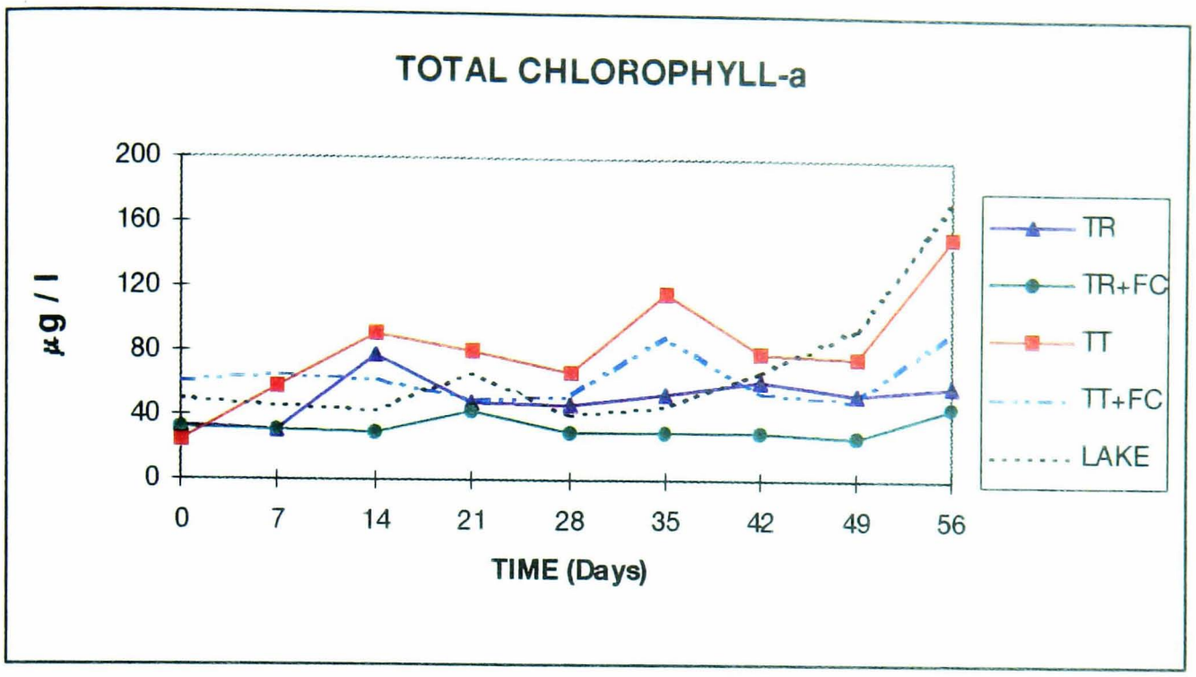
Figure 5.4: Evolution of turbidity and its overall mean values for each treatment during the course of the experiment II.

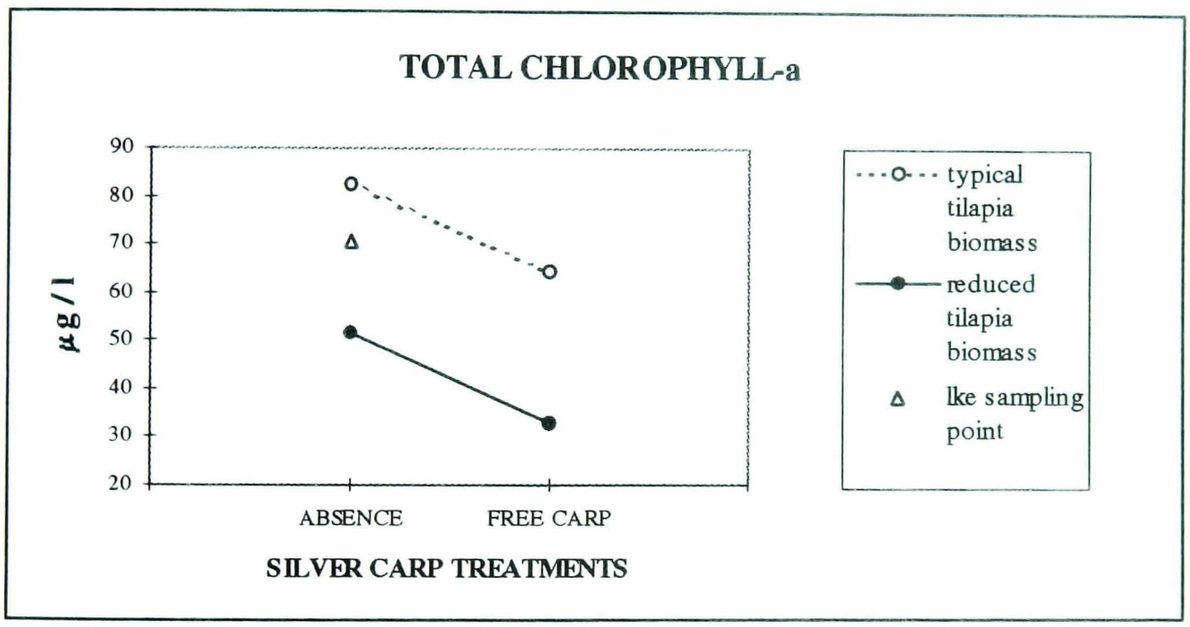




Treatments:		%Reduction
TT	(typical tilapia biomass from lake or control)	0
TT+FC	(typical tilapia biomass plus silver carp addition)	20.8
TR	(reduction of tilapia overpopulation)	30.5
TR+FC	(combination of tilapia reduction with silver carp)	38.2

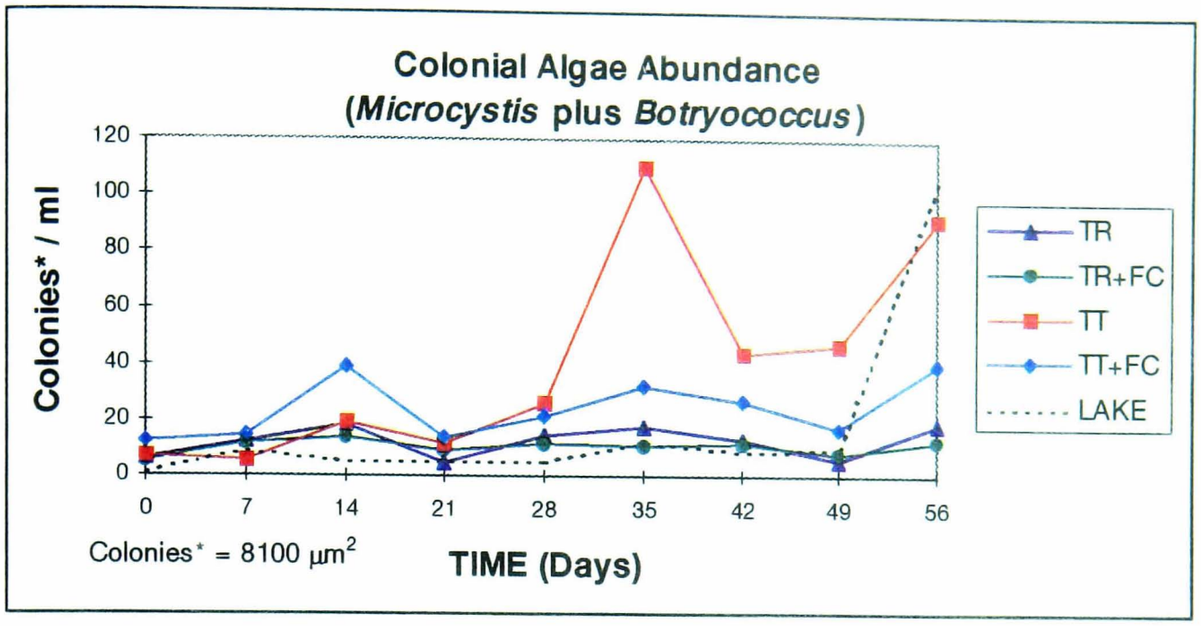
Figure 5.5: Evolution of total phosphorus and its overall mean values for each treatment during the course of the experiment II.

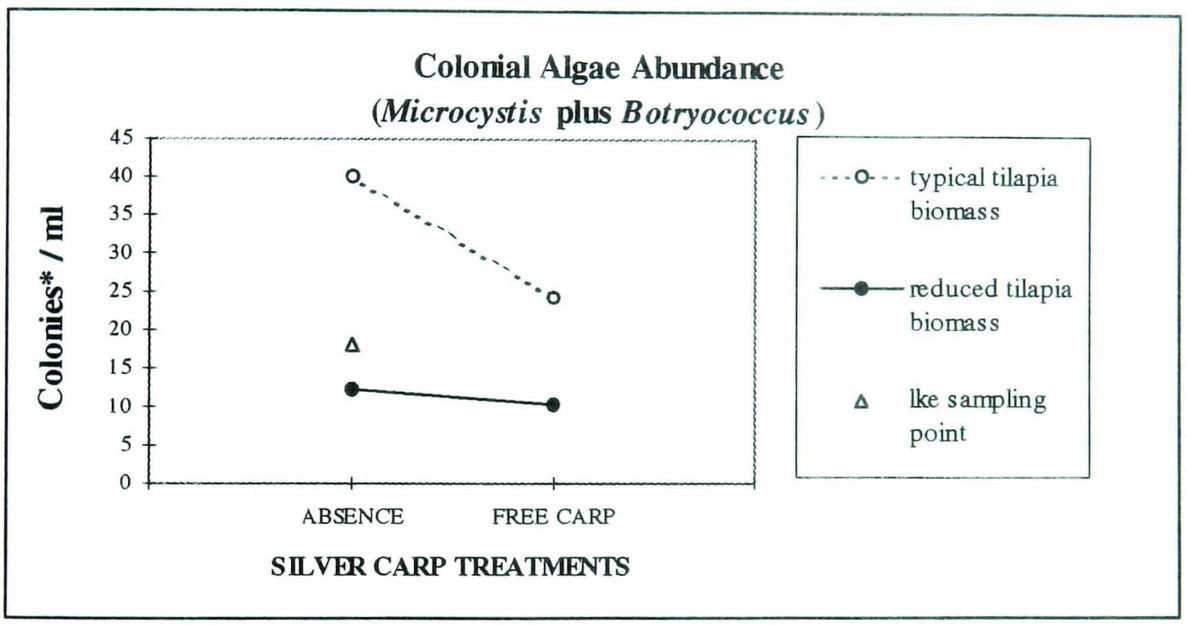




Treatments:		%Reduction
TT	(typical tilapia biomass from lake or control)	0
TT+FC	(typical tilapia biomass plus silver carp addition)	21.8
TR	(reduction of tilapia overpopulation)	37.8
TR+FC	(combination of tilapia reduction with silver carp)	60.4

Figure 5.6: Evolution of total chlorophyll-a and its overall mean values for each treatment during the course of the experiment II.





Treatments:		%Reduction
TT	(typical tilapia biomass from lake or control)	0
TT+FC	(typical tilapia biomass plus silver carp addition)	40.0
TR	(reduction of tilapia overpopulation)	44.0
TR+FC	(combination of tilapia reduction with silver carp)	74.1

Figure 5.7: Evolution of colonial algae abundance and its overall mean values for each treatment during the course of the experiment II.

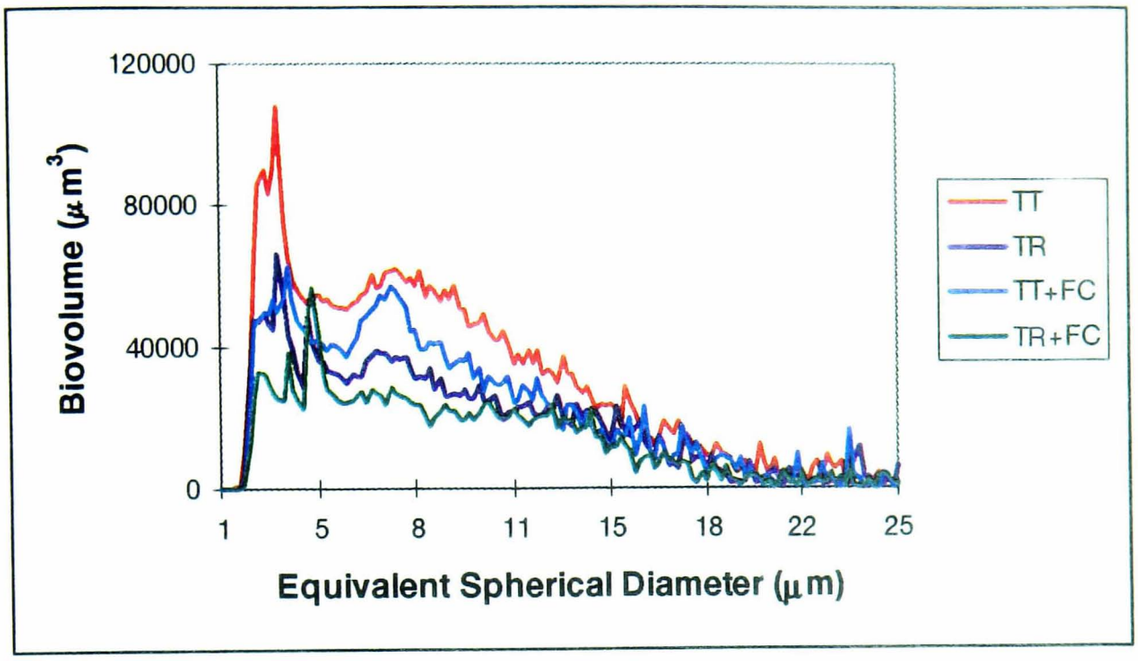


Figure 5.8: Total seston biovolume accounted for by average particle spherical diameter (1-25 µm) classes on 9 sampling dates for different treatments of experiment II.

Table V.13 : Final reservoir fish capture (in terms of weight, number and biomass) in each limnocorral a the end of experiment II.

	Treatments / Limnocorral										
	TR 4	TR 6	TR 10	TR+FC 1	TR+FC 8	TT 3	TT 5	TT 7	TT+FC 2	TT+FC 9	Mean
Weight /number tilapia (g)	1540 (10)	2085 (21)	1310 (12)	550 (6)	1370 (12)	3695 (29)	2939 (27)	4930 (41)	4135 (41)	3500 (35)	2605 (23)
Weight/number small tilapia (g)	370 (8)	1170 (45)	125 (3)	135 (5)	335 (13)	1045 (58)	590 (13)	225 (9)	395 (14)	355 (17)	475 (19)
Weight /number other fish (g)	25 (1)	125 (5)	0	0	0	125 (2)	45 (3)	0	0	0	32 (1)
Weight fish larvae (g)	335	295	1865	1715	485	180	185	395	1305	640	740
Total weight fish from lake(g)	2270	3675	3300	2400	2190	5045	3759	5550	6005	4495	3869
Total lake fish biomass (g/m^3)	19.6	32.8	33.5	24.4	21.4	45.4	30.7	50.6	56.9	41.9	35.7
% of initial lake fish biomass	83.3	138.4	124.3	90.7	81.6	46.9	33.3	52.9	55.1	42.7	74.9

Consequently, the initial difference between typical and reduced tilapia biomasses was slightly attenuated from 74% to 61% at the end. Despite such small changes, the basis of original experimental design in terms of treatments definition was maintained, as can be seen in Table V.14.

Table V.14: Overall fish biomass changes during the course of limnocorral experiment II.

	Treatments			
	TR	**TR+FC**	**TT**	**TT+FC**
Total stocked fish biomass (g/m^3)	24.7	78	95	150.5
Total recaptured fish biomass (g/m^3)	28.6	71.4	42.2	99.8
Total average fish biomass (g/m^3)	26.5	74.4	68.6	125.2

All silver carp stocked into limnocorrals were recovered at the end of experiment II. Unfortunately, fish from different limnocorrals were accidentally pooled together making impossible to express the results on a limnocorral basis. However, an overall final weight of silver carp (20,490 g for limnocorrals 1, 2, 8 and 9 together) slightly inferior to the stocking weight (20,985 g), resulted in a negative silver carp growth rate of 2.41% during experiment II.

V.4. Discussion

The limnocorral experiments performed here represent the final practical test on a whole reservoir scale of two restoration techniques based on fisheries management. While in experiment I (wet season), a significant suppression of nuisance colonial blooming algae by silver carp grazing was observed, in experiment II (dry season) several water quality improvements were associated with the reduction in tilapia biomass in addition to the suppression of phytoplankton abundance by silver carp. Considering the lack of synergism between these two approaches, as indicated by the absence of significant interaction effect from RMA, they will be discussed separately.

V.4.1 - Stocking free-roaming silver carp into Lago Paranoá

Since the last decade, the potential for biological control of phytoplankton in Lago Paranoá using silver carp herbivory, has been recommended (Dornelles & Dias-Neto, 1985) and tested for the first time on an experimental basis (Starling, 1989). Since the first experimental evaluation of the impacts of different planktivore species on Lago Paranoá water quality, it has been observed that silver carp efficiently graze on dominant filamentous cyanobacteria during laboratory feeding experiments and prevented phytoplankton from increasing in small "bag-type" enclosures densely stocked with fishes (3,000 kg/ha or 117 g/m^3; Starling & Rocha, 1990).

As shown in the subsequent experimental work (Starling, 1993a), significant suppression of phytoplankton biomass could be achieved by lowering silver carp stocking biomass to a moderate level (850 kg/ha or 40 g/m^3). The "optimal" range of silver carp biomass to be maintained in Lago Paranoá to efficiently control

phytoplankton proliferation was evaluated in large replicated littoral limnocorrals (Chapter IV). The best water quality improvements occurred in limnocorrals stocked with silver carp at low to moderate biomasses (40 and 60 g/m^3, equivalent to 300-400 kg/ha in this situation).

By choosing a similar silver carp biomass in the above experiment I (525 kg/ha or 40 g/m^3), it was possible to produce a significant suppression of *Microcystis* abundance, although total phytoplankton biomass remained unchanged. Consequently, for the limnocorral experiment II, silver carp stocking rate was slightly increased (to 650 kg/ha or 50 g/m^3), resulting in a significant decline in total algal biomass compared to the typical tilapia biomass situation.

Such differences in the successful results obtained between limnocorral experiments I and II may not only be attributed to the silver carp biomass used but could also be a consequence of contrasting abundance and composition of the planktonic community established in the limnocorrals between wet and dry seasons. Following the typical Lago Paranoá seasonal pattern, both total phytoplankton biomass (expressed as chlorophyll-a) and *Microcystis* abundance were relatively higher at adjacent reservoir sampling points during the dry season (experiment II). Furthermore, the relative abundance of net-phytoplankton was twice that in the adjacent reservoir sampling point during the dry season (66% of total chlorophyll-a in experiment II) than during the rainy season (32% of total chlorophyll-a in experiment I).

As discussed in Chapter IV, the success of silver carp in the biological control of phytoplankton is supported by data in the literature, mainly from tropics and subtropics. Many works have already demonstrated a reduction in the relative percentage of net-phytoplankton as a result of silver carp grazing activity (Leventer, 1981; Milstein *et al.*, 1985a and 1985b; Spataru & Gophen, 1985; Carruthers, 1986;

Milstein *et al.*, 1988; Laws & Weisburd, 1990; Costa-Pierce, 1992; Sagi, 1992). However, in only a few situations have such reductions led to an overall decrease in total phytoplankton biomass (Kajak *et al.*, 1975; Buck, 1977; Smith, 1985; Leventer & Teltsch, 1990; Miura, 1990; Telstch *et al.*, 1991). One of the main reasons for these examples being successful is the very high share of large-sized algae in the phytoplankton assemblages in such situations.

However, in most circumstances where silver carp fail to control phytoplankton, zooplankton suppression followed by nanoplankton enhancements and an overall increase in total algae biomass have been recorded (Opuszynski, 1979; Milstein *et al.*, 1985a and 1985b; Spataru & Gophen, 1985; Burke *et al.*, 1986; Barthelmes, 1989; and Laws & Weisburd, 1990). In these studies, zooplankton seems to exert at least an appreciable grazing pressure on phytoplankton communities not so dominated by net-phytoplankton as in Lago Paranoá.

Therefore, the feasibility of suppressing 65% of *Microcystis* abundance and reduce 25% of total phytoplankton biomass in Lago Paranoá by using free-roaming silver carp as a biological control agent must be stressed to be emphasized. In addition to the economic advantage of this low cost biomanipulation technique, stocking silver carp will also bring an ecological benefit by the replacement of the application of a heavy metal (copper sulfate), which has cumulative effect on both sediment and biota ,by a completely natural method.

V.4.2 - Reduction of tilapias over-population in Lago Paranoá

Tilapias, *Tilapia rendalli* and *Oreochromis niloticus*, introduced into Lago Paranoá during the early sixties, exemplify well adapted and opportunistic exotic planktivores that very shortly became the most abundant fishes in this ecosystem (França *et al.*, 1964; Grando, 1989).

The literature contains numerous examples of successful colonization of lakes and reservoirs by tilapias worldwide (Fernando, 1983). Such phenomenon can be easily understood by looking at some specific features of the biology of tilapias.

African cichlids belonging to the genera *Sarotherodon*, *Oreochromis* and *Tilapia*, commonly referred to as tilapias, are herbivorous fishes whose diets are so heterogeneous that they can be divided into three different categories: omnivores, phytoplankton feeders and macrophyte feeders (Lowe-McConnell, 1982). Although Nile tilapia (*Oreochromis niloticus*) and Congo Tilapia (*Tilapia rendalli*) are classified respectively as omnivore and macrophytes feeder, both of them include cyanobacteria, green algae, diatoms, macrophytes, amorphous detritus as common dietary components (Okeyo, 1989). Other food items such as zooplankton, molluscs, insect larvae and adults, fish eggs and embryos are also frequently ingested by tilapias (Caulton, 1977; Campbell, 1981; Lazzaro, 1991). In general, tilapias should better be considered as opportunistic omnivores with a strong tendency towards herbivory (Beveridge & Baird, *in prep.*).

The diet of Nile tilapia and Congo tilapia in Lago Paranoá is very illustrative of omnivory and feeding convergence, as both species feed on detritus (i.e., organic matter from sediment), zooplankton, macrophytes, phytoplankton, periphyton,

benthic invertebrates, fish larvae, and eggs on a similar quantitative basis (Grando, 1989).

To exploit such a variety of food items, tilapia display the ability to shift feeding modes from filter-feeding to visual feeding (Lazzaro, 1991), to biting (Dempster *et al.*, 1993), or to more unusual behaviors such as capturing zooplankton prey trapped in the water surface film (Starling, 1989) and/or digging into the bottom sediment to collect detritus while expelling unwanted particles through the gills (Campbell, 1981).

In addition to this advantageous flexibility of feeding habits, through an efficient exploitation of all variety of food resources, tilapia also present an enormous reproductive capacity. Parental care of brood (Fryer & Iles, 1972), potential ability to exploit zooplankton more effectively than other fishes in their early stages (Fernando, 1994), year-round spawning activity (Kolding, 1993), and plasticity in maturation size according to the prevailing environmental conditions (Lowe-McConnell, *in prep.*) confer on tilapias the advantage of being able to out-compete other fish species in the typical unstable littoral habitats of tropical lacustrine ecosystems.

Despite the extraordinary success of tilapia which gives rise to high fish yields (Fernando, 1994) and plays a limnological role in the circulation of nutrients keeping water bodies productive (Lowe-McConnell, *in prep.*), their excessive proliferation in eutrophic lacustrine ecosystems is of great concern in terms of water quality deterioration.

Although tilapias are among the most important and widespread freshwater fish species worldwide (Fernando, 1991), relatively little attention has been paid to their impacts on water quality and to the evaluation of their use as biomanipulation tool. Through laboratory feeding trials and/or field experiments in aquaria and outdoor tanks involving *Oreochromis* species (*O. aureus* and *O. gallilaea*), Perschbacher (1975), Pierce (1983), Drenner *et al.* (1984b, 1987), and Vinyard *et al.* (1988) have

demonstrated that tilapias, as filter-feeding fish, suppress both net-phytoplankton and zooplankton, enhance nanoplankton, and in some circumstances of net-phytoplankton dominance, also reduce the overall algal biomass. In an evaluation of the trophic cascade effects of Nile tilapia in tropical fish ponds, Diana *et al.* (1991) showed that tilapia significantly suppressed zooplankton but had no effects on phytoplankton, while Elhigzi *et al.* (1995) interpreted the significant reduction of large-bodied zooplankton and consequent increase in the total phytoplankton biovolume in the presence of tilapia as a cascade effect of tilapia preying on *Daphnia*.

Although most of these studies indicate that tilapia feed on net-phytoplankton and may be able to control the abundance of nuisance algae, current evidences of fish contribution in terms of nutrients from both bottom-feeding habits and excessive proliferation may lead to the completely opposite conclusion concerning the suitability of tilapia as a biomanipulation tool.

Bottom-feeding habit is frequently observed in tilapia species especially in sediment-rich fish ponds where fish behave mainly as detritivores (Cardona, 1995). By creating an exclosure area in a tropical fish pond where no access to sediments was given to tilapia, Riise & Roos (1997) showed that sediments in the fishless area were richer in organic matter and benthic fauna as fish channel sedimented organic matter into higher trophic levels through feeding activity. This is advantageous from an aquaculture point of view, as tilapia may decrease the loss of nutrients during the sedimentation process, reducing the relative role of sediment in the overall mineralisation. However, as far as lake restoration is concerned, the maintenance of high nutrient levels in the water column by reducing mineralisation rates through bottom feeding habits is considered one of the most negative effects fish may have on water quality (see Lazzaro, 1987; Northcote, 1988; Gophen, 1990a for useful reviews).

Also, in other different types of ecosystems such as rice fields in Thailand (Chapman & Fernando, 1994), as well as natural habitats in Africa (Campbell, 1981), tilapia species are found to be primarily detritus feeder. By analyzing over 1,000 stomachs of *Tilapia rendalli* and *Oreochromis niloticus* from different areas of Lago Paranoá, Grando (1989) concluded that sediment material (i.e., detritus) made up over 50% of the total amount of food consumed by both species, on a volumetric basis. In such eutrophic ecosystem where a thick layer of organic sediment has been accumulating over the past 30 years of accelerated eutrophication (Altafin *et al.*, 1995), the bottom-feeding activity of the tilapia over-population may represent an important contribution to internal nutrient loading.

As discussed by Starling & Lazzaro (1997), two potential mechanisms may explain the intensification of eutrophication symptoms (i.e., reduction in transparency and increase in turbidity, phosphorus, chlorophyll-a and *Microcystis* abundance) caused by the presence of tilapia over-population in Lago Paranoá: (1) the 'classical' trophic cascade effect of fish preying on zooplankton, reducing grazing pressure and favoring algal development, and/or (2) the nutrient-mediated effect of fish supplying N and P to phytoplankton growth via excretion and disturbance of bottom sediment. Taking into account the bottle-neck between phytoplankton and zooplankton in the food chain of Lago Paranoá, as dominant net-phytoplankton is not subdued to an intensive grazing pressure by abundant microzooplankton (mainly rotifers), Starling & Lazzaro (*op. cit.*) disregard the first hypothesis.

This assumption was confirmed in the present limnocorral experiments where no zooplankton change was associated with tilapia manipulation but a similar deterioration of water quality (reduction in transparency and increases in turbidity, total phosphorus, total chlorophyll-a and *Microcystis* plus *Botryococcus* abundance)

occurred in 'control' limnocorrals containing the typical tilapia over-population from the reservoir.

Although the relative importance of excretion and sediment disturbance in supplying nutrients to algal growth was not specifically investigated here, recent works in the literature have suggested that the enhancement of phytoplankton by benthivorous fish is not due to their physical disturbance of sediment and associated release of nutrients (Qin & Threlkeld, 1990), but is instead mainly caused by the fish excretion of soluble nutrients to phytoplankton growth (Brabrand *et al.*, 1990; Schindler *et al.*, 1993). The importance of the direct contribution in terms of nutrients to phytoplankton via fish excretion was confirmed by measurements of phosphorus release rates by tilapia in the laboratory, as will be discussed in Chapter VI.

As pointed out by Arcifa *et al.* (1995), such evidence of fish supplying nutrients to algal growth indicate that planktivore biomass, in addition to ecosystem trophic state, plankton community structure and nutrient loading rates, also control the strength of the planktivore impact on lower trophic levels and water quality. Indeed, some recent experimental studies have demonstrated that (1) impacts of planktivores are more dependent on fish biomass than on fish feeding behavior (Lazzaro *et al.*, 1992), and (2) filter-feeding omnivorous planktivores interact synergistically with trophic state so that negative impacts of these fish on water quality become more intense with increased eutrophication (Drenner *et al.*, 1996).

It is generally accepted that fish biomass increases directly with ecosystem trophic state (Oglesby, 1977). By examining data on trophic indicators and fish populations from 65 subtropical Florida (U.S.A) lakes, Bachmann *et al.* (1996) demonstrated that total fish biomass was positively correlated with total phosphorus, total nitrogen, chlorophyll-a, and inversely correlated with Secchi depth. Their estimates of total littoral fish biomass, using the same rotenone sampling method as here, reached a

maximum value of 1,046 kg/ha in hypertrophic Lake Hollingsworth which is in the same order of magnitude as the result obtained in the littoral hypertrophic area of Lago Paranoá in the present study (1,492 kg/ha or 176 g/m^3). Among the total of 65 fish species collected in Florida lakes, the only tilapia species present, *Tilapia aurea* was more likely to be found in eutrophic and hypertrophic lakes where it attained higher biomass levels. The same seems to be valid for Lago Paranoá, as higher abundance of tilapia is always found in hypertrophic branches (Grando, 1989). In fact, tilapia over-population in tropical lakes and reservoir is a very common phenomenon (Fernando, 1994; Lowe-McConnell, *in prep.*). For instance, during a 'boom' year in Lake Turkana, Kenya, Nile tilapia production reached approximately 16 000 tons on c.a. 10 km^2, which is equivalent to 16,000 kg/ha (Kolding, 1993).

Since the first study in small "bag-type" suspended enclosures (Starling, 1989), a feed-back relationship has been shown between the huge tilapia abundance and the high trophic state of the reservoir, as the elevated fish biomass was somehow responsible for sustaining cyanobacteria blooms. Although high tilapia stocking rate associated with algal enhancement in such enclosures was considered unrealistic (3,000 kg/ha or 117 g/m^3, estimated by cast-netting in shallow hypertrophic areas), it was not far above the real abundance of reservoir fishes in similar littoral areas estimated in the present study (1,492 kg/ha or 176 g/m^3, in Table V.2).

Accordingly, the large littoral isolated hypertrophic area of Lago Paranoá monitored by Starling & Lazzaro (1997) and subdued to severe eutrophication symptoms in association with a high fish biomass (1,105 kg/ha or 149 g/m^3, called HIGH BIOMASS area, see Chapter II) can not be considered as overpopulated relative to the present estimate using rotenone. In the same study, the other experimental area holding a fish biomass of 403 kg/ha or 42 g/m^3 was correctly considered as a LOW BIOMASS area.

In the present experiments I and II, differences in the initial and final total fish biomasses in limnocorrals may have affected the overall results obtained. In fact, the small difference of 16% in the final fish biomass recovered from limnocorrals of typical tilapia or control (TT) and reduced tilapia (RT) treatments from experiment I may explain why no significant effects of tilapia removal were detected during the wet season. On the other hand, as a substantial contrast of 61% between tilapia treatments was maintained until the end of the second experiment, there were significant water quality improvements associated with the control of tilapia over-population during the dry season.

These results indicate the need to develop a fisheries strategy for Lago Paranoá to ensure that an effective control of tilapia over-population can be achieved by implementing a professional tilapia fishery using cast-nets. In addition to the ecological benefit in terms of water quality improvement, the official permission from the Environment Ministry (IBAMA) to establish a commercial tilapia fishery will also bring direct benefits to the low-income population living in the suburbs of Brasília. The most important restriction for the official permission to establish a commercial tilapia fishery in the reservoir has been overcome by the recent completion of a join project involving CAESB and the Health Institute of Federal District (ISDF *et al.*, 1996) showing that fishes from Lago Paranoá are not contaminated with bacteria, heavy-metals or pesticides.

In a recent experience of massive planktivore removal from the hypertrophic subtropical Lake Appopka (12412 ha, Florida-U.S.A) to achieve export of nutrients, Godwin *et al.* (1994a) have pointed out that success of overall fish removal may be limited by logistics and economics as restricted market for the fish (gizzard shad, *Dorosoma cepedianum*) discouraged large-scale participation of fishermen and fish dealers. It was estimated that a total of 10 metric tons of phosphorus and 30 metric

tons of nitrogen would be exported annually by harvesting 1,400 tons of fish ww/year or 31% of the total lake adult stock of gizzard shad. From another study on the impacts of fish removal from a smaller hypertrophic Florida lake (104 ha), it was shown that a dramatic increase in the abundance of large-bodied zooplankton and a significant decline in phytoplankton biomass occurred during three years following a 85% reduction in fish biomass mostly composed by gizzard shad (Beaver *et al.*, 1994; Godwin *et al.*, 1994b). The successful control of tilapia proliferation in Lago Paranoá is very much dependent on an extremely intense and efficient fisheries effort as tilapia population can rarely be controlled by predation from piscivores and aquatic birds (Lowe-McConnell, *in prep.*).

A very illustrative example of undesirable excessive tilapia proliferation is described for a reservoir (25 ha) receiving heated sewage effluent from a power plant in South Africa (Ferreira & Schoonbee, 1983). Among several fish species introduced to combat the clogging of filter systems due to algal blooms, tilapia (*Oreochromis mossambicus*) dominated the fish community and soon over-populated the reservoir. Tilapia was favored by continuous breeding due to the maintenance of high temperatures throughout the year, and stunting became frequent in the population, as 3+ year old fish attained 150 g instead of 660 g as in previous periods. Since stocking the reservoir with piscivores, such as African catfish and black bass, had failed to control tilapia proliferation, it became necessary to remove up to 1 ton/ha of tilapia per year by fishing. Silver carp juveniles (at 400 fish/ha) were then released to the reservoir to control algal blooms. Analysis of the food consumed by tilapia and silver carp in the reservoir revealed that although the same food items were consumed by both fish species, tilapia consumed relatively much more detritus than silver carp. As stated by the authors, "the obtained results of the feeding biology of both tilapia and silver carp indicate that they both compete for the same food items, with silver carp

being able to filter the phytoplankton more selectively from the water. Since the introduction of silver carp, the need for the suppression of phytoplankton blooms has gradually diminished and the application of copper sulphate to the dam has been terminated".

In the present study, the absence of significant interaction effects between silver carp and tilapia indicates that tilapia and silver carp have contrasting and independent influences on plankton community and water quality. Taking into account that silver carp would act exclusively as planktivore (Starling, 1989; Starling *et. al., in prep.*) while tilapia food habit in Lago Paranoá is omnivorous with a tendency towards benthivory (Grando, 1989), they do not seem to display a high competition for food resources when simultaneously present in this ecosystem. During both limnocorral experiments, no significant effects of silver carp on tilapia biomass were detected, although there was a tendency for tilapia to reach relatively higher biomass levels in limnocorrals where silver carp was absent (17 and 22% higher, respectively for experiments I and II, Tables V.9 and V.13). However, considering the significantly higher amount of tilapia larvae * (< 5 cm TL) produced in limnocorrals with typical tilapia biomass and no silver carp during a previous experiment (t-test, P=0.022, Chapter IV), it seems possible that removal of seston by filter-feeding silver carp may negatively affects the availability of food resources and consequently the survivorship of tilapia in the early stage of larvae.

As well documented in polyculture fish ponds, silver carp has a synergistic effect on benthivorous fish by increasing the availability of food through sinking of fecal pellets while sediment disturbance by benthivorous fish-feeding activity cause an enhancement of seston particles to silver carp filter-feeding (Milstein, 1992). From present data obtained in three different limnocorral experiments involving the addition of contrasting tilapia and silver carp biomasses, it can be predicted that no

synergism is likely to occur between these two fish species when co-existing into Lago Paranoá. On the contrary, the only expected influence of silver carp on tilapia would be an antagonistic relationship resulting from competition between planktivorous silver carp and tilapia larvae for sestonic particles.

V.5 - Conclusions

- The control of tilapia over-population in Lago Paranoá represents a promising management strategy potentially able to reduce turbidity by 46%, total phosphorus by 31%, nuisance blooming algae by 70% and total phytoplankton biomass by 38%, resulting in a 33% increase in water transparency levels.
- Besides the reduction of internal nutrient loading, official permission to establish a commercial tilapia fisheries using standardized cast-nets, will also bring direct socio-economic benefits to the low-income population living in the suburbs of Brasilia.
- In agreement with previous experiments performed in Lago Paranoá and data from the literature, large-sized planktonic organisms, mainly nuisance cyanobacteria, were efficiently grazed by filter-feeding silver carp.
- The introduction of free-roaming silver carp represents an encouraging management strategy to replace the current application of algicide into Lago Paranoá, as it is potentially able to significantly suppress the undesirable bloom forming algae (*Microcystis*) and reduce total phytoplankton biomass.

- In addition to the filter-feeding potential of silver carp to biologically control algae, there is also a great potential for low-cost cage aquaculture of this fast growing fish in Lago Paranoá.
- Considering the absence of synergistic effects between the stocking of silver carp and the control of tilapia, concomitant adoption of both biomanipulation strategies should be recommended to promote significant water quality improvements and then accelerate the restoration of Lago Paranoá, one of the most important urban man-made lakes in Brazil.

VI - FISH CONTRIBUTION TO LAGO PARANOÁ INTERNAL P LOADING:

VI.1. Introduction

The importance of fish as a potential source of nutrients to phytoplankton has been neglected in most biomanipulation reviews (Lazzaro, 1987; Northcote, 1988; Gulati *et al.*, 1990). Although substantial evidence for nutrient supply from the digestive activity of common carp was apparent from enclosure experiments during the 1970's (Lamarra, 1975), fish had not been considered to contribute significantly to internal phosphorus cycling in lakes relative to other external sources (Kitchell *et al.*, 1975; Nakashima & Leggett, 1980). Following some debate (see comment from Shapiro & Carlson, 1982 and reply from Nakashima & Leggett, 1982), more recent evidence has demonstrated that fish may supply phytoplankton with nutrients in several ways. Directly, benthivorous fish feeding activity increases sediment-water nutrient exchange; fish excretion and egestion are responsible respectively for the rapid supply of soluble reactive phosphorus and ammonia, and slower nutrient availability after remineralization; and decomposition of fish carcasses may contribute with substantial releases of phosphorus and nitrogen (see review in Threlkeld, 1987; Northcote, 1988). Indirectly, by size-selective predation on large-bodied zooplankton, planktivores shift of zooplankton community composition towards small-bodied organisms which, following allometric laws, have higher mass-specific excretion rates (Carpenter & Kitchell , 1984).

The combination of enclosure experiments with laboratory-derived data on fish excretion coupled with bioenergetic models has provided further evidence for the role

of fish in lake phosphorus cycling. Mazumder *et al.* (1988) demonstrated that planktivorous fish can increase P recycling rates and therefore enhance phytoplankton biomass in experimental mesocosms where P turnover times were measured using $^{32}PO_4$. Vanni & Findlay (1990) indicated that phosphorus recycling by fish may be higher than that by zooplankton and promote changes in phytoplankton species composition. Brabrand *et al.* (1990) estimated that phosphorus excretion by bottom-feeding fish can exceed external loading. Carpenter *et al.* (1992) showed that littoral feeding by fish was a major input of P to the pelagic system of a lake exceeding the sum of the other P inputs. Using enclosures divided in compartments by 116 μm-mesh Nitex screen, Schindler (1992) provided further evidence of phytoplankton blooms being directly promoted by fish excretion, as phosphorus excretion rates from predators in fish enclosures were responsible for 95% of the variance in chlorophyll-a concentrations. Schindler *et al.* (1993) showed that more than 90% of the recycled phosphorus in a temperate oligotrophic planktivore-dominated lake derived from fish excretion.

There had been indirect evidence for the influence of fish on the supply of nutrients to phytoplankton growth in Lago Paranoá, since the first enclosure experiment where an extremely high fish biomass was used (Starling & Rocha, 1990). In a situation where the dominant cyanobacteria was not subdued by the microzooplankton grazing pressure, direct nutrient supply by fish excretion could be distinguished from the classical trophic-cascade effect as the main cause of algal enhancement.

From the practical test involving the control of tilapia over-population in large enclosed areas of the reservoir (Chapter II), the feasibility of improving water quality by permitting professional fisheries on tilapia has been demonstrated. Although the importance of fish excretion relative to external P loading was highlighted in this

study, estimates lacked precision, as tilapia excretion rates had not been determined. The absence of significant improvements in P and chlorophyll concentrations, and the persistence of excessive growth of nuisance cyanobacteria for 5 years following a dramatic 70% reduction in the external P input to Lago Paranoá in 1992 illustrates the role of sediments and biota in maintaining an intensive internal P loading in this ecosystem (Altafin *et al.*, 1995; Cavalcanti *et al.*, 1997).

The enclosure experiments simultaneously involving silver carp stocking and tilapia removal, described in Chapter V, offer a good opportunity to evaluate the relative importance of P recycled by fish. By combining such data with laboratory measurements of phosphorus excretion rates by tilapia and silver carp it is possible to estimate daily fish contribution in nutrient to phytoplankton and evaluate various scenarios for the implementation of biomanipulation strategies in Lago Paranoá.

The main objective of this Chapter is to evaluate P excretion rates by tilapia and silver carp in the laboratory, to predict direct nutrient supply by fish during enclosure experiments, and estimate the impacts of such biomanipulation-based fisheries management strategies on a whole reservoir scale.

Hypotheses tested are:

i) The rate of soluble reactive phosphorus (SRP) release is influenced by the type of food consumed; bottom-feeding tilapia should exhibit relatively higher P excretion rates than planktivorous silver carp in Lago Paranoá.

ii) Specific P excretion rate decreases with fish growth and smaller individuals are expected to release relatively more P per unit of fish biomass than larger ones.

iii) Significant reduction in total phytoplankton biomass by controlling tilapia over-population (limnocorral experiment II - dry season) was due to nutrient supply by fish rather than an indirect trophic cascade effect of fish predation on zooplankton.

iv) The relative contribution of tilapia over-population to the P budget of shallow areas in Lago Paranoá is substantial when compared to the external external P loading.

v) Although biological control of nuisance cyanobacteria by silver carp selective grazing involves fish addition to the ecosystem, an overall decrease in daily internal P input to Lago Paranoá is expected to occur following the implementation of biomanipulation (concomitant stocking of silver carp and control of tilapia over-population).

VI.2. Material and Methods

To address the above hypotheses, phosphorus excretion rates of tilapia and silver carp were measured under laboratory conditions. These data were used to estimate the future impact of biomanipulation on internal P loading to Lago Paranoá. Laboratory experiments consisted of applying Brabrand *et al's.* (1990) method to monitor P concentration in indoor tank containing fish previously fed in Lago Paranoá.

A pilot experiment was performed to adjust the fish biomass required to be stocked into tanks to produce detectable changes in soluble reactive phosphorus (SRP) or orthophosphate (PO_4-P) and total phosphorus (TP) concentrations over a two-day time period. A total fish biomass of 6 g ww/l (240 g in a volume of 40 liters of pre-filtered reservoir water through a 10-µm plankton net) was chosen based on P concentration and fish biomass data from similar experiments in the literature (Lamarra, 1975; Brabrand *et al.*, 1990). One tank stocked with six juvenile tilapia (three *Tilapia rendalli* and three *Oreochromis niloticus*, mean individual and total weight of 40 and 240 g, respectively) newly collected from a net-cage maintained in

the hypertrophic Bananal Branch was compared to a fishless control tank. Tanks (40-50 L in capacity) were aerated and maintained in the laboratory at room temperature (water temperature = 23 °C; 12 : 12 hs dark : light regime). Water samples for phosphorus analysis (molybdate method and digestion with potassium persulphate, respectively, for soluble reactive phosphorus and total phosphorus; see APHA, 1985) were collected in both tanks during 48 hours at 4 hour-intervals. As both forms of phosphorus increased over time in the fish tank relative to the fishless tank during the course of the pilot experiment, it was possible to calculate specific P excretion rates from tilapia (in µg P/g ww/h) for each 4-h time interval. The experimental procedure proved to be appropriate and was therefore applied to the experimental excretion trials.

In each of the final trials, nine tanks were each filled with 40 l of the same source of water from Lago Paranoá previously filtered through a 10-µm plankton net to remove most of planktonic food particles, such as zooplankton and net-phytoplankton. Silver carp maintained in net-cages and tilapia captured from the reservoir shores by cast-nets were collected between 15:00 and 15:30 h (i.e., the period of the day when fish were assumed to be most actively feeding) and carefully transported to the laboratory. The following three triplicated treatments were assigned to tanks: **Tilapia** (tanks 1-3, stocked with 6-16 fishes, 50% *Tilapia rendalli* and 50% *Oreochromis niloticus*), **Silver Carp** (tanks 4-6, stocked with 6-16 fish) and **Control** (maintained without fish). However, because of fish mortality during both trials, one replicate from each treatment had to be discounted, resulting in duplicate replications only.

Two experiments were performed: Trial I, using groups of 16 fishes (~ 16 g each), and Trial II, using groups of 6 fishes (~ 40 g each), averaging around 250 g of fish per tank in each trial. Increases in soluble reactive phosphorus (SRP) and total

phosphorus (TP) concentrations were monitored every 4 h over a 48-h period. Fish mortality was evaluated by regular inspection of each tank throughout the experimental periods. Both Trials I and II followed the same protocol described for the pilot experiment, water temperature was maintained at 25 ± 1 °C.

Specific SRP excretion rates were calculated for each 4-h interval throughout each experiment. The influence of different sources of SRP determination underestimates was considered, among which: (a) concentrations were frequently below the detection limit (2 µg/l) of the analytical method, (b) sampling intervals were long relative to the rapid P-uptake from picoplankton, nanoplankton and bacteria present in the filtered reservoir water, and (c) there is a possibility that fish with empty guts were among those collected from the reservoir. If this is true, the calculated overall mean SRP excretion rate by fish should not be considered as representative of the actual release rates. Instead, the maximum excretion rates measured throughout the sampling periods, although still under-estimated, were considered to be closer to the actual excretion rates. (see also Discussion of this chapter).

However, considering the rapid transformation of SRP to TP, as a result of its uptake by primary producers, fish excretion rates are best expressed as TP release rates over the experiment duration, until fish stop excreting a considerable amount of P. This period, called experiment linear phase, lasts for about 18-24 hours (at 17 °C; in Brabrand *et al.*, 1990), after which excretion falls to low but still detectable levels for a longer period. Nevertheless, the use of TP excretion rate suffers from the limitation of poor comparison with data from the literature which are usually expressed as SRP.

Daily SRP release rates of tilapia populations during enclosure experiments I and II (Chapter V) were estimated using initial and final fish biomass recovered from each limnocorral, combined with SRP release rates estimated from the laboratory

trials. The relative contribution of fish excretion to phytoplankton growth in limnocorrals was evaluated by simple linear regression analysis between overall chlorophyll-a concentrations in limnocorrals during each experimental period against tilapia daily excretion rates using Systat Statistical Package (Wilkinson, 1989).

Specific SRP and TP release rates of tilapia and silver carp were also used to estimate P budgets in Lago Paranoá following the implementation of various biomanipulation strategies. External P loading rates to Bananal Branch (data provided by CAESB) were compared with the internal P loading supplied by contrasting fish populations in shallow littoral areas (see biomass estimate by rotenone addition described in Chapter V).

VI.3. Results

VI.3.1. Laboratory P excretion experiments

During the pilot experiment, both SRP and TP concentrations increased in the fish tank until the end of the experiment but remained at relatively constant values in the control tank (Figure 6.1). A maximum SRP excretion rate of 0.136 µg SRP/g ww/h was obtained for the overall 48-h interval. As expected from fish digestive activity, excretion rates in terms of TP generally decreased to low levels 16-20 hs after the beginning of the experiment. The mean TP excretion rate during the linear phase of the experiment (0-20 h) was 0.440 µg TP/g ww/h.

A similar overall picture arising from trials I and II, was that SRP and TP concentrations in tanks stocked with fish (mainly tilapia) increased throughout the experiment but remained at low levels in the control tanks (Figure 6.2). Calculated

maximum SRP excretion rates for tilapia and silver carp were respectively 1.576 and 0.737 µg SRP/g ww/h for smaller fish (Trial I), and 0.527 and 0.391 µg SRP/g ww/h for larger fish (Trial II). Similar linear phases for TP release rates, lasting from 0-8 h, were observed in both trials, although some peaks in P release rates had occurred throughout trial I (Figure 6.3). In contrast to a decrease in TP excretion rates from 1.730 for large tilapia to 1.120 µg TP/g ww/h for small tilapia, silver carp displayed an unexpected trend of higher TP excretion rates for large (2.380 µg TP/g ww/h) relatively to small fish (1.232 µg TP/g ww/h). As in the pilot experiment, the occurrence of some TP pulses after the linear phase of the experiment, suggests that fish (mainly tilapia) may have fed on pico- and nanoplankton during trials. However, fish mortalities during the second half of both experiments should also to be considered as an alternative explanation for this phenomenon, despite the immediate removal of dead fish as soon as detected.

VI.3.2 The role of tilapia SRP excretion during limnocorrals experiments

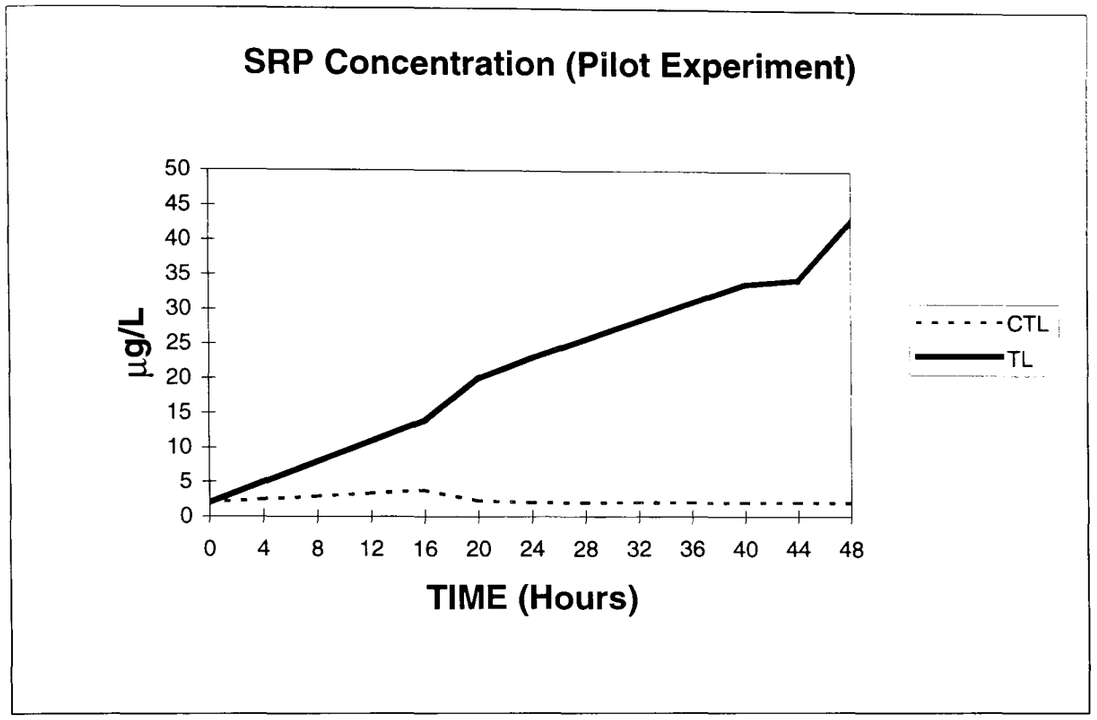
To estimate daily P contribution from tilapia stocked into limnocorrals in experiments I and II (Chapter V), the average of initial and final tilapia biomass values was used in the calculation of total P release rates per limnocorral from Reduced Tilapia (RT) and Typical Tilapia (TT) treatments. Calculations were based on maximum and average SRP excretion rates for smaller fish size (~ 16 g, Trial I). Although only adult tilapias were stocked into limnocorrals, the choice of using excretion rates of younger fish is justified because tilapia biomass in limnocorrals had gradually changed towards smaller fish as a result of fish reproduction. Indeed, fish larvae, which have the highest biomass-specific P excretion rates comprised, on average, 30% and 15% of final tilapia biomass in limnocorral experiments I and II,

respectively (Tables V.9 and V.13, from Chapter V). Calculations of daily tilapia excretion rates are presented in Table VI.1.

To evaluate whether such values are comparable with those reported in the literature, they must first all be converted to an equal unit of fish biomass stocked in the limnocorrals. For instance, given a fish stocking density of 100 kg/ha, the above calculated average daily maximum contribution of tilapia in TR and TT treatments would be respectively 0.303 and 0.294 µg SRP/l/d for experiment I and 0.281 and 0.300 µg SRP/l/d for experiment II. Accordingly, minimum fish contribution calculated from average SRP excretion rates for TR and TT treatments would be equal to 0.110 and 0.107 µg SRP/l/d for experiment I and 0.103 and 0.098 µg SRP/l/d for experiment II.

Published data on daily SRP release rates similarly converted to a fish biomass of 100 kg/ha, would vary from 0.180 µg SRP/l/d (Schindler, 1992) to 0.440 µg SRP/l/d (Vanni & Findlay, 1990). By comparing the above calculated maximum and minimum daily fish input of SRP to limnocorrals with data from the literature, it seems that the calculated maximum SRP excretion rates more closely approximates rates from the literature.

The relative importance of fish excretion in determining the phytoplankton biomass established in limnocorrals during experiments I and II was examined by regressing overall mean chlorophyll-a values for each limnocorral against corresponding calculated tilapia excretion rates. As expected from the absence of chlorophyll-a differences between treatments in limnocorrals of experiment I (Chapter V), there was no significant relationship between phytoplankton biomass and fish excretion rates during the rainy season (R^2=0.138; P=0.629). By contrast, the observed tilapia main effect on chlorophyll-a concentrations during limnocorral



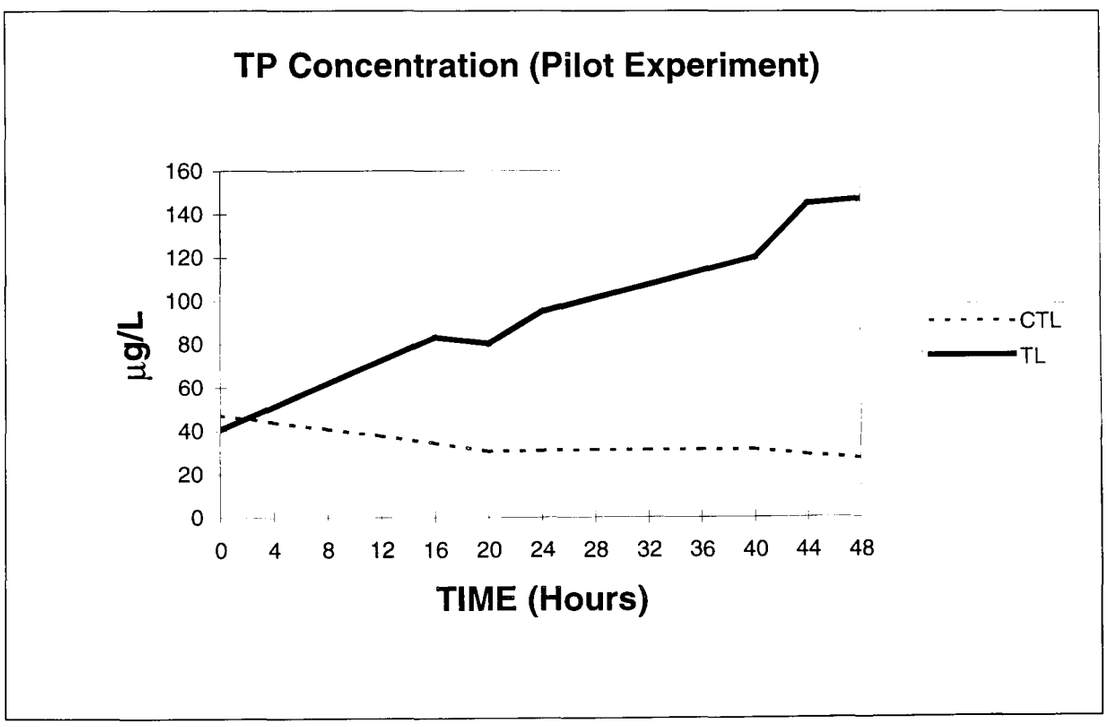
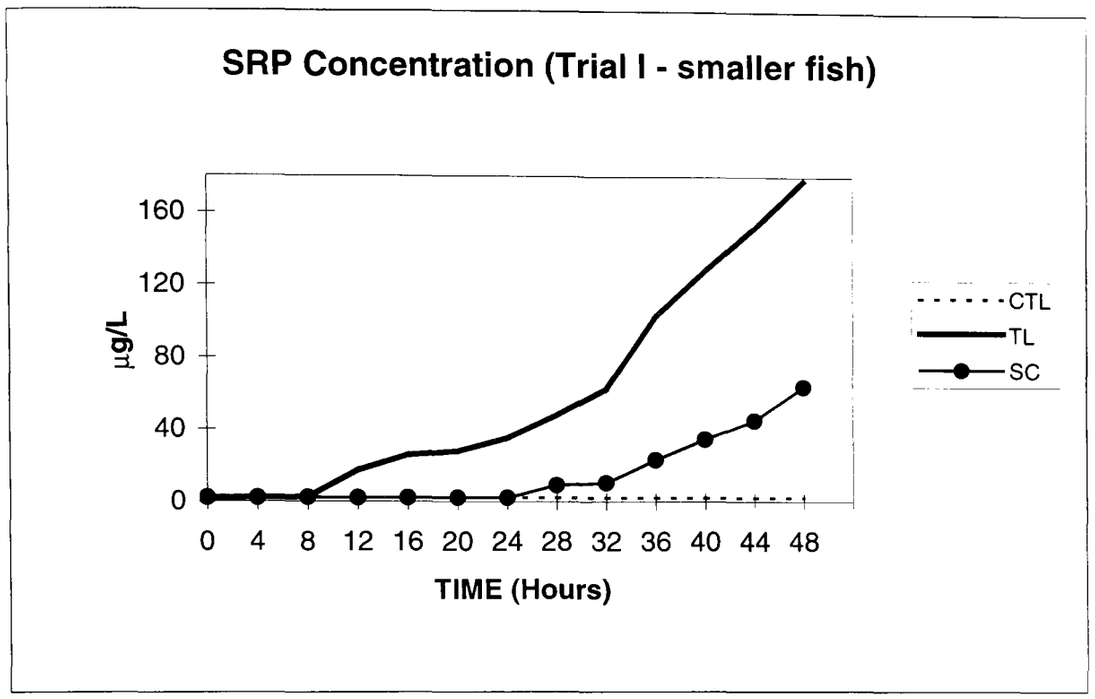


Figure 6.1: Increase in soluble reactive phosphorus (SRP) and total phosphorus (TP) in tanks without fish (CTL) and with tilapia (TL) during pilot experiment.



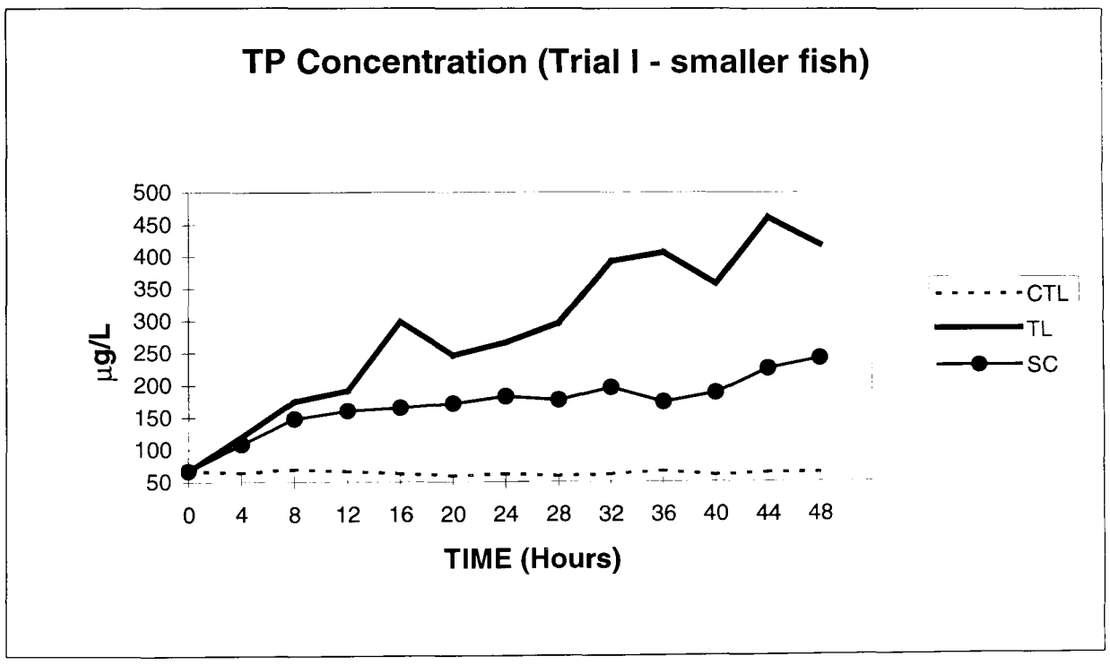
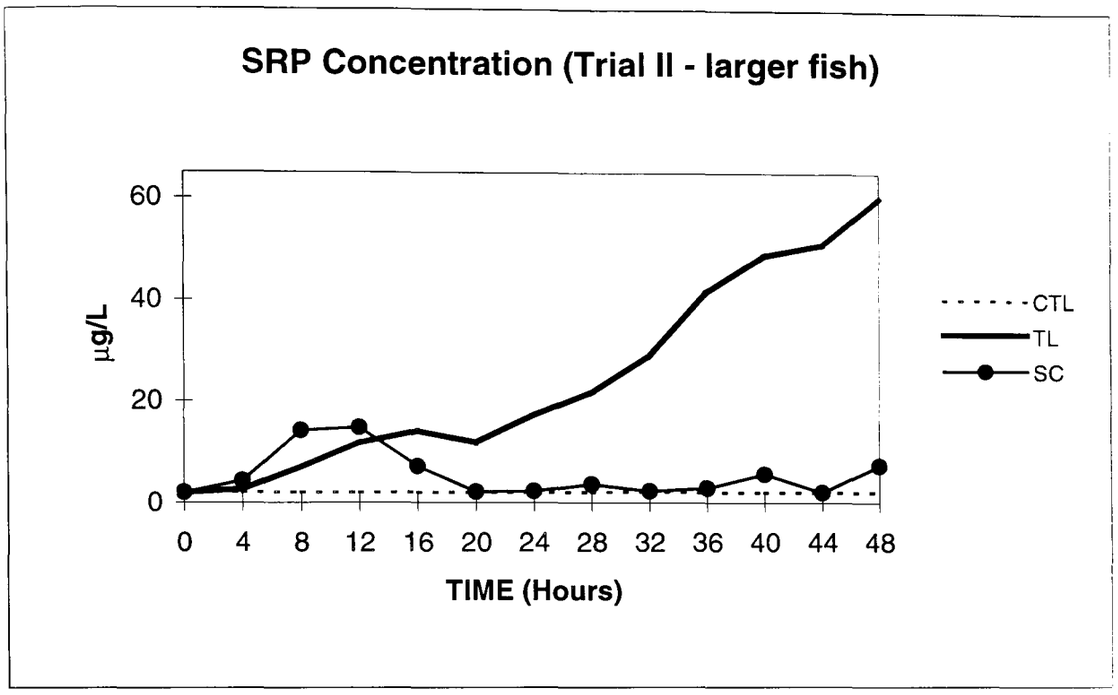


Figure 6.2: Increase in soluble reactive phosphorus (SRP) and total phosphorus (TP) in tanks without fish (CTL) with tilapia (TL) and with silver carp (SC) over the course of trial I (16-g fish) and trial II (40-g fish).



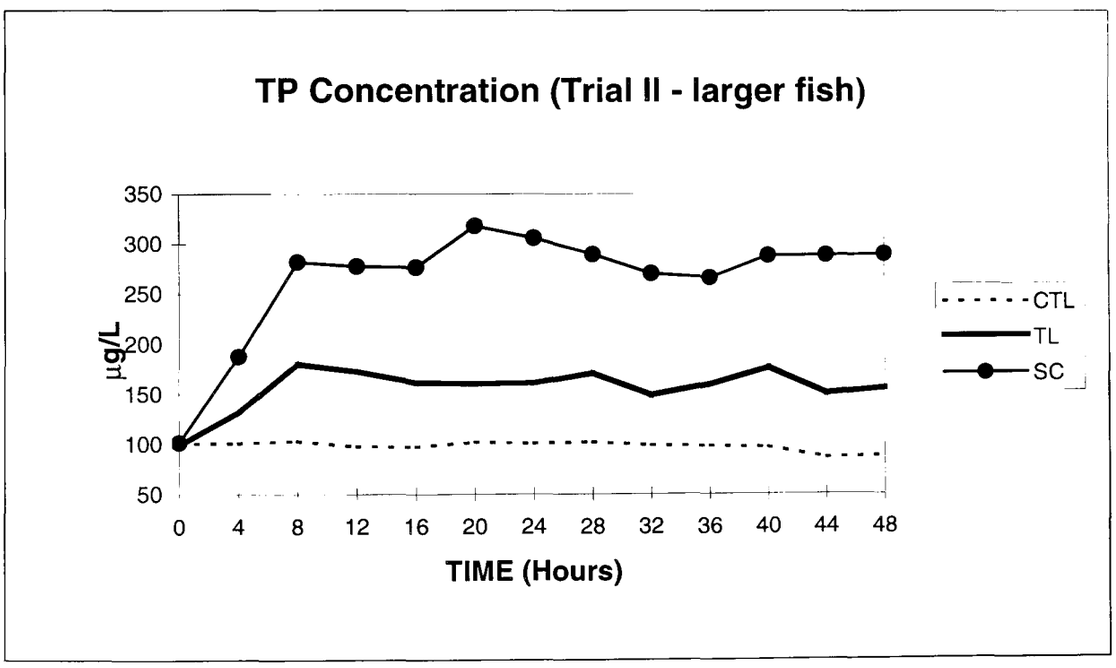
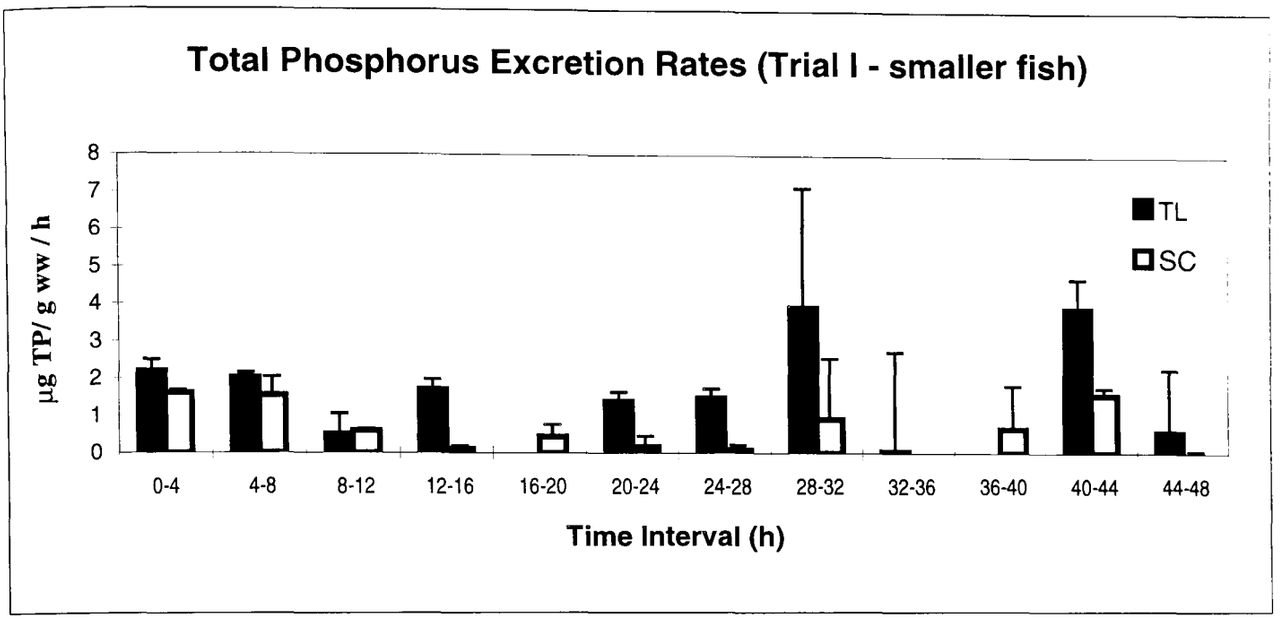


Figure 6.2: (cont.) Increase in soluble reactive phosphorus (SRP) and total phosphorus (TP) in tanks without fish (CTL) with tilapia (TL) and with silver carp (SC) over the course of trial I (16-g fish) and trial II (40-g fish).



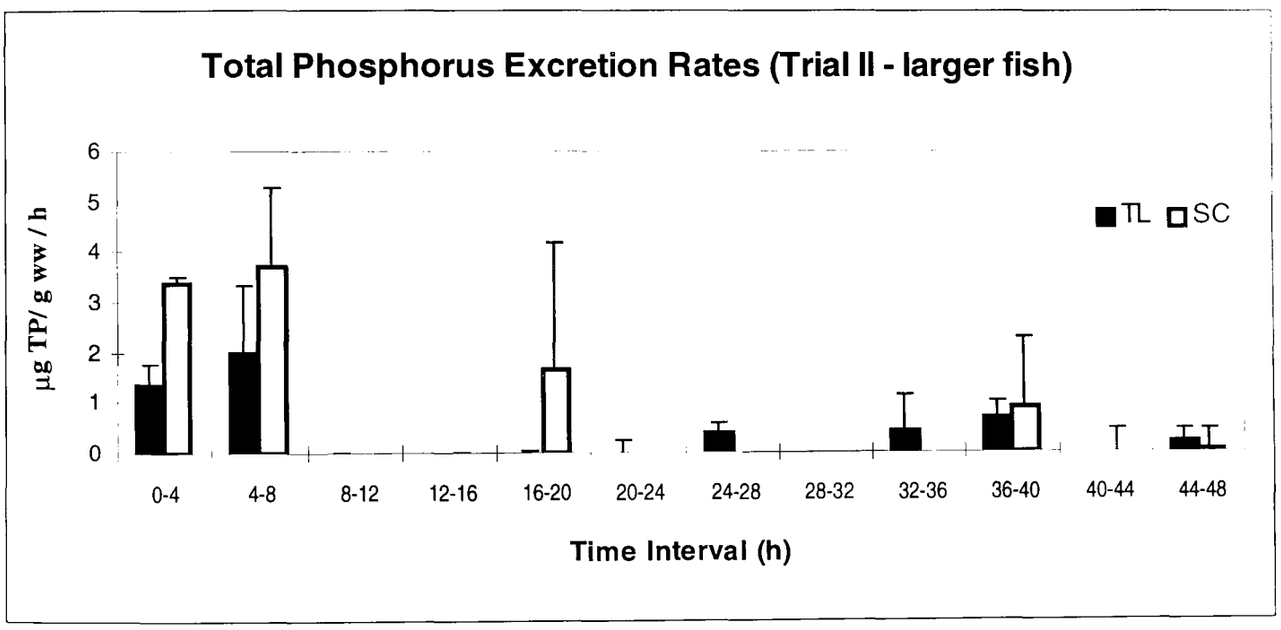


Figure 6.3: Tilapia (TL) and silver carp (SC) TP excretion rates for each 4-h time interval through 48 h of trial I (16-g fish) and trial II (40-g fish).

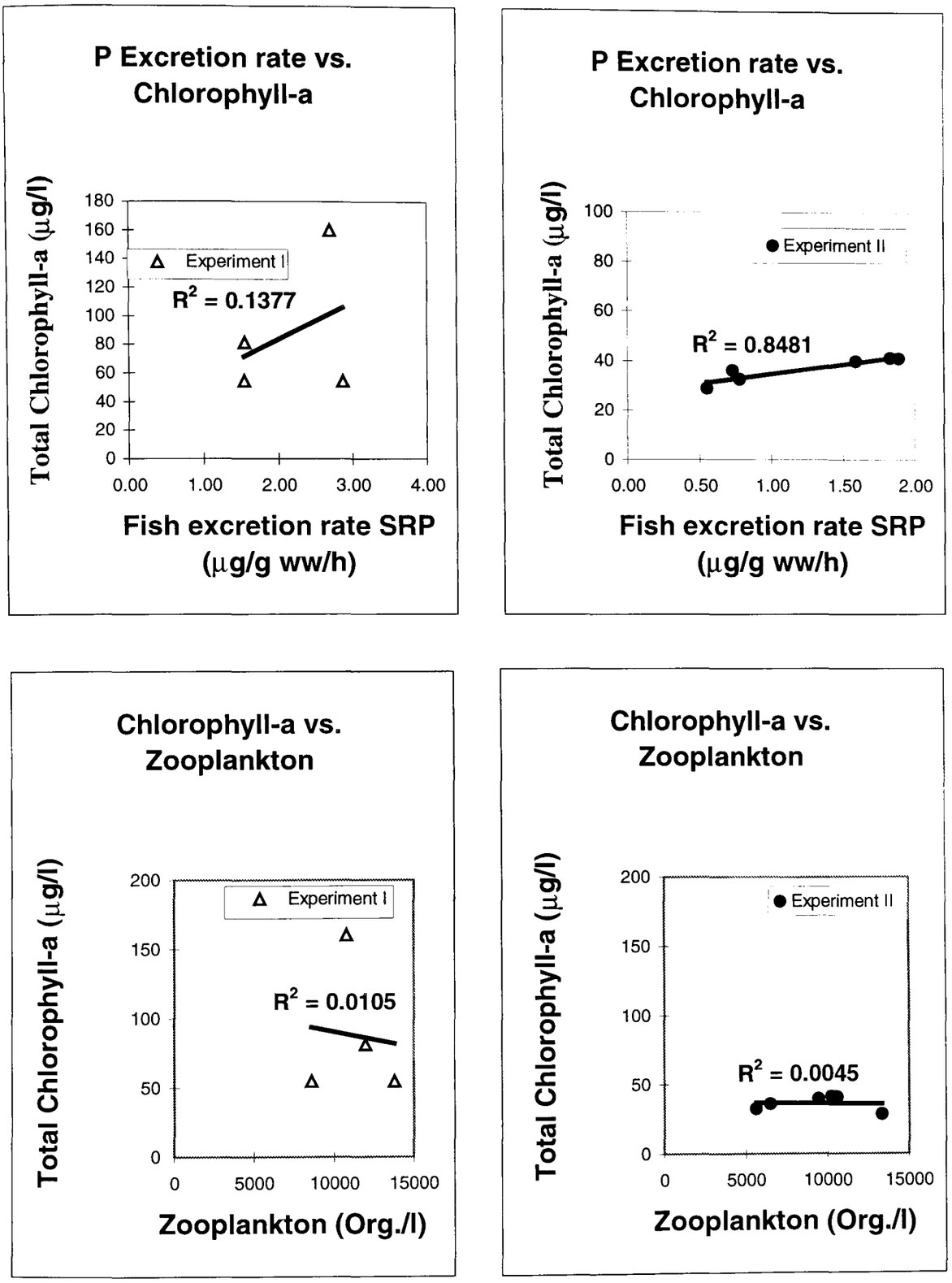


Figure 6.4: Regressions of Chlorophyll-a vs. P excretion rates for overall data from Limnocorral Experiment I (13 sampling dates, F=0.319, P=0.629) and Experiment II (11 sampling dates, F=22.341, P=0.009) and regressions of Chlorophyll-a vs. Zooplankton for overall data from Limnocorral Experiment I (13 sampling dates, F=0.021, P=0.893) and Experiment II (11 sampling dates, F=0,018, P=0.900)

Table VI.1: Estimates of tilapia SRP excretion rates during limnocorral experiments.

Limnocorral experiment I	
Average fish biomass:	Limnocorral 1 (TR) = (4340 + 6980)/2 = 5660 g Limnocorral 7 (TR) = (4050 + 8560)/2 = 6305 g Limnocorral 4 (TT) = (12035 + 11275)/2 = 11655 g Limnocorral 10 (TT) = (13014 + 7890)/2 = 10452 g
Tilapia SRP excretion rates (for 16 g fish):	Maximum = 1.576 µg SRP/g ww/h and Mean = 0.574 µg SRP/g ww/h
Estimates of SRP release rates:	Limnocorral 1 (TR) = 1.576 x 5660 x 24 = 0.214 g SRP/d or 2.27 µg SRP/l/d (max.) 0.574 x 5660 x 24 = 0.078 g SRP/d or 0.83 µg SRP/l/d (min.) Limnocorral 7 (TR) = 1.576 x 6305 x 24 = 0.238 g SRP/d or 2.26 µg SRP/l/d (max.) 0.574 x 5660 x 24 = 0.087 g SRP/d or 0.82 µg SRP/l/d (min.) Limnocorral 4 (TT) = 1.576 x 11655 x 24 = 0.441 g SRP/d or 3.94 µg SRP/l/d (max.) 0.574 x 5660 x 24 = 0.161 g SRP/d or 1.44 µg SRP/l/d (min.) Limnocorral 10 (TT) = 1.576 x 10452 x 24 = 0.395 g SRP/d or 4.19 µg SRP/l/d (max.) 0.574 x 5660 x 24 = 0.144 g SRP/d or 1.53 µg SRP/l/d (min.)
Limnocorral Experiment II	
Average fish biomass:	Limnocorral 4 (TR) = (2725 + 2270)/2 = 2498 g Limnocorral 6 (TR) = (2655 + 3675)/2 = 3165 g Limnocorral 10 (TR) = (2655 + 3330)/2 = 2993 g Limnocorral 3 (TT) = (10760 + 5045)/2 = 7903 g Limnocorral 5 (TT) = (11290 + 3759)/2 = 7525 g Limnocorral 7 (TT) = (10490 + 5550)/2 = 8020 g
Tilapia SRP excretion rates (for 16 g fish):	Maximum = 1.576 µg SRP/g ww/h and Mean = 0.574 µg SRP/g ww/h
Estimates of SRP release rates:	Limnocorral 4 (TR) = 1.576 x 2498 x 24 = 0.094 g SRP/d or 0.81 µg SRP/l/d (max.) 0.574 x 2498 x 24 = 0.034 g SRP/d or 0.30 µg SRP/l/d (min.) Limnocorral 6 (TR) = 1.576 x 3165 x 24 = 0.120 g SRP/d or 1.07 µg SRP/l/d (max.) 0.574 x 3165 x 24 = 0.044 g SRP/d or 0.39 µg SRP/l/d (min.) Limnocorral 10 (TR) =1.576 x 2993 x 24 = 0.113 g SRP/d or 1.15 µg SRP/l/d (max.) 0.574 x 2993 x 24 = 0.041 g SRP/d or 0.42 µg SRP/l/d (min.) Limnocorral 3 (TT) = 1.576 x 7903 x 24 = 0.299 g SRP/d or 2.69 µg SRP/l/d (max.) 0.574 x 7903 x 24 = 0.109 g SRP/d or 0.98 µg SRP/l/d (min.) Limnocorral 5 (TT) = 1.576 x 7525 x 24 = 0.285 g SRP/d or 2.33 µg SRP/l/d (max.) 0.574 x 7525 x 24 = 0.104 g SRP/d or 0.85 µg SRP/l/d (min.) Limnocorral 7 (TT) = 1.576 x 8020 x 24 = 0.303 g SRP/d or 2.77 µg SRP/l/d (max.) 0.574 x 8020 x 24 = 0.110 g SRP/d or 1.10 µg SRP/l/d (min.)

experiment II was mainly a result of P supply by fish, as evidenced by the highly significant regression between chlorophyll-a and P excretion rates (R^2=0.848; P=0.009). Thus, P excretion by tilapia explained 85% of the variability in phytoplankton biomass during the dry season (Figure 6.4). The possible alternative influence of trophic cascade effect through zooplankton grazing on phytoplankton during limnocorral experiments I and II was checked by performing similar regression analysis between total chlorophyll-a and total zooplankton enumeration (assuming numerical abundance is a good estimate of biomass, as overall mean size

of organisms did not vary among limnocorrals during the experiment). The regressions showed that zooplankton abundance did not play an important role in determining algal biomass in either experiment I (R^2=0.011; P=0.897) or experiment II (R^2=0.005; P=0.900).

VI.3.3 Fish relative contribution to the internal P loading in Lago Paranoá.

The availability of specific data on both tilapia biomass levels and P excretion rates in Lago Paranoá makes it possible to assess the role of fish in internal P loading to the Bananal Branch if compared to estimates from Chapter II. Relative importance of current tilapia over-population, external loading and future tilapia and silver carp contributions can be calculated as shown in Table VI.2.

Estimated current daily SRP excretion from tilapia over-population in littoral areas is eight times greater than the average contribution from external SRP loading to this branch of the reservoir. Considering a net removal of 60% of tilapia biomass in littoral areas by legalizing professional fisheries plus the maintenance of a maximum silver carp biomass of 600 kg/ha, future fish contributions to the P budget in littoral areas would be reduced by 50% (from 4680 to 2436 μg SRP/m^2/day). Silver carp excretion rates would be maintained at lower levels (< 564 μg SRP/m^2/day) as compared to external SRP loading (580 μg SRP/m^2/day).

This final picture in terms of fish contribution to internal P loading is very close to the estimates derived from the practical test in large isolated areas (Chapter II). At the same time that tilapia P excretion rates were found to be about three times lower than the literature value used in Chapter II (respectively 1.5 vs. 5 μg SRP/g ww/h), fish biomass in the littoral zone was shown to be approximately three times higher than that estimated from large isolated areas during the previous study (respectively 1,300 vs. 400 kg/ha).

Table VI.2: Relative importance of current tilapia over-population, external loading and future tilapia and silver carp contributions to SRP budget in Lago Paranoá.

Source of contribution	Calculations
(a) **Current overall tilapia SRP contribution in the littoral zone of Bananal Branch**	Tilapia biomass = 1,300 kg/ha (Table .., Chapter 5) or 130 g/m^2 P excretion rate = 1.5 µg SRP/g ww/h (data from 16 g fish) or 36 µg SRP/g ww/day Contribution = 130 g/m^2 x 36.10^{-6} g SRP/g ww/day = 4680 µg SRP /m^2/day
(b) **Watershed SRP loading to Bananal Branch (CAESB data):**	58 kg TP/ day or 5.8 kg SRP/ day or 596 µg SRP /m^2/day (as SRP=10% of TP)
(c) **Future overall tilapia SRP contribution (considering a 60% reduction in abundance):**	Tilapia biomass = 52 g/m^2 (based on biomass data from RT limnocorrals) Contribution = 52 g/m^2 x 36.10^{-6} g SRP/g ww/day= 1872 µg SRP/m^2/day
(d) **Future maximum silver carp SRP contribution (considering biomass ≤ 600 kg/ha):**	Maximum silver carp biomass = 600 kg/ha; or 60 g/m^2 (estimate from Chapter IV) P excretion rate=0.391 µg SRP/g ww/h (data for larger fish) or 9.4 µg SRP/g ww/day Contribution = 60 g/m^2 x 9.4 .10^{-6} g SRP/g ww/day = 564 µg SRP /m^2/day

There are some potential sources of inaccuracy in the present calculations. First, values for tilapia P excretion rates assumed an average weight of 16 g for tilapia population in littoral areas. As fish larvae were not included in the estimate of reservoir littoral fish population using rotenone (Table V.2, Chapter V), the mean tilapia weight of 30 g was an over-estimate. The relative proportions of large and small fish in tilapia biomass can only be evaluated by considering the fish community initially established in recently constructed limnocorrals before starting experiments I, II and III (Table IV.1, Chapter IV and Tables V.1 and V.5, Chapter V). The relative biomass contribution of larvae and adult tilapia (mean weight of 37 g) were respectively 61% and 17% in October 1995 (experiment I, Chapter 5), 49% and 20% in May 1996 (experiment II, Chapter 5) and 32% and 58% in October 1996 (experiment described in Chapter 4). Thus, fish < 3 g (i.e. larvae) make up one third of fish biomass in littoral areas and suggests the use of 16 g as mean tilapia weight is rather conservative. As for allometric reasons smaller fish are known to excrete much more SRP per unit fish biomass than larger fish (Brabrand *et al.*, 1990), P

contribution from overall tilapia population based on a mean fish size of 16 g is most likely an under-estimate.

Second, the future contribution of silver carp to the P budget was based on a mean weight of 40 g, which is much less than the proposed mean size of fish at stocking (100 g). Moreover, a progressive reduction of mean individual silver carp SRP excretion (per g ww of fish) as a result of growth can also be expected to occur. Thus, a considerably lower actual SRP input from silver carp would take place from the time of stocking into Lago Paranoá until the target biomass is reached.

Third, the future scenario in terms of P contribution by the fish community was based on the assumption that tilapia over-population could be reduced by up to 60% by legalizing professional fisheries which use cast-nets. Although there is a great demand for tilapia for human consumption in Lago Paranoá, there is no way to predict whether such drastic reduction in fish stocks could be achieved. Tilapia are well known for their ability to display a series of responses against increased predation pressure including all year-round and precocious reproduction. This may result in stunting and a shift towards small fish sizes which could prevent excretion rates from decreasing substantially, despite a reduction in total fish biomass.

As far as TP is concerned, an alternative picture emerges from estimates of fish P contribution to nutrient budget in Lago Paranoá. Contrasting biomanipulation scenarios from the management of tilapia and silver carp can be calculated as shown in Table VI.3.

These estimates of daily TP contribution from tilapia and silver carp relative to external P loading are rather distinct from the previous SRP estimates. First, current TP contribution from tilapia over-population in littoral areas (5395 μg TP/m^2/day) is of the same order of magnitude as the external TP loading to Bananal Branch (5960 μg TP/m^2/day). Although a reduction in tilapia TP contribution is expected from

increasing fisheries catch effort, stocking silver carp would compensate this loss and no appreciable changes in overall fish internal TP contribution would likely occur. However, a considerable decrease in algal abundance would be expected as a result of silver carp grazing.

Table VI.3: Relative importance of current tilapia over-population, external loading and future tilapia and silver carp contributions to TP budget in Lago Paranoá.

Source of contribution	Calculations
(a) **Current overall tilapia SRP contribution in the littoral zone of Bananal Branch**	Tilapia biomass = 1,300 kg/ha (Table .., Chapter V) or 130 g/m^2 P excretion rate = 1.73 µg TP/g ww/h (mean-smaller tilapia) or 41.5 µg TP/g ww/day Contribution = 130 g/m^2 x 41.5 .10^{-6} g TP/g ww/day = 5395 µg TP /m^2/day
(b) **Watershed SRP loading to Bananal Branch (CAESB data):**	58 kg TP/ day or 5960 µg TP /m^2/day.
(c) **Future overall tilapia SRP contribution (considering a 60% reduction in abundance):**	Tilapia biomass = 52 g/m^2 (based on biomass data from RT limnocorrals) Contribution = 52 g/m^2 x 41.5.10^{-6} g TP/g ww/day= 2158 µg TP/m^2/day
(d) **Future maximum silver carp SRP contribution (considering biomass ≤ 600 kg/ha):**	Maximum silver carp biomass = 600 kg/ha; or 60 g/m^2 (estimate from Chapter IV) P excretion rate=2.38 µg TP/g ww/h (mean-larger fish) or 57.1 µg TP/g ww/day Contribution = 60 g/m^2 x 57.1 .10^{-6} g TP/g ww/day = 3426 µg TP /m^2/day

VI.4. Discussion

VI.4.1. The role of nutrient excretion by fish as a top-down force determining algal biomass

The importance of nutrient supply by fish as a top-down force influencing algal biomass has only been recognized recently (Carpenter *et al.*, 1992; Vanni, 1996). In part this is because in most experimental studies evaluating the impacts of planktivores as both excretion of nutrients and selective fish predation on zooplankton occurred simultaneously in limnocorrals compared to those without fish,

confounding effects (Threlkeld, 1988). Indirect evidence of phytoplankton enhancement caused by fish nutrient excretion has been given in many papers (Andersson *et al.*, 1978; Reinertsen *et al.*, 1986; Drenner *et al.*, 1986; Qin & Threlkeld, 1990; Starling & Rocha, 1990). However, fish-mediated effects of nutrient supply on phytoplankton growth have only been demonstrated by direct measurement of fish excretion rates and estimation of its influence on algal biomass and nutrient budgets in lakes (Lamarra, 1975; Brabrand *et al.*, 1990; Schindler *et al.*, 1993; Marther *et al.*, 1995).

Based on data from enclosure experiments in which algal enhancement had occurred in the presence of planktivores, Vanni & Findlay (1990) showed by simple regression analysis that estimated total P excretion (mainly by fish) was responsible for nearly all of the variance in total phytoplankton biomass (R^2=0.95; P<0.01). Using the same approach, Schindler (1992) demonstrated that 95% of the total variation in chlorophyll-a concentrations between fish treatments during limnocorral experiments could be explained from estimated fish P excretion rates alone. In the present study, tilapia P excretion during limnocorral experiments (estimated using direct measurements of P release rates from laboratory trials) explained 85% of the significant contrast in total chlorophyll-a between treatments during limnocorral experiment II (R^2=0.848; P=0.009). During the same experiment, zooplankton abundance was not found to influence total chlorophyll-a (R^2=0.005; P=0.900). As expected from the weak grazing pressure of abundant microzooplankton on dominant net-phytoplankton (Pinto-Coelho, 1983) and as previously indicated by enclosure experiments (Starling & Rocha, 1990; Starling & Lazzaro, 1997; Chapter V), phytoplankton enhancement by tilapia over-population in Lago Paranoá results from a direct effect of fish supplying nutrients rather than from an indirect trophic cascade effect of zooplankton suppression by fish.

Different phytoplankton groups may display contrasting responses to nutrients supplied by fish and zooplankton. Results from laboratory and enclosure experiments showed that patchy P release by fish increased the dominance of cyanobacteria (*Anabaena)* relative to the more homogeneous release by zooplankton (Reinertsen *et al.*, 1986). Sakshaug & Olsen (1986) demonstrated in laboratory experiments the ability of *Microcystis* to out-compete *Staurastrum* when P was supplied once a day instead of continuously. Some features such as intracellular storage of phosphorus, low but steady growth rates, even at low nutrient concentration, and low light saturation values confer competitive advantages to cyanobacteria in lakes with epilimnetic P supplied by fish (Brabrand *et al.*, 1990). Vanni & Findlay (1990) observed that phytoplankton taxa which require high P supply rates, such as Chlorophyta and Cyanobacteria, were those that had higher abundance in fish enclosures relative to fishless controls. During their experiments, among such dominating taxa in fish enclosures was the colonial green algae *Botryococcus braunni*, an important species present in Lago Paranoá known to exhibit optimum growth at relatively high P concentration. This direct relationship between phytoplankton and P supplied by fish was evident in the present study through the increased dominance of floating bloom-forming Chrorophyta and Cyanobacteria (*Botryococcus braunni* and *Microcystis aeruginosa*) in the presence of high tilapia biomass in all limnocorrals experiments (Figures 5.1 and 5.7, Chapter V).

VI.4.2. P excretion rates by tilapia and silver carp: effects of fish size, species and P fraction

It is well known that phytoplankton take up phosphate as soluble reactive phosphorus (SRP) or orthophosphate (Cembella *et al.*, 1984). The fact that SRP exists in extremely low concentrations in lakes (Taylor & Lean, 1991) makes SRP release

by fish an important potential direct source of nutrients for algal growth (Schindler *et al.*, 1993).

Studies on P metabolism in finfish show that the main loss of phosphate from the body is in the urine, from which about 90% is excreted renally (Lall, 1990). Although estimates of the percentage of soluble and particulate P released by fish to the medium may vary greatly, respectively from 70/30 vs. 30/70 (Dosdat, 1992), direct measurements of common carp excretion rates in terms of SRP, dissolved organic P, particulate P and TP in enclosures showed that 50% of all P excreted by fish is in form of SRP (Lamarra, 1975). Recent laboratory P excretion experiments revealed that 85-95% of the P released by fish is in the form of SRP, with the remainder being precipitated as fecal pellets (Brabrand *et al.*, 1990).

During the present study, even the maximum values for SRP release rates from tilapia (0.53 and 1.58 µg SRP/g ww/h respectively for 40-g and 16-g fish) and silver carp (0.39 and 0.74 µg SRP/g ww/h respectively for 40-g and 16-g fish) obtained from laboratory trials were much lower than values reported in the literature. From the enclosures experiments of Lamarra (1975), common carp SRP excretion rates varied from 1.0 to 10 µg SRP/g ww/h for small fish (10 g) and from 0.4 to 1.0 for large fish (> 500 g). Fish release rates varying from 0.78 to 3.16 µg SRP/g ww/h (Nakashima & Leggett, 1980) for yellow perch and from 2.0 to 7.6 µg SRP/g ww/h (Brabrand *et al.*, 1990) for bream, perch and roach have been obtained in laboratory trials. More recently, Marther *et al.* (1995) reported SRP release rates ranging from 1.0 to 2.3 µg SRP/g ww/h and from 1.2 to 5.7 µg SRP/g ww/h respectively for unfed and fed bluegill sunfish (2 g), and values ranging from 1.0 to 1.3 µg SRP/g ww/h and from 2.0 to 13.7 µg SRP/g ww/h respectively for unfed and fed gizzard shad (18.7 g). The low SRP excretion rates derived from present trials were only comparable to

those from starvation experiments involving common carp (0.2 µg SRP/g ww/h, in Nuttall & Richardson, 1991).

There are two possible explanations for the relatively low SRP release rates obtained in the present study: either fish had not excreted much SRP or for some reason it had not been properly determined from the water samples. First, it is necessary to examine whether such small SRP values might have come from low food consumption in the lake or premature release of urine and/or faeces during transport to the laboratory. Although gut contents of fish collected in the reservoir prior to laboratory experiments had not been analyzed, it seems reasonable to assume by the time of the day fish were captured (from 15:00-15:30 hs) and by the rapid and careful transportation (overall time from fishing to stocking to laboratory tanks was ca. 45 minutes) that at least some of the digestive tracts from experimental fish still contained some food at the beginning of laboratory trials.

Second, during the course of laboratory trials most of the excreted SRP might have been taken up by pico- and nanoplankton and bacteria present in reservoir filtered water before the following sampling four hours later. By measuring the rate of orthophosphate uptake in laboratory, Boyd & Musig (1981) showed that phytoplankton and associated bacteria absorb at least 30 µg SRP/l/h with a maximum uptake of 145 µg SRP/l/h for chlorophyll-a values similar to those recorded in tanks during the present study (46.6 µg/l). During laboratory measurements of zooplankton P excretion rates in tanks containing filtered water from an eutrophic Brazilian reservoir, Pinto-Coelho *et al.* (1997) demonstrated through experimental treatments involving use of antibiotics that bacterial P absorption under-estimates the actual amount of SRP released by zooplankton.

In the present study, SRP concentrations in the water were almost always below detection limits and represented only 2-3% of total phosphorus in fishless tanks

throughout all experimental periods. Although the relative contribution of SRP to TP had increased in fish treatments during the course of both trials, it never represented more than 40% of TP. A similar situation occurs in Lago Paranoá, where SRP usually represents only 10% of TP (Altafin *et al.*, 1995). Thus, it seems reasonable to assume that uptake by primary producers might have been a source of underestimation of SRP excretion rates. Similar experiments in the literature may not have encountered the same problem of under-estimates caused by SRP uptake by phytoplankton and bacteria. Although Lamarra (1975) used unfiltered lake water, samples were taken at shorter 2 h time intervals while Mather *et al.* (1995) conducted laboratory trials in a mixture of declorinated tap water and deionized water. Although Brabrand *et al.* (1990) also used filtered lake water, the occurrence of SRP under-estimation can not be evaluated as no information on the frequency that P was measured in laboratory tanks was provided.

By examining total TP excretion rates for each laboratory trial, the degree to which the low SRP excretion rates may have been influenced by uptake by primary producers can be assessed. Total phosphorus release rates from tilapia (maximum values of 4.47 and 2.00 µg TP/g ww/h and average values of 1.73 and 1.12 µg TP/g ww/h, respectively for 40-g and 16-g fish) and silver carp (maximum values of 1.63 and 3.74 µg TP/g ww/h and average values of 1.23 and 2.38 µg TP/g ww/h, respectively for 40-g and 16-g) were still comparatively lower than the values quoted in the literature. However, the TP release rates for silver carp are comparable to estimates from laboratory experiments involving silver carp starved for 24 hours (1.35 and 2.21 µg TP/g ww/h, respectively, for 40-g and 16-g fish) at 30 °C quoted by Shaolian *et al.* (1991). Thus, it seems that both poor food intake by fish but mainly SRP uptake by phytoplankton and bacteria during laboratory trials could explain the low SRP excretion rates observed in the present study.

In agreement with the literature (Brabrand *et al.*, 1990; Schindler *et al.* 1993), SRP excretion rates were significantly higher for smaller fish (t-test, P=0.07) when compared with larger fish, but did not differ between species (t-test, P=0.221). With regard to TP, the unexpectedly higher excretion rates for larger silver carp may have resulted from increased feeding rhythm or food intake as a response to the plankton abundance at the time of fish capture from cages. Marther *et al.* (1995) explained differences in the absolute P excretion rates of adult gizzard shad between trials as "inescapable" natural variability in feeding from field collected fish.

As pointed out by Bowen *et al.* (1995), phytoplanktivorous fish such as silver carp maximise growth by maximising algal ingestion while omnivorous tilapia increase growth by complementing protein from scarce animal prey by energy from abundant primary but low quality foods such as detritus. These contrasting feeding mechanisms between tilapia and silver carp explain how excessive phytoplankton biomass can be reduced in Lago Paranoá using complementary biomanipulation approaches based on omnivorous fishes: cutting down additional P supply from bottom-feeding tilapia and directly removing dominant net-phytoplankton by silver carp grazing, as illustrated in Figure VI.5. The role of bottom feeding in the P excreted by tilapia was apparent in the comparatively low SRP excretion rates (0.358 µg SRP/g ww/h) obtained for tilapia prevented from assessing sediments collected from net-cages during the pilot experiment compared with SRP values from free-roaming tilapia of the same size during trial II (0.527 µg SRP/g ww/h).

VI.4.3. P contribution by tilapia: reduction of fish contribution by controlling its abundance.

In Lago Paranoá, tilapias (*Tilapia rendalli* and *Oreochromis niloticus*) are opportunistic omnivores which feed on detritus, benthic invertebrates, zooplankton, phytoplankton, macrophytes and fish eggs and larvae (Grando, 1989). Stomach contents analysis from more than 1,000 fish revealed that detritus aggregates from bottom sediments comprises 50% by volume of the food in tilapia gut contents from this ecosystem (Grando, *op.cit.*). There are many works in the literature describing the bottom feeding habits of tilapia (Campbell, 1981; Lowe-McConnell, 1987; Chapman & Fernando, 1994; Riise & Roos, 1997). By feeding on bottom sediments, tilapia recycle "new" phosphorus to the open water, in the same manner as external loading. That is the rationale to compare the contribution from bottom-feeding dominant fish to the external P loading to the ecosystem (Shapiro & Carlson, 1982).

Within a few years of its introduction to Lago Paranoá in the 1960's, tilapia became the most abundant fish in this ecosystem, attaining high biomass in shallow areas (Dornelles & Dias-Neto, 1985; Starling, 1989). During the past two decades, as eutrophication intensified, an increase in tilapia landings has been reported by local fishermen, although no data on tilapia biomass have been collected (Grando, 1989). The only attempt to assess fish stock from Lago Paranoá was an echosounding survey performed almost 20 years ago (Dornelles & Dias-Neto, 1985). Although the resultant picture of 3,000 t is now considered to be an under-estimate, taking into account the progressive eutrophication during this period, this still corresponds to an average reservoir fish biomass as high as 750 kg/ha. As far as shallow hypertrophic areas are concerned, fish biomass can exceed 1,500 kg/ha, of which tilapia comprise nearly 90% (see Table V.2, Chapter V).

Tilapia over-population in shallow hypertrophic areas of Lago Paranoá, where detritus from nutrient-rich sediments is being utilized as the main source of food by fish, has been shown to increase eutrophication and favor cyanobacteria blooms (Chapters II and V) by adding large quantities of SRP from excretion to the water column. Estimates of the total amount of SRP excreted by tilapia in the littoral zone exceeded the external loading to the same area of the reservoir by a factor of eight. Even comparing TP budgets, the tilapia contribution was found to be as important as the external P loading to the Bananal Branch. Shapiro & Carlson (1982) pointed out that the contribution from P excreted by benthivorous fish may be of the same order of magnitude as the external P loading to lakes. Brabrand *et al.* (1990) emphasized the importance of sediment-feeding fish populations in the eutrophication process of lakes by estimating TP supply from bottom-feeding roach to be twice that of the external TP supply during periods of P limitation to phytoplankton.

To evaluate whether present estimates of fish contribution to internal P loading are realistic, the P loading rates for a tilapia biomass of 1,300 kg/ha should be compared to values from the literature. Current tilapia loading rate of 5.4 mg TP/m^2/day in Lago Paranoá is only about twice as high as values calculated for common carp in Kuska Pond (200 kg/ha; 2.18 mg TP/m^2/day, in Lamarra, 1975) and the values for roach in Lake Gjersjoen (300 kg/ha; 3 mg TP/m^2/day, in Brabrand *et al.*, 1990). It is thus clear that the high P contribution from tilapia is due to its abundance in the reservoir rather than to its specific P excretion rates.

Tilapia are well known to frequently occur at high abundance in tropical lacustrine ecosystems, giving enormous fish yields (Fernando, 1994). In addition to the omnivorous diet and plasticity in feeding behavior (Lowe-McConnell, *in prep.*), tilapia have other adaptative characteristics which help them to colonize unstable littoral habitats in such ecosystems. As part of their reproductive strategy, tilapia

display parental care of its brood, year-round spawning, and a high flexibility in growth rate and maturation size according to the prevailing environmental conditions (Kolding, 1993). As a possible response to increased predation pressure, tilapia populations may display a reduction in fish size at maturity known as stunting or dwarfing (De Silva & Amarasinghe, 1989).

These adaptative strategies for keeping high fish biomass and obtain maximum profit from the rich resources in shallow waters despite high predation pressure, represent a potential obstacle to the reduction of tilapia contributions to the internal P loading through opening professional cast-net fisheries in Lago Paranoá. If the tilapia population stunts and displays a greater intrinsic rate of population increase by shortening generation time as a reaction to increased fishing pressure, then the expected reduction in total P excretion rate to some extent may be compensated for by higher weight specific individual P excretion rates.

The lack of experience in fisheries management at whole reservoir scale makes it difficult to predict whether it is feasible to achieve the net removal of 780 kg/ha tilapia in shallow areas by opening a professional cast-net fisheries. However, it seems reasonable to consider that existing high demand for tilapia associated with the high number of unemployed people living in the suburbs of Brasilia will guarantee a permanent high fisheries pressure throughout the year in the Lago Paranoá.

Successful experience in other tropical and subtropical lakes and reservoirs demonstrates the feasibility of harvesting tilapia as a way of removing nutrients from the ecosystem. In Seletar Reservoir (Singapore), a total of 194 tons of *Tilapia mossambicus* was removed in 11 years (55 kg/ha/year) as part of a program involving the use of caged filter-feeding Chinese carp to control eutrophication in the reservoir (Ling, 1982). In order to improve silver carp grazing on phytoplankton and reduce fish stunting in a small reservoir (25 ha) in South Africa, tilapia (*Oreochromis*

mossambicus) over-population was controlled by removing a total of 720-1000 kg/ha/year through fisheries (Ferreira & Schoonbee, 1983). During an exceptionaly productive year, Lake Turkana (Kenya) yielded approximately 16,000 kg/ha of Nile tilapia (Kolding, 1993).

The possibility of reducing eutrophication by nutrient removal through fish harvesting has also been tested in subtropical Florida lakes (USA). In Lake Appopka, during 1993 and 1994, over 400 tons (32 kg/ha) and 800 tons (64 kg/ha) of gizzard shad were removed respectively in 48 and 55 days using floating gill nets. The target fish harvest of 1,400 tons/year which would export 10 tons of phosphorus and 30 tons of nitrogen was not reached because the limited market discouraged large-scale involvement of fisherman in fish harvesting (Godwin *et al.*, 1994b). A contrasting successful removal of 85% of planktivores in a small hypertrophic Florida lake resulted in marked improvements in water quality variables such as phytoplankton biomass (Godwin *et al.*, 1994a).

Although fish growth may represent a substantial P sink, responsible for appreciable losses of epilimnetic P in lakes (Kitchell *et al.*, 1975; Nakashima & Neggett, 1980; Kraft, 1992), the relative importance of fish harvesting as a way to export nutrients from an ecosystem is debatable. Tatrai (1990) proposed the intensification of fisheries in order to reduce nutrients in Lake Balaton as up to 12% of total nitrogen exports from the lake are stored in bream biomass. On the other hand, Voros *et al.* (*in press.*) considered that fish were of greater importance in internal P cycling than in P retention in Marcali Reservoir (Hungary), as only 9% of the total P retained by the reservoir was incorporated into fish biomass.

The amount of P that might be exported from Lago Paranoá by reducing tilapia over-population by 60% can be estimated by considering fish dry mass as 23.9% of wet mass and P content of fish as 2.39% of dry mass (Drenner *et al.*, 1997).

Assuming that the target reduced tilapia biomass can be met within one year period of intensive fisheries, around 780 kg/ha might be removed from the ecosystem. Taking into account the area of Bananal Branch (973 ha), such fish biomass would correspond to 759 tons of tilapia, which represents 4.3 tons of TP being exported from the reservoir. During the same year period, 21.2 tons of TP may have reached the branch from external sources and 11.5 tons of TP would be prevented from releasing to epilimnion through a net reduction in tilapia excretion rates via fish biomass reduction. Thus, although the removal of P incorporated as fish biomass would be more than 2.5 times lower than the concomitant decrease in P supplied by fish excretion, this would represent 20.5% of the total external P loading: a considerable amount. Figure VI.6 summarizes estimates of future TP budget to Bananal Branch in Paranoá Reservoir based on a 60% reduction in tilapia over-population.

Although the present discussion is based on tilapia biomass assessment and corresponding estimates of P contribution for the littoral zone of hypertrophic Bananal Branch in Lago Paranoá, it seems important to consider the possibility of diel movements of fish to the pelagic zone. Despite the scarcity of data on fish migration in Lago Paranoá, there is evidence from 24- hours gill-net captures showing diel inshore-outshore movements of tilapia (Ribeiro, *pers. comm.*).

As demonstrated by Brabrand *et al.* (1990), P transported by roach from the littoral to the pelagic zone was responsible for up to 80% of the pelagic loading in an eutrophic lake. Carpenter *et al.* (1992) also emphasized the importance of littoral feeding by fish as a major input to the pelagic zone of a small temperate lake, and Schindler *et al.* (1993) estimated that 50% of all recycled phosphorus in a planktivore- dominated lake was derived from the littoral zone. As pointed out by

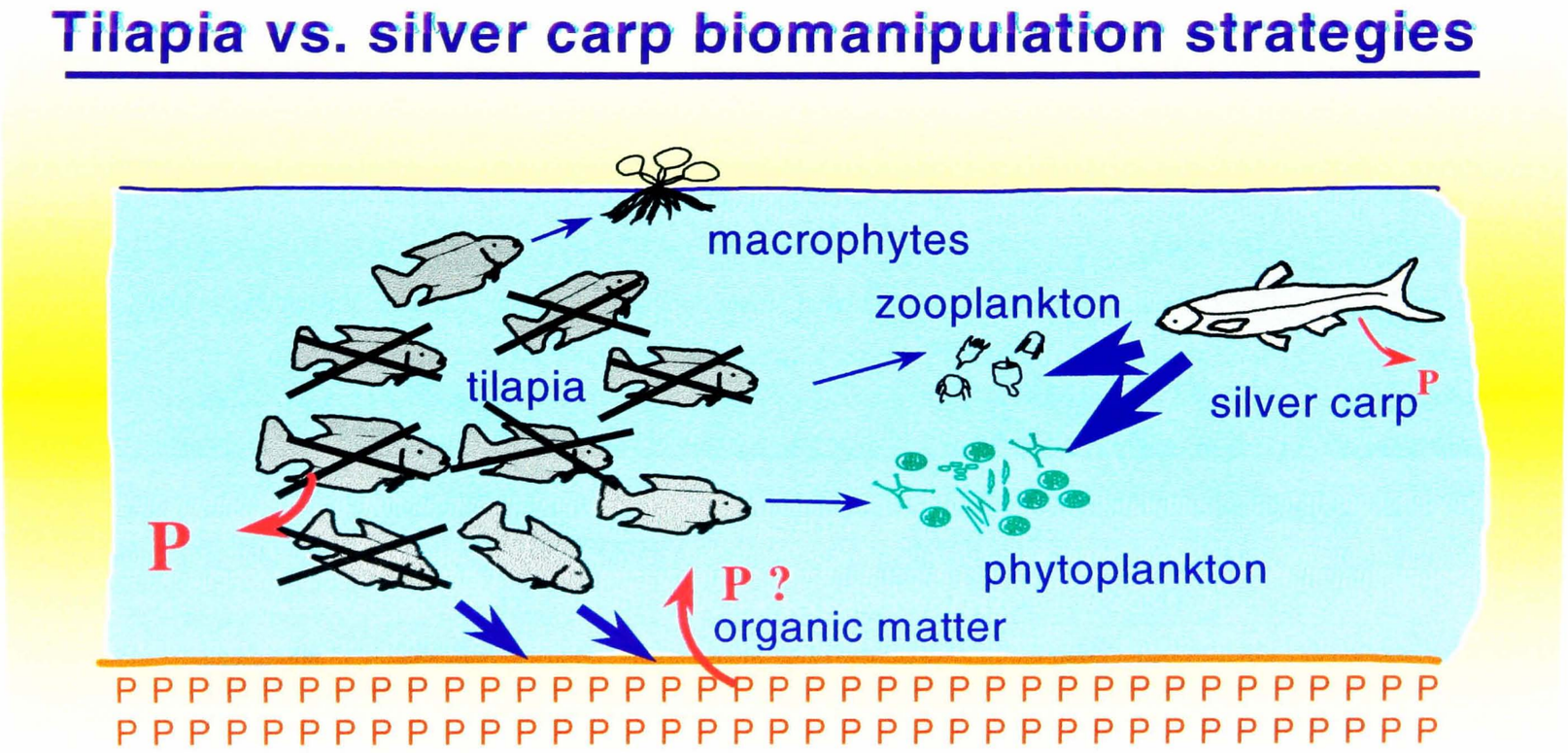


Figure 6.5: Schematic representation of complementary biomanipualtion approaches based on onmivorous fishes in Lago Paranoá: cutting down additiona P supply from botton feeding tilapia and directly removing dominant net-phytoplankton by silver carp grazing.

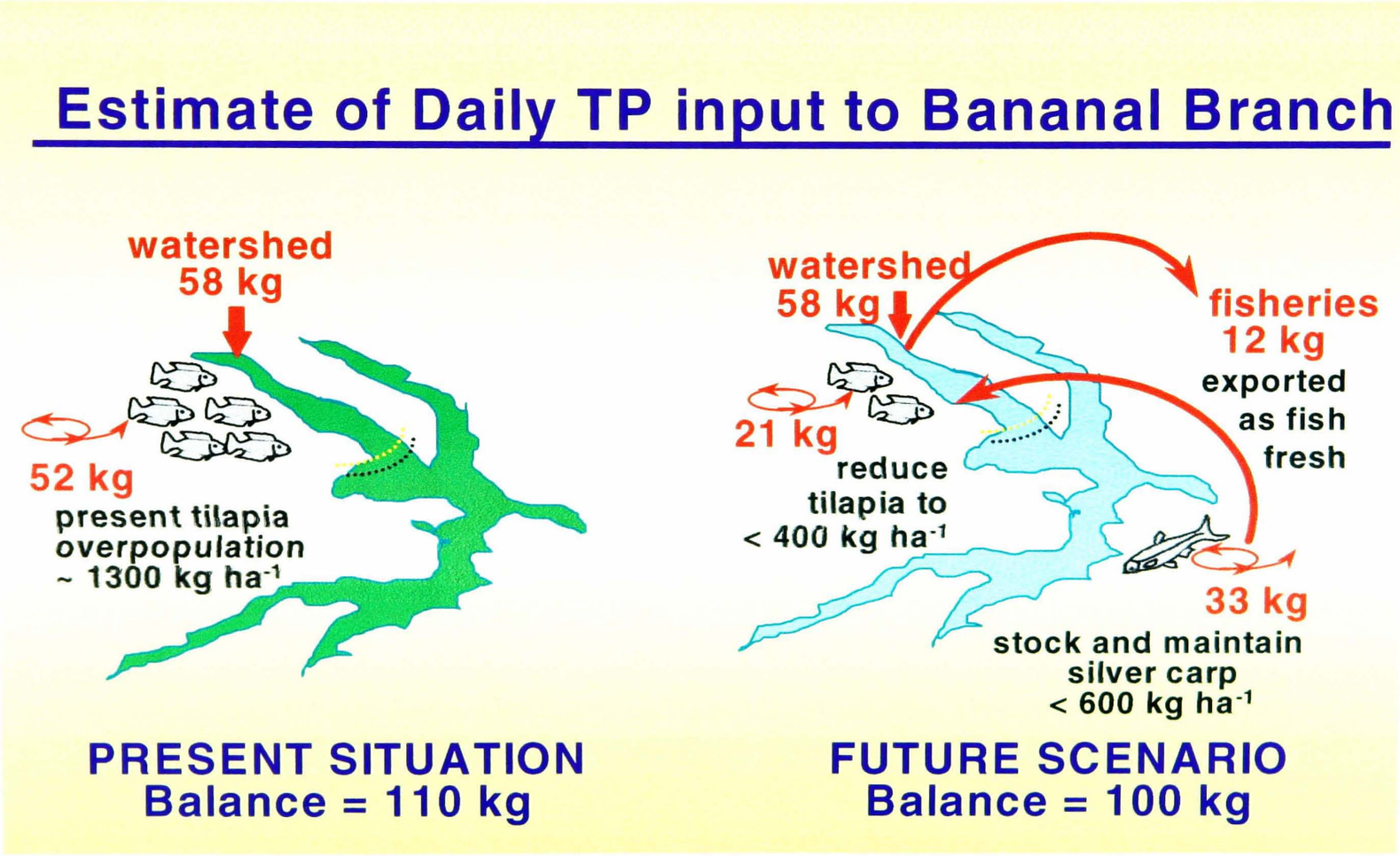


Figure 6.6: Estimates of future daily TP budget to Bananal Branch in Paranoá Resevoir as a result of 60% reduction in tilapia overpopulation.

Lamarra (1975), the role of bottom-feeding fish as a P pump, making a major contribution to the internal nutrient loading is emphasized in ecosystems with extensive littoral areas with highly organic sediments, which is the case with Lago Paranoá.

VI.4.4. P contribution by silver carp: regulating P supply in dimensioned population size

Silver carp is a fast growing fish species unable to reproduce naturally in reservoirs (Costa-Pierce, 1992). Although population size is limited by the number of fish stocked, rapid increases in individual body weight may generate a significant enhancement in total fish biomass. As previously illustrated by experimental studies in the literature (Table IV.8, Chapter IV) and demonstrated by present laboratory P excretion trials, the negative effect of fish excretion on algal growth is directly proportional to fish biomass. From the series of limnocorrals experiments (Chapters IV and V), a maximum silver carp biomass of 600 kg/ha is recommended to be maintained in Lago Paranoá to reduce algal abundance.

Estimates of future P contribution from this target silver carp biomass based on SRP and TP excretion rates in the laboratory suggest that such P supply will neither exceed the external P loading to the reservoir, nor cause an overall increase in P fish supply relative to other sources when compared to the present situation.

Furthermore, a gradual reduction in P supplied by fish to the epilimnion is also expected to occur following the replacement of tilapia by silver carp as a result of two factors. First, from the time of stocking to the reservoir, silver carp (100 g initial weight) will progressively increase in individual mean body size relatively to adult tilapia (mean average weight of 100-200 g at capture). Considering that larger fish excrete comparatively less P per unit weight than smaller ones, there will be a steady

decrease in specific P excretion rates per unit fish biomass. For instance, Lamarra (1975) recorded SRP release rates for common carp larger than 500 g as being in average six times lower than those for 10-g fish, and Shaolian *et al.* (1991) estimated that 100-g silver carp excrete almost four times more TP than 1000-g fish on a per unit body mass basis. Second, exclusive planktophagy from silver carp does not bring "new" P to the epilimnion as the bottom feeding habit of tilapia does. Although some authors have suggested that silver carp may feed on bottom sediment under circumstances of limited planktonic resources (see review in Lazzaro, 1987 and comment and reply in Smith, 1994 and Laws & Weisburd, 1994), sediments were not observed in the digestive tract of silver carp recovered from the three limnocorrals experiments carried out in a very shallow area with nutrient-enriched bottom sediments in Lago Paranoá.

Besides the bioturbation of sediments, one argument frequently used against the stocking of silver carp to directly control phytoplankton abundance in lakes and reservoirs is the poor digestion of some of the ingested algae (Costa-Pierce, 1992). It has been demonstrated that ingested algae can absorb nutrients during passage through the digestive tract of ciprinid fishes and consequently enhance chlorophyll-a and primary productivity when resuspended in the epilimnion (Nakamoto & Okino, 1972; Itawa, 1977; Miura & Wang, 1985; Miura *et al.*, 1989).

Laboratory studies of the digestion efficiency of phytoplankton by silver carp showed that fish reduce chlorophyll-a content and gross phytoplankton primary production by more than 50% and 75%, respectively, during intestinal passage (Miura & Wang, 1985; Miura *et al.*, 1989). It has been shown that both primary production and biomass of green algae (*Staurastrum*) and cyanobacteria (*Anabaena*) are enhanced by fish faeces concentrations of 3 mg/l and 50 mg/l, respectively (Herodek *et al.*, 1989).

Filamentous cyanobacteria, such as *Anabaena* and *Aphanizomenon*, quickly disintegrate in the fore gut of silver carp through the action of highly effective and specialized digestive enzymes (Ruzicka & Ruzickova, 1988; Herodek *et al.*, 1989). By contrast, algae with thick cell walls and envelopes, such as *Microcystis* and *Gomphosphaeria,* may still be found partially digested in the distal section of the digestive tract (Ruzicka & Ruzickova, *op.cit.*; Herodek *et al.*, *op. cit.*). During *in vitro* digestion experiments using silver carp gut fluids, Voros *et al.* (1997) demonstrated that algae with thick cells or gelatinous sheet (Chlorococcales belonging to genus *Scenedesmus, Pediastrum, Oocystis* and *Tetraedron*) decreased by only 20% while Euglenophyta showed a decrease of 40% and filamentous Cyanobacteria (*Aphanisomenon flos-aquae* and *Cylindrospermopsis raciborskii*) were completely digested in the first 10 minutes of the experiment.

As pointed out by Costa-Pierce (1992), the ultimate fate of algae eventually present in silver carp faeces depends very much on whether fecal material sediments to the bottom or floats and breaks apart releasing intact phytoplankton cells into the epilimniun. Thus, in deep areas silver carp filtration may also be able to suppress the abundance of such undigestible algae as fecal pellets sink into the non photic layer, while in shallow areas viable algae present in egested fecal pellets continuously resuspended in the euphotic zone may take advantage of nutrient availability in the fish gut and promote phytoplankton enhancement (Voros *et al.*, 1997).

Although phytoplankton digestibility by silver carp has not been evaluated in the present study, from both high consumption of *Microcystis* and *Botryococcus* by silver carp (Chapter 3) and the significant decrease in algal abundance over 35 to 84 days of silver carp stocking into shallow enclosed hypertrophic area of the reservoir (Chapters IV and V), it seems as though the colonial bloom-forming algae may have been sufficiently digested to prevent their subsequent enhancement. The impact of

silver carp in Lago Paranoá may be to cause an even greater suppression of such nuisance bloom-forming algae as this species is also likely to occupy the deeper, open water areas of the pelagic zone where faeces will not remain in the euphotic zone.

Another important aspect regarding the effects of fish faeces is the direct release of nutrients back into the water column. Part of the fecal P settled to the sediment is released in soluble form during anaerobic processes (Enell, 1987). While Persson (1988) reported that 40% of P excreted by fish remained in sludge, Holby & Hall (1991) estimated that only 2 to 4% of the annual sedimented P from cage fish farms in a temperate lake is released back to the over-lying water. In laboratory measurements of potential maximum resolubilization rates of phosphorus from fish faeces, Eskelinen (1984, *in* Dosdat, 1992) showed that 26-48% of P was released in 15 days (T=17 °C; 50% of the water volume changed every hour), while Pettersson (1988) found that 9-35% of P from fish faeces at concentration of 5,000 mg/l was released back to the medium containing lake water over a 7-d period (at room temperature in the dark, on a shaking table).

Release rates of TP from silver carp faeces (3.45 g dry weight, resuspended in 2 l of water) were reported to be 53%, 56% and 47% respectively on 24, 48 and 72 h (Shaolian *et al.*, 1991), indicating that almost all P resolubilization occurred in the first 24 h. As present laboratory trials lasted 48 h, most of the P contribution from silver carp faeces might have been already included in the calculated specific excretion rates. Indeed, as already indicated, TP excretion rates for 16 g and 40 g silver carp reported by Shaolian *et al.* (*op. cit.*) and those obtained during the present study are very similar.

Although P from silver carp faeces is not expected to be as important as that from urine, because the majority of released SRP from fish comes from excretion not egestion (Brabrand *et al.*, 1990; Lall, 1990), the role of fish faeces as a potential

source of increase in algal biomass could be considered. Laboratory experiments in which incremental amount of faeces of silver carp fed on cyanobacteria (*Anabaena*) were shaken for 5 d in flasks containing lake water under illumination have been carried out by Herodek *et al.* (1989). Neither algal biomass nor primary production were affected by faeces concentrations of ≤ 10 mg/l, but increased by factors of 3 and 4, respectively, at faeces concentrations of 50 mg/l. Establishing the values of 10 mg/l and 50 mg/l as safe and critical limits for faeces concentrations egested over a 10-day period of the experiment, and considering that silver carp produce 10% of their body weight as faeces per day (Herodek *et al.*, 1989), safe and critical silver carp biomass values would be 300 kg/ha and 1500 kg/ha. However, to include a margin of safety, the authors consider that allowable silver carp biomass might not exceed 30-50 kg/ha (I. Tatrai, *pers. comm.*). Although the calculations are based on the unrealistic assumption that silver carp faeces would be maintained in the optimum condition of constant movement and permanent illumination for 5 days, they emphasize the need for keeping silver carp biomass under control.

The maximum target silver carp stocking density of 600 kg/ha for Lago Paranoá has been defined on the basis of practical tests evaluating silver carp effects on water quality and phytoplankton biomass. Experiments were performed in very shallow areas of hypertrophic Bananal Branch, where the potential fish side effects of egesting undigested algae and disturbing bottom sediment would be maximized. However, even being kept in such unfavorable situation at the littoral zone, silver carp improved water quality by controlling undesirable bloom-forming cyanobacteria (Chapters IV and V).

As very active fast-swimming obligate planktivore, silver carp is unlikely to be restricted to the littoral zone of Lago Paranoá, but may prefer open water areas where planktonic resources are equally abundant. In such situations, beneficial effects

resulting from seston removal by filtration, packing and sinking to the bottom sediment as fish faeces are expected to be even more pronounced.

Voros *et al.* (*in press.*) describe an example in which silver carp was successfully used to increase P retention rates in a pollution control reservoir constructed to reduce P inputs into Lake Balaton (Hungary). By stocking and harvesting planktivores (500-2800 kg/ha; 80% silver carp) in two-year cycles, only 7% of the external P load was retained as fish biomass, but 64% was accumulated in the sediment. Fish were found to have an important role in P cycling as the contribution of excretion and egestion was equivalent to 50% of the external P loading. Silver carp filtration activity increased sedimentation of sestonic particles and prevented blooms of filamentous cyanobacteria to occur in Marcali Reservoir relatively to similar adjacent reservoirs suffering algal blooms and lacking phytoplanktivorous fish.

Another practical example of successful control of cyanobacteria blooms (*Microcystis*) by stocking and harvesting silver carp was described by Miura (1990). The role of silver carp recycling nutrients in East Lake (China) was studied by a combination of laboratory experiments, field data and mathematical modeling (Miura, 1989; Shaolian *et al.*, 1991). At the same time that silver carp feeding was found to accelerate the release of nutrients to algal bloom by 40%, it greatly increased the utilization of aquatic production by removing from the water a larger amount of P (5.3% of the total P in the water body) than that released from faeces and excreta (3.4% of the total P in the water body). Shaolian *et al.* (1991) concluded that fisheries have an important role in regulating the ecological balance of East Lake, as since the 1970's they have not only supplied food to the population but also slowed down the eutrophication process.

These examples illustrate the need to develop a well planned and enforced fisheries strategy for Lago Paranoá to maintain future stocking densities of silver carp at suitable levels and guarantee the effectiveness of the biomanipulation technique.

VI.5. Conclusions

- Algal enhancement and cyanobacteria blooms associated with tilapia over-population in Lago Paranoá result from a direct effect of fish supplying nutrients rather than from an indirect trophic cascade effect of zooplankton suppression by fish, as P excretion rates by tilapia explained more than 80% of the variability in chlorophyll-a during limnocorral experiment.
- Although still comparatively lower than data in the literature, maximum SRP excretion rates derived from laboratory trials decreased with increasing fish size, but were not significantly higher for bottom-feeding tilapia than for planktivorous silver carp.
- SRP uptake by phytoplankton and bacteria during laboratory trials and perhaps poor feeding by fish used in experiments may have played an important role in the low values of SRP excretion rates observed during the present study.
- If a 60% reduction in tilapia biomass through the opening of professional fisheries is achieved, an amount of P equivalent to 20% of the annual TP external loading could be harvested as fish biomass and the contribution from the remaining tilapia population to the internal TP loading would represent only half of the external P input.
- Estimates of future P contribution from a target silver carp biomass of 600 kg/ha suggest that it will not cause an overall increase in P loadings relative to other sources when compared to the present situation.

- Even in the unfavorable situation of enclosed shallow hypertrophic areas, where possible side effects of silver carp egesting undigested algae and disturbing bottom sediment would be maximized, target silver carp biomass was found to be under safe level with regard to the production of faeces. It was clearly demonstrated in limnocorral experiments that the species improved water quality by controlling undesirable bloom-forming cyanobacteria.
- Considering the importance of P excretion by fish to the internal nutrient loading, a fisheries strategy must be developed in order to maintain silver carp biomass at beneficial levels.
- The maintenance of eutrophic conditions in Lago Paranoá hypertrophic branches during 5- year period following a 70% reduction in external P input, as evidenced by high Chlorophyll-a values and by the use of algicide to prevent algal blooms in some critical areas, indicates the need to control the fish contribution to the internal P loading in order to accelerate the restoration process.

VII - FINAL REMARKS AND RECOMMENDATIONS

VII.1. Control of exotic omnivorous fish over-population in Lago Paranoá

Excessive proliferation of tilapia was proven to be an important source of internal nutrient loading (mainly phosphorus) which delays the restoration of hypertrophic areas in Lago Paranoá. Experiments in the laboratory, limnocorrals and large isolated littoral areas (Chapters II, V and VI) have provided both indirect and direct evidences of tilapia promoting blooms of cyanobacteria by additional nutrient supply through excretion, rather than from selective predation on zooplankton. Although stomach content analysis (Grando, 1989) demonstrated an active bottom feeding habit of tilapia in this reservoir, the relative importance of P ressuspended from stirring up the mud vs. P excreted from consumed nutrient-rich detritus aggregate have not yet been determined. In addition, the supplementary source of internal nutrient loading represented by P release from sediment in aerobic and anaerobic conditions remains unknown.

Although exposed to predation by carnivorous fishes (*Cichla ocellaris, Hoplias malabaricus* and *Rhamdia sp.*) since their introduction into Lago Paranoá in the 60's (França *et al.*, 1964), tilapia gradually dominated the entire reservoir. They now attain biomass levels of at least 1,500 kg/ha in shallow hypertrophic areas (Chapter V) as a consequence of their enormous ability to colonize littoral habitats, broad and opportunistic food spectrum, and high reproduction efficiency. The scenario of this artificial urban lake largely dominated by exotic omnivorous planktivores responsible for additional water quality deterioration problems propitiate the release of a commercial cast-net fisheries strategy at the whole-reservoir scale. Indeed, the use of

cast-nets, well known as highly selective and efficient fishing gear to capture tilapia, has been banned since the reservoir was filled in 1960 and is still currently prohibited. The main potential obstacle for this implementation has recently been overcome by the completion of a detailed field survey addressing the sanitary quality of the main fish species collected from various areas of Lago Paranoá (ISDF *et al.*, 1996). From a public-health point of view, the absence of hazardous levels of contamination by heavy-metals, pesticides and pathogenic bacteria in fish is in favor of the release of the fisheries prohibition. However, a future fisheries management program should be gradually and carefully adopted to simultaneously control the dynamics of tilapia populations in those hypertrophic areas selected for fisheries release. Evaluation of social and economic benefits to the low-income population of Brasília directly and indirectly involved in activities related to fisheries, fish technologies, and commercialization are critical challenges. Prior to the official release of fisheries prohibition, the spatial distribution and overall fish stock in Lago Paranoá need to be assessed using echossounding technique to identify critical areas where tilapia over-population would be removed first.

An important point which must be addressed is the adaptative response of tilapia against predation/fishing pressure and its ecological consequences. As frequently recorded in the literature, precocious breeding and shortening generation time are part of tilapia strategy to maximize reproductive success under conditions of intense predation or fishing pressure. The resulting "stunting" and "dwarfing" may shift the population size distribution towards smaller fish and larvae which, because of their relatively higher P excretion rates, may prevent P internal loading to fall to the expected low levels. An additional stocking of carnivorous fish may also help to control tilapia excessive recruitment by enhancing predation pressure on smaller fish.

However, the extent to which the overall tilapia biomass would be effectively reduced by fisheries from a practical point of view, is still rather unpredictable.

Besides the expected significant reduction of internal nutrient loading by controlling tilapia over-population there would also be a considerable indirect removal of nutrients bound in fish tissue through fish harvesting. All such sources and sinks of nutrients should be incorporated into a bioenergetic model to explore the role of fish in the P cycling within Lago Paranoá. Additional data on tilapia abundance from critical hypertrophic areas plus measurements of tilapia excretion rates for a broader range of fish size including abundant larvae, are also urgently required for accurately model is the expected outcomes of distinct whole-lake fisheries management scenarios.

The most recent data on P loading to Lago Paranoá show that a further drastic reduction of external P loading to Riacho Fundo Branch and Bananal Branch has taken place as a result of improvements in the tertiary sewage treatment by New Watewater Treatment Plants. For instance, overall external P loading to Bananal Branch decreased from 58 kg/day in 1994 to 29, 19 and 11 kg/day in 1995, 1996 and 1997 respectively (CAESB, *unpubl. Data*). The recent further reductions of about 80% in the external loading, make the fish contribution to the P budget even more important and emphasise the need to implement a complementary fisheries management strategy as soon as possible.

To ensure the success of biomanipulation, more than ever, external nutrient loading should be kept under control by preventing new human settlements in the watershed of Lago Paranoá. In addition, special attention should be paid to the role of bottom sediment as source of potentially releasable "new" phosphorus capable of keeping the reservoir under eutrophic conditions.

VII.2. Stocking sterile free-roaming silver carp into Lago Paranoá

The feasibility of reducing total phytoplankton biomass and controlling nuisance cyanobacteria blooms via silver carp grazing have been successively demonstrated in several experiments using tube-type enclosures and large limnocorrals in Lago Paranoá (Starling, 1993a; Chapters IV and V). Besides determining the desirable range of fish biomass to be maintained for maximizing algal suppression and estimating the expected growth rates of silver carp from stocking size (Chapter IV), some other important information concerning future silver carp population dynamics in Lago Paranoá is urgently needed.

In a recent experimental fisheries campaign throughout the reservoir (Lazzaro *et al.*, *in prep.*), five silver carps ranging from 3 to 14 kg were captured using surface experimental gill-nets (50 m x 2 m, mesh size ranging from 60 to 120 mm) in the neighborhood of areas where they supposedly escaped from enclosure experiments conducted some years ago. This illustrates the efficiency and usefulness of this fishing gear for controlling the number of large individuals in the future and maintaining a desirable size structure of silver carp population.

Despite the evidence indicating that silver carp would most probably stay in the same branch where they have been released, the displacement range of this species between different areas of Lago Paranoá needs to be followed using radio-tracking technique. This will enable evaluation of whether the stocked biomass in a given branch is likely to be consistently altered by silver carp migration. In order to ensure the success of stocking silver carp into the reservoir by increasing their chance to escape from predators, all fingerlings produced by fish farms should be reared to

minimum safe size of 10-15 cm in floating net-cages prior to their release into Lago Paranoá opened water.

In Lago Paranoá, the biological control of nuisance algae by silver carp grazing is expected to be even greater than that observed in limnocorrals (Chapters IV and V), as silver carp is known to occupy pelagic surface areas where potential detrimental effects, such as excretion and egestion, are minimized as compared to the littoral zone. It is encouraging that, even in the most unfavorable situation represented by an enclosed shallow hypertrophic areas (< 1.8-m depth each limnocorral), silver carp significantly improved water quality by reducing phytoplankton biomass and suppressing undesirable bloom-forming cyanobacteria. The major ecological advantage of the proposed food web manipulation is the consequent gradual replacement of the algaecide copper sulphate, a chemical product with accumulative effects on the sediment, by silver carp

VII.3. Environment friendly silver carp cage culture

The usefulness of extensive low-cost silver carp cage culture in Lago Paranoá is supported by the high growth rates obtained for juveniles and adult fish maintained in floating net-cages feeding exclusively on abundant plankton resources from the reservoir (Chapter III). The substantial removal of sestonic particles by filter-feeding silver carp (mainly colonial cyanobacteria as illustrated by digestive tract analysis) represents a potential improvement in water quality. Furthermore, silver carp cage culture would enable the removal of a considerable fraction of the excessive nutrients incorporated as plankton biomass by periodically harvesting the fish.

The adoption of this aquaculture strategy at reservoir scale will depend, however, on its compatibility with other important factors, such as recreation and landscape

values. As part of an immense governmental program, Project ORLA, for stimulating tourism in the Capital of Brazil, many hotels and new clubs will be constructed all along the shoreline of Lago Paranoá to improve aquatic sports and leisure. Thus, the only few areas not dedicated to recreation and landscaping, c.a. neighborhood of Northern and Southern Sewage Plants, would be available for installing net-cages. Because these areas correspond exactly to those mostly affected by eutrophication, they could be transformed into aquaculture parks to produce silver carp at low cost in net-cages with concomitant control of nuisance cyanobacteria.

However, a careful assessment of the local impacts on sediment from confining a high silver carp biomass in cages, plus an evaluation of the economical benefits of producing this additional source of animal protein would be necessary before implementing this activity on a large scale.

VII.4. Potential for developing an specific tropical and subtropical biomanipulation approach

As a general rule, food web manipulations aim at alleviating eutrophication symptoms, and should only be implemented following the control of external nutrient input to the ecosystem. The excessive nutrient enrichment of lakes and reservoirs in tropical and subtropical regions is normally associated with increases in the biomass of omnivorous fish, such as the various tilapia species in Africa and South America and gizzard shad (*Dorosoma cepedianum)* in the southern United States. By reaching a high in-lake carrying capacity, such fish species play an important role in accelerating nutrient recycling and release from nutrient-enriched sediment caused by fish bottom-feeding habits. Thus, once ensuring adequate fish sanitary quality, the control of omnivore excessive proliferation by an oriented intensive fishing effort

may also bring ecological plus socio-economical benefits to many other eutrophic lacustrine ecosystems at low latitudes.

The most important consequence of eutrophication represented by the excessive growth of undesirable net-phytoplankton is aggravated in tropical and subtropical systems as recycling processes are accelerated by higher water temperature throughout the year, whereas large herbivorous zooplankton which could have been potentially able to control algae abundance by grazing are virtually absent. In such circumstances of a bottle-neck at the phyto-zooplankton link, the direct use of vertebrate grazers to suppress nuisance cyanobacteria is the only alternative of any interest. However, as the selected fish species should not reproduce naturally in the environment to avoid additional nutrient supply from excessive fish biomass, the use of sterile individuals is highly recommended.

Despite the great potential for silver carp to control cyanobacteria, some unexpected fish side effects reported in the literature, such as indirect nanoplankton enhancement, potential physical disturbance of nesting and spawning sites of indigenous fish species, plus risk of transmitting allochtonous fish diseases should be carefully considered before embarking in any biomanipulation project involving exotic fish introduction.

The proposed alternative is a complementary internal re-oligotrophication method based on food web intervention. It should be more intensively tested in different ecosystems to build up the basis of a biomanipulation approach oriented towards restoration of tropical and subtropical ecosystems. In common circumstances of budget limitations for applying traditional in-lake engineering techniques in developing countries, the use of an ecologically-based harvesting approach to alleviate eutrophication with clear social and economic benefits for low-income

human populations should receive greater attention from scientists and local authorities.

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