IMPACTS OF AQUATIC VEGETATION MANAGEMENT ON THE ECOLOGY OF SMALL IMPOUNDMENTS

A Thesis

by

TREVOR JASON KNIGHT

Submitted to the Office of Graduate Studies of
Texas A&M University
in partial fulfillment of the requirements for the degree of

MASTER OF SCIENCE

May 2009

Major Subject: Wildlife & Fisheries Sciences

IMPACTS OF AQUATIC VEGETATION MANAGEMENT ON THE ECOLOGY OF SMALL IMPOUNDMENTS

A Thesis

by

TREVOR JASON KNIGHT

Submitted to the Office of Graduate Studies of
Texas A&M University
in partial fulfillment of the requirements for the degree of

MASTER OF SCIENCE

Approved by:

Chair of Committee,	Michael Masser
Committee Members,	Frances Gelwick
	Daniel Roelke
	William Rogers
Head of Department,	Thomas Lacher, Jr.

May 2009

Major Subject: Wildlife & Fisheries Sciences

ABSTRACT

Impacts of Aquatic Vegetation Management on the Ecology of Small Impoundments.

(May 2009)

Trevor Jason Knight, B.S., Delaware State University

Chair of Advisory Committee: Dr. Michael P. Masser

Aquatic vegetation management and fisheries management are inseparable, however conflicts are often perceived between the two. We investigated the impact of biological, chemical, and no vegetation control on the ecology of private impoundments stocked with largemouth bass and bluegill sunfish. The primary purpose of this study was to determine if aquatic vegetation management had significant impact on pond ecology. A secondary purpose of this study was to collect data for a separate descriptive study on the impact of vegetation management on plankton populations.

Nine 0.10 acre ponds were obtained at the Aquaculture Research & Teaching Facility of Texas A&M University in the fall of 2005. Southern naiad (*Najas guadalupenis*) was transplanted into each pond at a stocking rate of one ton per surface acre. One of three treatments was then randomly assigned to each pond. The treatments were replicated three times and consisted of: an herbicide treatment using Reward and Cutrine, a triploid grass carp treatment, and a control treatment. Fathead minnows (*Pimephales* promelas), bluegills (*Lepomis macrochirus*), and largemouth bass (*Micropterus salmoides*) fingerlings were stocked in each pond. The treatments were

initiated on May 31, 2006. Prior to the initiation of the treatments, sampling of each pond occurred for hardness, total phosphorus, nitrite, nitrate, ammonia-nitrogen, dissolved oxygen, turbidity, pH, and temperature. Macroinvertebrate samples were collected from each pond. Post-treatment sampling was conducted on the herbicide treatment and the control at day 2, day 7, day 14, day 28, and monthly thereafter. Post-treatment sampling on the triploid grass carp treatment was conducted at day 14, day 28, and monthly thereafter.

One-way ANOVA tests were conducted on the data using SPSS 15.0, and multivariate analysis was conducted using CANOCO software. Significant differences between treatments were found for the parameters turbidity, macrophyte percent coverage, macroinvertebrate species richness, largemouth bass mean weight, and largemouth mean length. Herbicide application and grass scarp stocking significantly decreased the percent coverage of macrophytes in the ponds. Turbidity was significantly increased in the herbicide and grass carp treatments. Largemouth bass mean weight and length were significantly higher in the grass carp ponds. No significant relationships were found in the multivariate analysis; however, there appeared to be several trends within the multivariate analysis that provide insight into potential ecological relationships between the various parameters. The results of this study provide great insight into the impact that various aquatic vegetation management strategies have on the ecology of small impoundments and will help private pond owners and managers conduct better pond management when dealing with aquatic vegetation problems.

DEDICATION

To my dad,

he always made time to take me fishing and instilled in me a passion for the sport of bass fishing that has become an integral part of my life

ACKNOWLEDGEMENTS

I would like to thank my committee chair, Dr. Michael Masser, for welcoming me into his home when I moved to Texas not knowing anyone, mentoring me for the past several years, turning me into an international traveler, introducing me to many new people and places, including Whataburger, and putting up with me spending too much time fishing bass tournaments.

I would like to thank Syngenta's Jim Petta and the Aquatic Ecosystems Restoration Foundation for funding this research project. I would like to thank the USDA Rio Grande Basin Initiative for their support. I would also like to thank Harry Knight with Applied Biochemists for providing us with algaecides and great conversations over lunches and dinners.

I would like to thank my other committee members, Dr. Frances Gelwick, Dr. Daniel Roelke, and Dr. William Rogers, for their help, guidance, and use of their equipment during this project.

I would like to thank Peter Woods for always making the long drives to the Valley entertaining and comical.

Finally, I would like to thank my parents, Gary and Elvina Knight, for always making sure I had what I needed to succeed and allowing me to pursue my dream of professional bass fishing.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

1. INTRODUCTION

One of the world's most scarce and vital resources is water. Water is required by all living organisms and is crucial to agriculture, industry, and human health (National Science and Technology Council 2007). Less than 1% of the water on Earth is freshwater (Wetzel 2001). Even though freshwater is a finite resource, human population continues to grow. It is estimated that the world population is currently more than 6.6 billion and that by the year 2020 the population will have risen to 7.6 billion people (U.S. Census Bureau 2008). An increasing population will result in a higher demand for water, food, and energy. This will place a higher demand on agriculture, which will require expanses in irrigation usage. The demand for power from hydroelectric plants as well as research into alternative fuel sources will place a higher demand on water for energy production as well (National Science and Technology Council 2007). The availability of freshwater will become the most limiting resource for the human population in the near future.

The problem of water scarcity is of particular concern in the United States. Early in U.S. history, the small American population utilized a virtually unlimited supply of freshwater. The population then expanded rapidly due to modern industrialization and medicine resulting in a population of more than 150 million by 1950 (U.S. Census Bureau 2008). By 2006, the U.S. population doubled. Agriculture, climate change, pollution, and urbanization combined with an increasing population have resulted in a

---

This thesis follows the style of Transactions of the American Fisheries Society .

reduction in the amount of clean freshwater available for use (National Science and Technology Council 2007). The U.S. population is growing by roughly 1% each year with several regions and metropolitan areas growing at much higher rates (U.S. Census Bureau 2008).

One region experiencing rapid population increases is Texas. It is likely that Texas will have a population of more than 25 million by 2010 and more than 50 million by 2040 (Texas State Data Center 2006). The demands of a rapidly growing population in Texas have placed severe stress on the quantity and quality of freshwater in the state. One example of this occurred in the 1950s when claimed water rights in the Rio Grande Valley exceeded the available water due to drought (Texas Water Development Board 2004). There are more than 200 reservoirs supported by 15 major rivers throughout the state of Texas. These waterways provide crucial habitat for fish and wildlife, irrigation water, drinking water and power for millions of people, and recreational opportunities such as swimming, boating, and fishing. Texas must also share some of this water with Mexico as well the Border States Arkansas, Louisiana, New Mexico, and Oklahoma.

The majority of land in Texas is in private ownership. Many of these landowners have constructed ponds on their property for a number of reasons such as livestock watering, irrigation, wildlife habitat, and fishing. It is estimated that there are more than 1 million private ponds in Texas (Schonrock 2005). Most farm ponds and small impoundments in Texas are not managed to maintain good water quality or at their highest potential for fish production. This is problematic, since good water quality is essential for healthy livestock and wildlife and approximately 20 percent of fishing trips

in Texas are on private impoundments (Texas Chapter American Fisheries Society 2005). Since manmade ponds are not natural systems, landowners cannot expect these ponds to manage themselves. They require proper management in order for them to provide the desired benefits of the landowner (Masser 1996).

Regardless of the reasons pond owners constructed their impoundments, they all rely on the availability of adequate amounts of clean water. Pond owners are faced with many issues that negatively impact the amount and the quality of the water they have available. Pond owners have no control over some of these issues, such as annual precipitation, but the majority of problems landowners encounter in managing their ponds are controllable.

The most commonly reported problem that landowners encounter with their ponds is the excessive growth of aquatic vegetation (Schonrock 2005). However, the presence of aquatic vegetation is not always a problem for the landowner or the aquatic ecology of the pond. Aquatic vegetation can have positive impacts on pond ecology. It provides structural complexity, habitat, and food for a variety of aquatic organisms influencing their distribution, reproduction, and foraging behavior (Crowder and Cooper 1979; Bettoli et al. 1993). Increased levels of habitat complexity provide small fish species with shelter from predators. Aquatic vegetation can enhance the recruitment of largemouth bass in a system because of the protection from predators (Maceina and Slipke 2004). The complex structure of aquatic plants also provides substrate for macroinvertebrates, which are a primary source of food for many fish. This leads to an increase in the survival of small fish species such as bluegill (Miranda and Pugh 1997).

Several studies have shown that the abundance of small bluegill decreases following the removal of aquatic vegetation as a result of increased vulnerability to predation (Bettoli et al. 1993; Pothoven et al. 1999). In warm water fisheries, the consensus is that an intermediate level of aquatic vegetation between 15 and 30 % can be beneficial to the overall health of the system (Bettoli et al. 1993; Brown and Maceina 2002; Hoyer and Canfield 1996; Maceina and Slipke 2004; Miranda and Pugh 1997).

Aquatic vegetation becomes a problem when it reaches excessive amounts in landowners' ponds. Aquatic vegetation can have negative impacts such as increased water loss via plant transpiration, hindered navigation, increased siltation, and can negatively impact recreational activities such as swimming and fishing (Pieterse 1990). Aquatic vegetation also harbors noxious aquatic insects such as mosquitoes, which can be vectors for diseases like West Nile Virus (Mulrennan 1962). Excessive vegetation can also hinder irrigation for lawns and crops (Brown and Maceina 2002). Excessive amounts of vegetation can lead to reduced prey capture, limited plankton growth, stunted fish growth, and poor condition in piscivorous fish (Luedke 1987; Pothoven et al. 1999; Sammons et al. 2005). At high levels of habitat complexity, predation success for largemouth bass is reduced (Hoyer and Canfield 1996). When plant stem density is high, the increase in visual barriers makes prey detection more difficult for bass. This is undesirable because many property owners and fisheries management programs want healthy, sustainable bass populations. Prey species such as bluegill also adopt different behavioral strategies when vegetation density is high making their detection more difficult for largemouth bass (Savino and Stein 1982). High macrophyte densities can

lead to hypoxic conditions resulting in fish kills (Kilgore and Hoover 2001). It can also lead to changes in fish assemblage composition with a shift towards assemblages comprised of species more tolerant to low dissolved oxygen levels (Smale and Rabeni 1995).

There are many factors that can lead to problems associated with the overabundance of aquatic vegetation in private ponds. Some of the more common reasons that aquatic vegetation becomes problematic are improper pond construction, excessive nutrient input, and negligent introduction of aggressive plants. Due to the difficulty in achieving eradication once a problem occurs, it is advisable that pond owners take the proper preemptive actions to prevent aquatic vegetation from becoming a problem.

Proper pond construction will prevent most problems associated with aquatic vegetation. Most aquatic plants require shallow water, two feet or less, to become established (Shelton and Murphy 1989). In order to prevent vegetation from establishing in new ponds, pond banks should be sloped at an angle that minimizes the amount of water less than two feet deep. This limits light penetration to the bottom and prevents the growth of aquatic vegetation. A bank slope of either 2:1 or 3:1 is preferred when constructing ponds (USDA 1997).

Another factor to consider is the nutrient concentration of the water in the pond. In a nutrient poor system, water clarity tends to be higher as a result of a low biomass of planktonic algae, which rely on nutrients in the water column. This clear water state lends itself to the establishment of aquatic macrophytes as a result of increased sunlight

penetration to the bottom. Many aquatic macrophytes are successful at establishing themselves in ponds with relatively low nutrient concentrations because they absorb their nutrients from the soil through their root system. Once established, macrophytes can absorb nutrients directly from the water column, which suppresses the growth of phytoplankton. Conversely, in a nutrient rich system, water clarity tends to be low as a result of a high biomass of planktonic algae. The planktonic algal bloom reduces light penetration, which inhibits the establishment and growth of macrophytes. As long as the algae have an adequate supply of nutrients, the algae will out-compete the macrophytes and a plankton dominated system will persist (Scheffer 1998). Pond fertilization is an option landowners can utilize to begin and maintain a healthy algal bloom, which will prevent aquatic macrophytes from becoming established. It is important to note that livestock can have an impact on nutrient levels within the watershed. Waste products from livestock can cause increase nutrient loading and eutrophication of ponds within the watershed. The increased nutrient load from livestock can cause filamentous algal blooms or increased macrophyte coverage.

It is important for landowners to take an active role in preventing the introduction of aggressive aquatic plants into their ponds from outside sources. Landowners should inspect any watercraft they allow onto their ponds to make sure they are not unknowingly transporting aquatic plants into the system from other areas. It is also possible for aquatic plants to be transported during fish stocking. Landowners must also be careful not to transplant aquatic plants into their ponds from wild populations because of the chance of roots, tubers, or seeds of exotic plants being present in the soil attached

to the stocked plants. Landowners wishing to add some aquatic plants to their ponds for aesthetic purposes should purchase native species from reputable dealers.

If a landowner does experience a problem with aquatic vegetation in their pond, then the first step to managing the problem is to properly identify the species of concern. Proper plant identification is critical in order to determine the most effective and efficient method of control. Aquatic plant identification can be a difficult task even for a trained expert. Aquatic vegetation falls into two different categories: native and non-native. Native aquatic plants are those species whose natural range is within the area that they have been found. Non-native aquatic plants are those species that have been introduced to an area outside of its natural range. In Texas, the exotic aquatic plant species are alligator weed, *Alternanthera philoxeroides*, curly-leaf pondweed, *Potamogeton crispus*, egeria, *Egeria densa*, Eurasian watermilfoil, *Myriophyllum spicatum*, giant reed, *Arundo donax*, common salvinia, *Salvinia minima*, giant salvinia, *Salvinia molesta*, hydrilla, *Hydrilla verticillata*, parrotfeather, *Myriophyllum aquaticum*, torpedograss, *Panicum repens*, water hyacinth, *Eichhornia crassipes*, and water lettuce, *Pistia stratiotes* (AquaPlant 2008).

Generally, aquatic macrophytes are broken down into three groups. The three groups are emergent plants, submerged plants, and floating plants. Algae are sometimes considered a fourth group for management purposes. Algae are not true plants even though some branched alga, such as *Chara* and *Nitella*, closely resemble macrophytes. Algae differ from plants because they lack roots, stems, leaves, and vascular tissue.

Other forms of algae include planktonic and filamentous. Most problems associated with algae are related to the branched and filamentous species.

Floating plants are those which float on the water surface. They have roots that typically hang below the water surface and are not attached to the sediment. Floating plants can form dense surface mats that block the exchange of gases and penetration of sunlight. Some examples of floating plants are duckweed, watermeal, water hyacinth, water lettuce, and salvinia.

Emergent plants are rooted plants that have rigid stems allowing them to stand above the water surface and are typically found in shallow water along shoreline areas. Emergent plants can be found in moist soil areas or out to a depth of approximately four feet of water. Some examples of emergent plants include cattails, rushes, reeds, primrose, smartweeds, and water lilies.

Submerged plants are rooted plants that have most of their vegetative mass below the water surface. Some portions of submerged plants can extend above the water surface. Unlike emergent plants, submerged plants normally have soft, flexible or flaccid stems. This is one reason why they do not usually extend above the water surface more than a few inches. Submerged species can be found in water less than one foot deep out to depths of more than 25 feet given sufficient water clarity. Some examples of submerged aquatic plants include naiads, coontail, hydrilla, watermilfoil, and pondweeds.

After the landowner has identified the plant species, the next step is to determine the management method to pursue. The three types of aquatic plant management are

mechanical, chemical, and biological control. Mechanical control involves physically removing the vegetation by means of hand, rakes, mowers, dredges, or by creating water drawdowns to desiccate the plants. While this method usually only requires simple machines or tools, it can become very expensive due to the costs of labor and fuel required. Other drawbacks to mechanical control are disposal of the large amount of plants and the results are usually only temporary making it an inefficient method of aquatic vegetation control in most situations (Noble 1980; Pieterse 1990). For those reasons, mechanical control is not often practiced in private ponds.

Chemical control is a method of controlling the vegetation through the use of herbicides. Herbicides can be classified as either contact or systemic based on their mode of action. Contact herbicides are fast acting and cause plant cell destruction with anything they come in contact with. Systemic herbicides are typically slow acting and must be absorbed by the plants and move to the site of action. The type of herbicide used depends mainly on the type of plant targeted, site characteristics, level of infestation, and budget. The advantage of using chemical control is that it can be applied to precise areas and achieve relatively quick results. The downsides to chemical control are that it can become expensive depending on the size of the area sprayed and some chemicals require an application license to be applied (Pieterse 1990; Maceina and Slipke 2004; Glomski et al. 2005; Nelson and Shearer 2005).

In this study we focused on the use of the contact herbicide Reward (i.e. Reward Landscape and Aquatic Herbicide) and the algaecide Cutrine-Plus. Reward, diquat dibromide [6,7-dihydrodipyrido (1,2-a:2',1'-c) pyrazinediium dibromide], controls

aquatic plants by interfering with electron flow in photosynthesis within green plant tissue resulting in desiccation (Ashton and Crafts 1973; Langeland et al. 2002). Reward is quickly absorbed by the leaves of aquatic plants. Translocation of Reward to other parts of the plant typically does not occur due to the rapid desiccation of plant tissues that come in contact with the herbicide. Reward's chemical structure is not changed or degraded within plants. Reward is broken down on the surface by photochemical degradation. Since Reward is rapidly absorbed by aquatic vegetation, the concentration of the herbicide becomes much higher in plant tissue than in the surrounding water. For this reason, low concentrations of Reward are adequate for controlling aquatic vegetation. Reward is effective at control for many submersed, emergent, and floating-leafed aquatic macrophytes.

Cationic herbicides, such as Reward, are strongly absorbed by negatively charged clay particles and organic matter (Poovey and Getsinger 2002). The strong chemical bonds formed by Reward adsorption to suspended particles render the herbicide biologically and chemically inactive. This can be a problem when trying to use Reward to control aquatic vegetation in turbid waters. High levels of turbidity block Reward's ability to effectively control vegetation at all application rates (Poovey and Getsinger 2002).

Some studies have shown that diatoms and cyanobacteria are sensitive to diquat based herbicides (i.e. Reward) (Peterson et al. 1997). Diquat is highly toxic to some invertebrates such as the amphipod *Hyalella azteca* and the apple snail *Pomacea pludosa*. Diquat is also highly toxic to some zooplankton species such as *Daphnia pulex*

(Emmett 2002). Largemouth bass and bluegill do not appear to be affected by diquat when applied at typical application rates (Emmett 2002). Therefore, there appear to be minimal ecological risks from the proper use of diquat in aquatic systems (Bartell et al. 2000).

Cutrine-Plus, copper triethanolamine chelate, is an algaecide in a liquid formulation containing 9.0% active ingredient with a specific gravity of 1.21. It is effective at controlling algae all three forms of algae: planktonic, filamentous, and microalgae at levels of 0.2 to 0.4 ppm copper (Hallingse and Phlips 1996). Chelated copper formulations like Cutrine-Plus stay in solution longer than copper sulfate and result in better algae control (Watson 1989). Cutrine-Plus acts by inhibiting photosynthesis by binding to the chloroplast membrane and disrupting photosynthetic electron transport (Weed Science Society of America 1994). Copper based algaecides (i.e. Cutrine-Plus) can be toxic to some zooplankton species. One study found *Daphnia magna* to be the most sensitive zooplankton to copper compounds (Mastin and Rodgers 2000). Typically, plankton communities respond to treatment with Cutrine-Plus or Reward by increased growth rates and population biomass as nutrients are made available after the reduction of macrophytes or microalgae (Carter and Hestand 1977a; Hestand and Carter 1978). The copper II ion can be toxic to fish dependent upon pH and water hardness. Copper toxicity for Centrarchids ranges from 700-110,000 µg/L (Washington State Department of Ecology 1992). Chu et al. 1978 found that copper concentrations of 0.040 µg/L were toxic to salmonid eggs, juveniles, and adults at

hardness levels near 20 mg/L. At hardness levels higher than 20 mg/L, salmonids were less sensitive.

Biological control is a method which relies on the use of natural plant enemies to consume or weaken the nuisance vegetation. Typically, biological agents are herbivorous animals native to the exotic plants natural range that feed on some portion of the plant resulting in either death or growth inhibition of the plants. Some examples of such agents that have to be used in the U.S. are the grass carp, *Ctenopharyngodon idella*, the hydrilla fly, *Hydrellia pakistanae*, the milfoil weevil, *Euhrychiopsis lecontei*, and the salvinia weevil, *Cyrtobagous singularis*. Some biological agents such as the salvinia weevil are host specific and will not have adverse indirect impacts on non-target plant species. Other agents such as grass carp are less selective and may potentially have adverse indirect impacts on non-target plant species. The advantage of using biological control is that the cost is relatively low compared with other methods of vegetation control particularly over time as they usually are effective for several years at controlling aquatic macrophytes. Another advantage is that biological control does not carry with it any water use restrictions that many herbicides have. Therefore, biological control can be used in situations where water use restrictions may be a problem such as with irrigation water, drinking water reservoirs, etc. The major drawbacks of using biological control are the possibility of negative impacts on non target species, seasonal availability of some agents, the imprecision of application, and slow action. For example, grass carp have been stocked in one cove of a lake to try and control some non-native macrophyte present, but the carp moved to another part of the lake and fed on

another macrophyte present that they may have found to be more palatable resulting in an increase in the amount of the non-native macrophyte due to less competition. It is experiences like this that make it imperative that the landowner determine the proper management option in order to achieve the desired results.

Grass carp are the most common means of biological control utilized in private impoundments in Texas and the Southeast U.S. The grass carp's native range is the river systems of eastern Asia. Grass carp have been introduced into more than 50 countries and were introduced in the U.S. by the U.S. Fish and Wildlife Service in 1963 for the purpose of controlling aquatic vegetation (Masser 2002). Grass carp are a member of the Family Cyprinidae, which includes the minnows and carps. Grass carp differ from common carp, *Cyprinus carpio*, by having a more elongated body and the lack of barbels around the mouth. The dorsal and anal fins are short and spineless while the caudal fin is deeply forked (Masser 2002). All Cyprinidae have pharyngeal teeth. Pharyngeal teeth are in two rows and enable the grass carp to shred the vegetation it consumes. Grass carp have similar water quality requirements to those of most warmwater fish. Dissolved oxygen concentrations of 4 mg/L or higher are preferable for maximum food consumption. Grass carp can tolerate moderate salinity but consumption decreases as salinity increases above 1.3 ppt. Prolonged exposure to 9 to 10 ppt salinity can be lethal (Masser 2002).

Grass carp have proven to be effective in controlling many species of algae and submerged aquatic vegetation (Shireman and Smith 1983). Since their introduction, grass carp have been stocked in most states. Diploid grass carp have escaped into U.S.

river systems and appear to have established reproducing populations in the Mississippi, Missouri and the Trinity River drainages. Many state fisheries managers feared that grass carp might devastate beneficial native flora in public waters, which prompted many states to ban or restrict further stocking (Allen and Wattendorf 1987; Masser 2002). The introduction of grass carp to an impoundment can result in increased nutrient levels as grass carp consume macrophytes and excrete the digested plant matter back into the water column. The increase in nutrients and removal of macrophytes allows phytoplankton to utilize the available nutrients and create a sustainable algal bloom. The increase in phytoplankton biomass can lead to an increase in zooplankton biomass as well. The increased plankton biomass and disturbance of sediment caused by a lack of macrophytes reduces water clarity (Bettoli et al. 1993; Kirkagac and Demir 2004). Due to the potential ecological impacts and previous illegal stockings of diploid grass carp, only triploid grass carp are allowed to be stocked in Texas waters. Triploid grass carp have 3 sets of chromosomes rendering them sterile. This is achieved by exposing fertilized grass carp eggs to hydrostatic pressure causing the eggs to retain an extra set of chromosomes that normally are expelled during cell division (Cassani and Caton 1986).

Traditionally, aquatic vegetation management research has focused on determining the effects of various management options on the target and non-target plant species. Other areas of aquatic vegetation management research have focused on studying human dimensions related to aquatic vegetation management or the impact of management on a single niche in the aquatic ecosystem. There has been limited research concerning the ecological impacts of aquatic vegetation management (Harman et al.

2005). A lack of information concerning ecological impacts means pond owners and managers are taking potentially large economical and ecological risks in conducting aquatic vegetation management without knowing the full spectrum of consequences for such actions. Therefore, it is imperative that more research be conducted to determine the effects of vegetation management on the ecology of small impoundments.

The primary objective of this study was to determine the impacts of the two most widely used vegetation management options, chemical and biological control, on several aspects of pond ecology in small impoundments. The study focused on monitoring macroinvertebrates, fish assemblages, and macrophytes. Water quality was also monitored during the course of the study in an effort to distinguish between biotic and abiotic impacts. A secondary objective of this study was to gather data on phytoplankton and zooplankton assemblages to develop a descriptive study of the impacts of our chosen aquatic vegetation management protocols on the plankton community. This study focused on small impoundments for the purpose of benefiting landowners and managers of small private ponds in Texas. The information gathered from this study will help the Texas AgriLife Extension make recommendations to pond owners and private managers concerning the best management practices for their ponds.

The experimental hypothesis was that in comparison to control impoundments (i.e. no vegetation control): (1) chemical treatment with Reward and Cutrine Plus would result in decreased percent coverage of macrophytes, increased growth rates of individual bluegill and largemouth bass, decreased survival of juvenile bluegill, decreased survival of largemouth bass, increased nutrient concentrations, and increased

turbidity due to algal blooms; (2) treatment with grass carp would result in decreased percent coverage of macrophytes, increased growth rates of individual bluegill and largemouth bass, decreased survival of largemouth bass, decreased survival of juvenile bluegill, decreased macroinvertebrate taxonomic diversity, increased nutrient concentration, and increased turbidity due to grass carp activity.

2. METHODS

Nine 0.10 acre ponds, located at the Texas A&M University Aquaculture Research and Teaching Facility (ARTF) near Snook, Texas, were utilized for this two year study. In December 2005, six months prior to initiation of the study, ponds were planted (one ton per surface acre) with mature southern naiad (*Najas guadalupensis*), and the vegetation was given several months to become established within each pond. Southern naiad was chosen because it is a native macrophyte in Texas, is readily eaten by grass carp, and can be effectively controlled with diquat herbicide. Southern naiad was established in the reservoir of the ARTF and therefore could be easily transplanted into each treatment pond.

The three vegetation treatments in this study were 1) chemical control using diquat (Reward®, Syngenta, Greensboro, North Carolina) in conjunction with chelated copper (Cutrine Plus®, Applied Biochemists, Germantown, Wisconsin), 2) biological control using triploid grass carp (*Ctenopharyngodon idella*), and 3) no treatment of vegetation (the control). Reward was used for the chemical treatment because it's active ingredient, diquat, is a contact herbicide that is effective for controlling Southern naiad (Hiltibran et al. 1972). Cutrine Plus was applied in separate applications to help control branched algae, in particular *Chara* sp., which is a common invasive in the ponds at the ARTF. Grass carp were used for the biological treatment because they have proven effective for controlling a variety of submerged species of aquatic vegetation including

Southern naiad and *Chara* (Masser 2002). Each treatment was replicated 3 times. Each pond was randomly assigned a treatment.

On December 13, 2005, fathead minnows (*Pimephales promelas*) and bluegill sunfish (*Lepomis macrochirus*) were stocked into each pond at recommended rates for unfertilized ponds. On March 23, 2006, largemouth bass (*Micropterus salmoides*) fingerlings were also stocked into each pond at recommended rates for unfertilized ponds. The Texas AgriLife Extension recommended rates of 20 lbs of fathead minnows, 500 bluegill, and 50 largemouth bass per surface acre (Texas Agricultural Extension Service 1996). Stocking occurred in early spring in order to allow fathead minnows and bluegill time to develop and spawn before the experiment began in May 2006. The percentage of pond surface covered by vegetation had to be at least 40% prior to the start of treatments. Coverage greater than 40% has been shown to have negative impacts on growth and abundance of largemouth bass (Sammons and Maceina 2005). If percent coverage was less than 40%, additional vegetation was transplanted. Table 1 shows the percent coverage of macrophytes for each pond.

Prior to treatment, each pond was sampled to evaluate fish, plankton, and invertebrate communities, as well as water quality. Fish were collected with an electroshocker mounted on an aluminum boat. The electroshocking was conducted for a period of ten minutes in each pond and all fish shocked within that time frame were collected. Fish abundance and individual length and weight were recorded. Phytoplankton were collected using a plankton tow net (20μm mesh size) near the water surface and preserved in glutaraldehyde. Glutaraldehyde was used in order to preserve

the pigments of the phytoplankton, facilitating identification. Zooplankton were collected using a Schindler-Patalas Trap (10L volume) at a depth of approximately 1 meter and preserved in formalin solution (Wetzel and Likens 2000). Phytoplankton and zooplankton were identified to lowest practical phylogenetic level using the inverted-microscope method (Hasle 1978). Taxonomic identification to lowest practical level gave insight into feeding strategy and ecological function of sampled plankton (Lenz 2000). Samples for chlorophyll *a* production were collected near the water surface from the shallow, middle, and deep ends of each pond and analyzed using a fluorometer. Benthic macroinvertebrates were sampled using a D-frame kick net along the pond bottom within a randomly determined single 1-m² square area and preserved in formalin solution (Rabeni 1996). Invertebrates were also identified to lowest practical phylogenetic level and abundance recorded. Figure 1 illustrates layout of the ponds and sampling sites.

Prior to treatment, vegetation status was determined by measuring percentage of pond surface area covered by vegetation. Visual estimation was conducted at the same location each time. Water quality parameters were evaluated as mean values across three subsamples (taken from the shallow, middle, and deep ends of each pond). Turbidity was measured using a Secchi-disk, dissolved oxygen concentrations and temperatures were measured using an YSI 85 probe, and a Hach sension1 meter was used to measure pH. A Hach DR/2010 spectrophotometer was used to measure the water quality parameters of hardness, total phosphorus, nitrate, nitrite, and ammonia-nitrogen.

Table 1. Percent coverage of macrophytes in each pond. Ponds A4, A6, and B8 are the herbicide treatments. Ponds A5, B6, and B7 are the grass carp treatments. Ponds A7, A8, and B5 are the controls.

	Ponds								
	A4	A5	A6	A7	A8	B5	B6	B7	B8
5/30/2006	70	98	65	100	65	40	45	80	45
6/1/2006	70	98	65	100	65	40	45	80	45
6/6/2006	65	100	65	100	70	40	40	70	45
6/13/2006	10	80	40	100	70	50	15	50	35
6/27/2006	0	75	5	100	75	55	5	20	0
7/28/2006	15	55	25	98	75	65	0	0	10
9/2/2006	40	50	30	100	90	70	0	0	20
10/1/2006	0	55	5	65	95	60	0	0	0
10/31/2006	5	55	0	50	95	65	0	0	10
12/2/2006	10	50	5	65	100	55	0	0	15
12/31/2006	5	50	10	70	100	55	0	0	20
2/3/2007	15	55	25	70	100	55	0	0	0
3/4/2007	25	50	35	60	100	45	0	0	15
4/6/2007	10	55	10	45	95	45	0	0	0
5/4/2007	0	55	5	50	98	30	0	0	0
6/1/2007	25	60	0	65	100	15	0	0	15
7/3/2007	40	75	0	75	100	5	0	0	35
8/1/2007	85	95	0	80	95	10	0	0	50
9/1/2007	20	100	0	75	65	10	0	0	0
10/5/2007	20	100	0	75	75	15	0	0	0
10/28/2007	25	95	0	70	95	20	0	0	0
11/18/2007	25	90	0	65	100	20	0	0	0

Treatments with Reward and Cutrine Plus began on May 31, 2006, and were conducted using a backpack sprayer from along the banks of each pond. Reward was applied at the rate of 1.5 gal/acre, and Cutrine was applied at a rate of 3.6 gal/acre in accordance with the label. Each application of chemical was broken down into two treatments. Half of the pond was initially treated, followed by treatment of the other half

of the pond one week later. This was done so that oxygen depletion would not occur as plant matter decomposed. If vegetation reestablished in the ponds and reached 20% coverage, additional applications were conducted. Table 2 contains application dates and rates. On May 31, 2006, three grass carp were stocked into each pond in the biological control treatment. On July 13, 2006 additional grass carp were stocked into each pond, which raised the number of carp to six per pond bringing the stocking rate to 60 per acre. Each pond received grass carp of similar weight. All grass carp were weighed prior to stocking. During both stocking dates, grass carp were acclimated to the pond water before being stocked into each pond.

Post-treatment sampling was conducted in the herbicide and control treatment ponds at day 2, day 7, day 14, day 28, and then monthly for the 18 months of the study. Post-treatment sampling in the grass carp treatment ponds was conducted at day 14, day 28, and monthly for the duration of the study. Post-treatment sampling times differed among the treatments because the reduction of macrophyte biomass from herbicide application is known to occur more rapidly than the reduction resulting from the consumption by grass carp. Therefore, it was necessary that sampling occurred at shorter time intervals initially for herbicide treatment ponds than for grass carp treatment ponds. At the end of the experiment all ponds were drained, and all fish and plant biomass were analyzed.

Within aquatic ecosystems, there are numerous direct and indirect impacts between biotic and abiotic factors. In order to determine if the treatments had any significant impacts on the suite of biotic and abiotic variables and to determine if any

significant relationships existed between the variables, multivariate canonical correspondence analyses were performed. Redundancy Analyses were conducted in a BACI design by performing Monte Carlo simulations on the water quality, macroinvertebrate, macrophyte, chlorophyll a, and fish assemblage data (Leps and Smilauer 2003; Palmer 1993; and Ter Braak and Verdonschot 1995). Principle Component Analysis was conducted on all of the data in order to try and explain the variance in the data. CANOCO software was used to perform these statistical analyses (Ter Braak and Smilauer 2002). To evaluate the significance of treatment effects on individual parameters, SPSS 15 software was used to conduct one-way ANOVA analyses for water quality, macroinvertebrates, macrophytes, fish, and chlorophyll a in order to determine if the means of various parameters were equal between the three treatments. If the means were different, post hoc tests using multiple comparisons with a Bonferroni adjustment to determine if the differences were significant.

Table 2. Application schedule and treatment rates for herbicide treatment ponds.

Date	Ponds	Chemical	Application Rate
5/31/2006	A4/A6/B8	Reward	0.75 gal./surface acre
6/18/2006	A4/A6/B8	Reward	1.5 gal./surface acre
6/24/2006	A4/A6/B8	Cutrine Plus	3.6 gal./surface acre
7/27/2006	A4/A6/B8	Reward	1.5 gal./surface acre
8/5/2006	A4/A6/B8	Cutrine Plus	3.6 gal./surface acre
8/13/2006	A4/A6/B8	Reward	1.5 gal./surface acre
12/10/2006	A4/A6/B8	Reward	1.5 gal./surface acre
12/17/2006	A4/A6/B8	Cutrine Plus	3.6 gal./surface acre
3/4/2007	A4/A6/B8	Reward	1.5 gal./surface acre
3/20/2007	A4/A6/B8	Cutrine Plus	3.6 gal./surface acre
8/1/2007	A4/B8	Reward	1.5 gal./surface acre
8/15/2007	A4/B8	Reward	1.5 gal./surface acre
8/23/2007	A4/B8	Cutrine Plus	3.6 gal./surface acre

3. RESULTS

The majority of the parameter means were not different between the three treatments. Only the parameters with different means between the three treatments are discussed in this section. Appendix A contains all of the temporal data collected during the study. Appendix B contains the phytoplankton and zooplankton data, which will be analyzed in the secondary study.

3.1 Water Quality Parameters

Turbidity measured as secchi depth was the only water quality parameter to have significantly different mean values between treatments (Figure 1). The ANOVA test resulted in an F value of 11.811 with degrees of freedom of 2 and 177. The Bonferroni adjustment resulted in mean difference values that were significant at the 0.05 level when the herbicide treatment was compared with the control as well as when the grass carp treatment was compared with the control. There was no significant difference between the herbicide and grass carp treatments (Table 3).

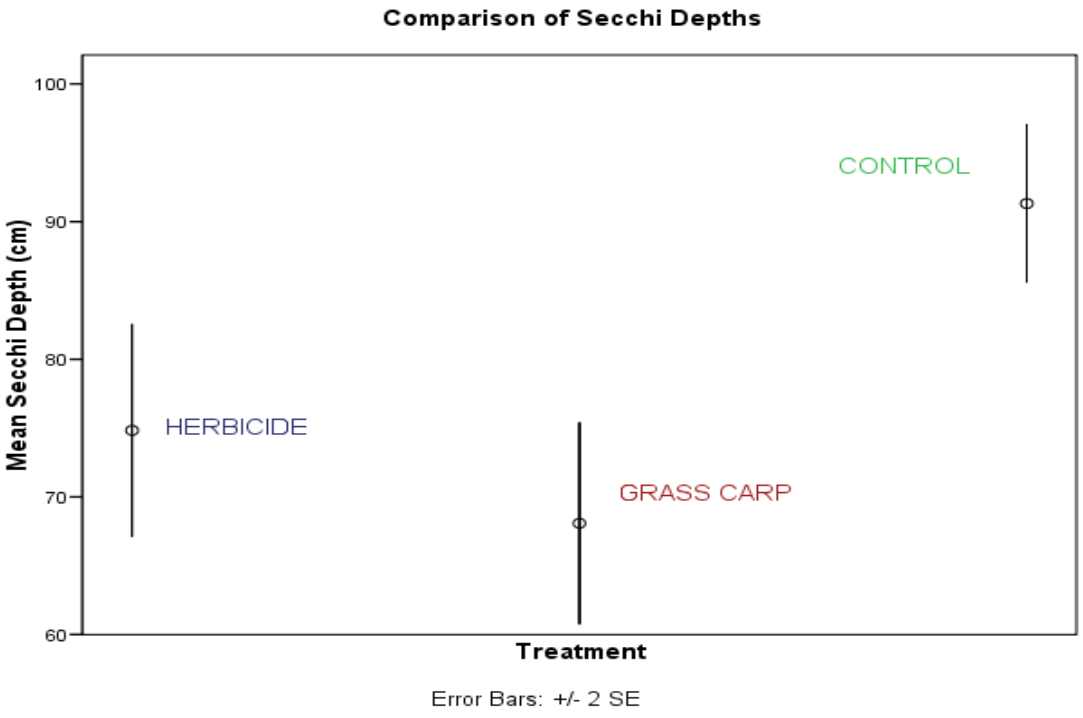

Figure 1. Comparison between treatments of mean secchi depths.

Table 3. Results of ANOVA and multiple comparison tests for secchi depth.

Secchi

	Sum of Squares	df	Mean Square	F	Sig.
Between Groups	17141.01111	2	8570.505556	11.81117961	1.53131E-05
Within Groups	128435.9	177	725.6265537		
Total	145576.9111	179			

Dependent Variable: Secchi
Bonferroni

(I) Treatment	(J) Treatment	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
					Upper Bound	Lower Bound
Herbicide	Grass Carp	6.750	4.918	.515	-5.14	18.64
	Control	-16.483(*)	4.918	.003	-28.37	-4.60
Grass Carp	Herbicide	-6.750	4.918	.515	-18.64	5.14
	Control	-23.233(*)	4.918	.000	-35.12	-11.35
Control	Herbicide	16.483(*)	4.918	.003	4.60	28.37
	Grass Carp	23.233(*)	4.918	.000	11.35	35.12

* The mean difference is significant at the .05 level.

The ANOVA for ammonia yielded an F value of 2.525 with degrees of freedom of 2 and 177 (Figure 2). However, the Bonferroni adjustment indicated that the differences between the treatment means were not significant at the 0.05 level (Table 4). Similarly, the ANOVA for nitrite yielded an F value of 0.883 with degrees of freedom of 2 and 177 (Figure 3). The Bonferroni adjustment indicated that there were no significant differences between the means at the 0.05 level (Table 5). There were no other water quality parameters that had F values higher than the critical values for F distributions.

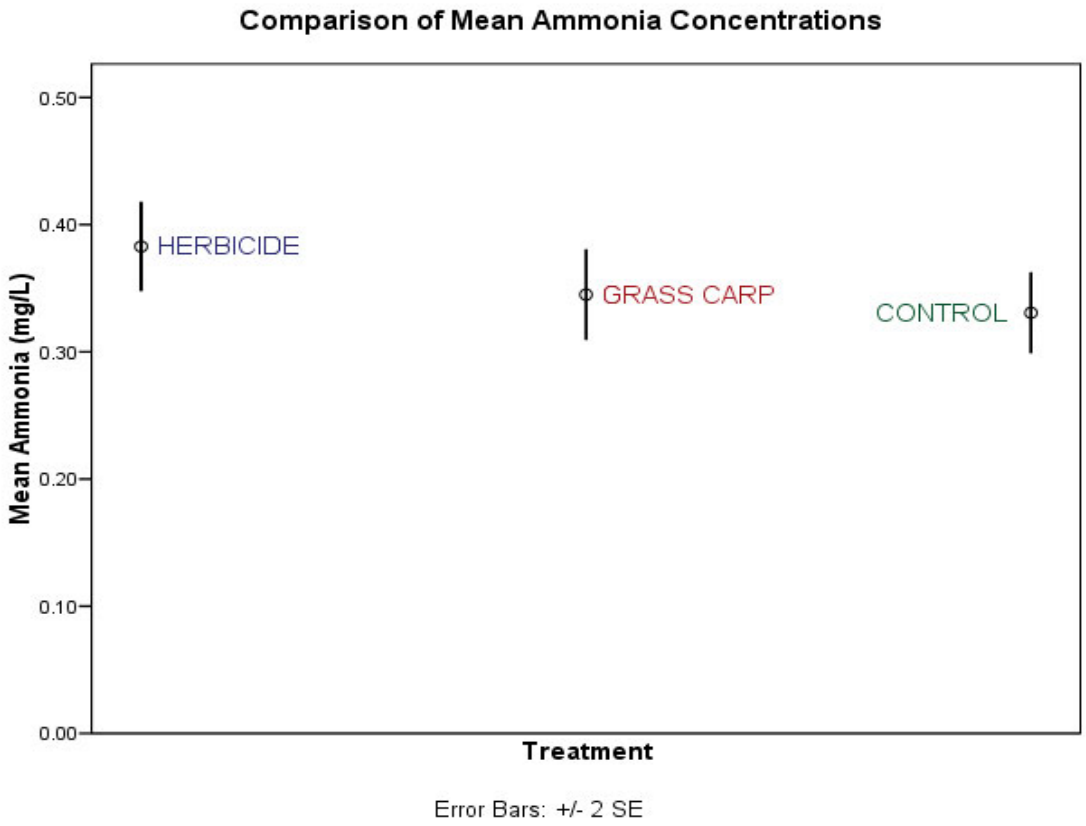

Figure 2. Comparison between treatments of mean ammonia concentrations.

Table 4. Results of ANOVA and multiple comparison tests for ammonia.

Ammonia

	Sum of Squares	df	Mean Square	F	Sig.
Between Groups	.087	2	.044	2.525	.083
Within Groups	3.055	177	.017		
Total	3.142	179			

Dependent Variable: Ammonia
Bonferroni

(I) Treatment	(J) Treatment	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
					Upper Bound	Lower Bound
Herbicide	Grass Carp	.03783	.02399	.350	-.0201	.0958
	Control	.05217	.02399	.093	-.0058	.1101
Grass Carp	Herbicide	-.03783	.02399	.350	-.0958	.0201
	Control	.01433	.02399	1.000	-.0436	.0723
Control	Herbicide	-.05217	.02399	.093	-.1101	.0058
	Grass Carp	-.01433	.02399	1.000	-.0723	.0436

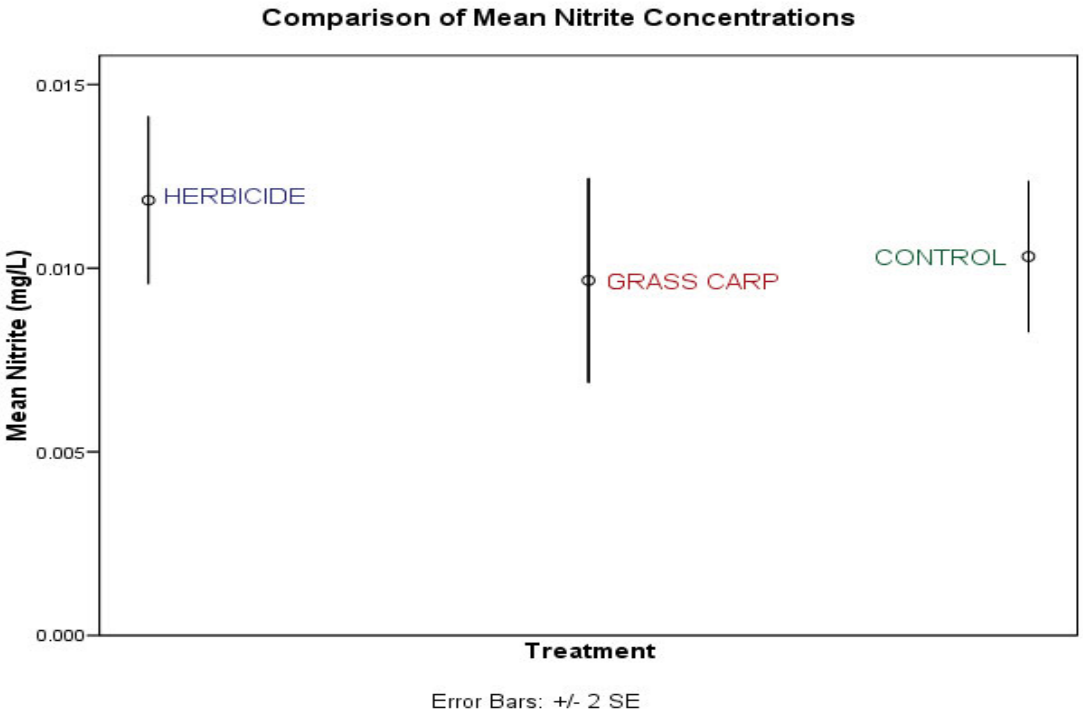

Figure 3. Comparison between treatments of mean nitrite concentrations.

Table 5. Results of ANOVA and multiple comparison tests for nitrite.

Nitrite

	Sum of Squares	df	Mean Square	F	Sig.
Between Groups	.000	2	.000	.883	.415
Within Groups	.015	177	.000		
Total	.015	179			

Dependent Variable: Nitrite
Bonferroni

(I) Treatment	(J) Treatment	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
					Upper Bound	Lower Bound
Herbicide	Grass Carp	.002183	.001687	.592	-.00189	.00626
	Control	.001533	.001687	1.000	-.00254	.00561
Grass Carp	Herbicide	-.002183	.001687	.592	-.00626	.00189
	Control	-.000650	.001687	1.000	-.00473	.00343
Control	Herbicide	-.001533	.001687	1.000	-.00561	.00254
	Grass Carp	.000650	.001687	1.000	-.00343	.00473

3.2 Chlorophyll a

The ANOVA test was performed using the chlorophyll a data and yielded an F value of 1.071 with degrees of freedom of 2 and 177 (Figure 4). The Bonferroni adjustment resulted in mean difference values that were not significant at the 0.05 level (Table 6).

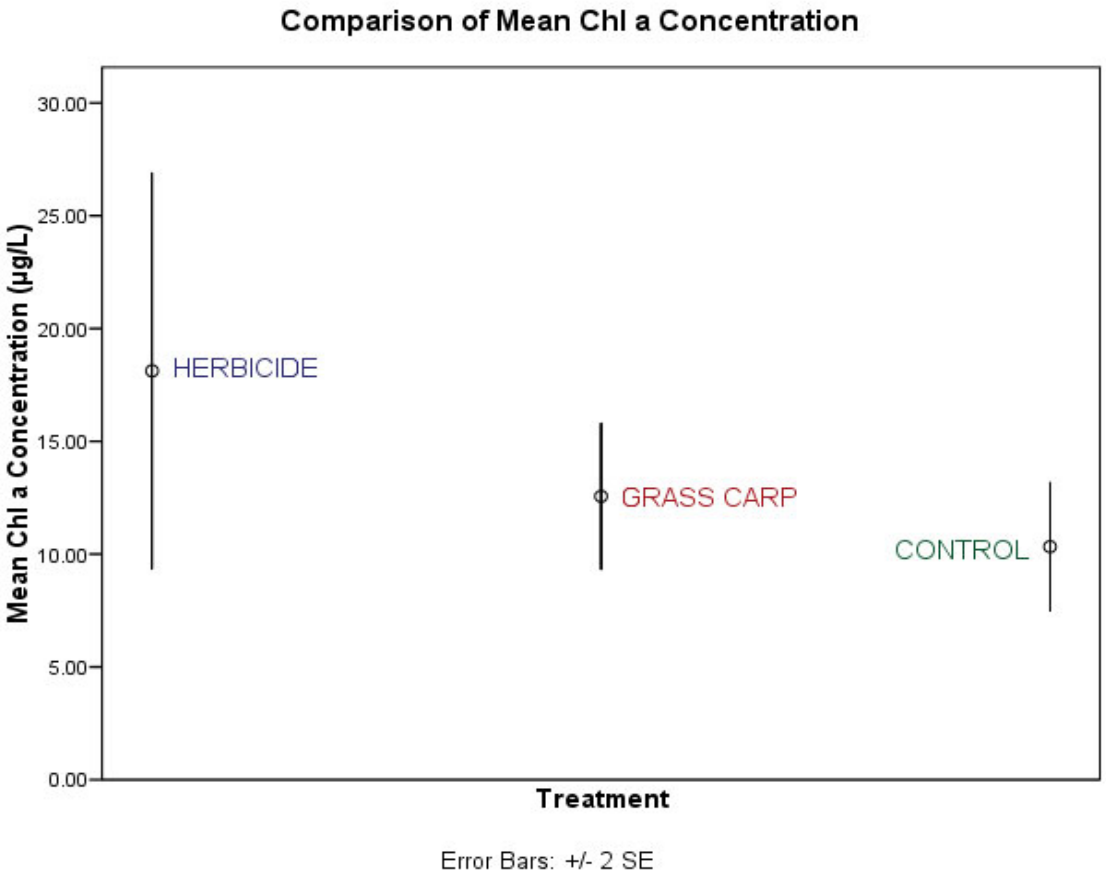

Figure 4. Comparison between treatments of mean chlorophyll a concentrations.

Table 6. Results of ANOVA and multiple comparison tests for chlorophyll *a*.

Chla

	Sum of Squares	df	Mean Square	F	Sig.
Between Groups	1089.778	2	544.889	1.071	.345
Within Groups	90043.899	177	508.723		
Total	91133.677	179			

Dependent Variable: Chla
Bonferroni

(I) Treatment	(J) Treatment	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
					Upper Bound	Lower Bound
Herbicide	Grass Carp	5.43317	4.11794	.566	-4.5197	15.3860
	Control	4.97600	4.11794	.686	-4.9768	14.9288
Grass Carp	Herbicide	-5.43317	4.11794	.566	-15.3860	4.5197
	Control	-.45717	4.11794	1.000	-10.4100	9.4957
Control	Herbicide	-4.97600	4.11794	.686	-14.9288	4.9768
	Grass Carp	.45717	4.11794	1.000	-9.4957	10.4100

3.3 Percent Coverage Macrophytes

The ANOVA test was performed using the percent coverage of macrophytes data and resulted in an F value of 47.469 with degrees of freedom of 2 and 195 (Figure 5). The Bonferroni adjustment resulted in mean difference values that were significant at the 0.05 level when the herbicide treatment was compared to the control as well as when the grass carp treatment was compared to the control. There was no significant difference between the herbicide treatment and the grass carp treatment means (Table 7).

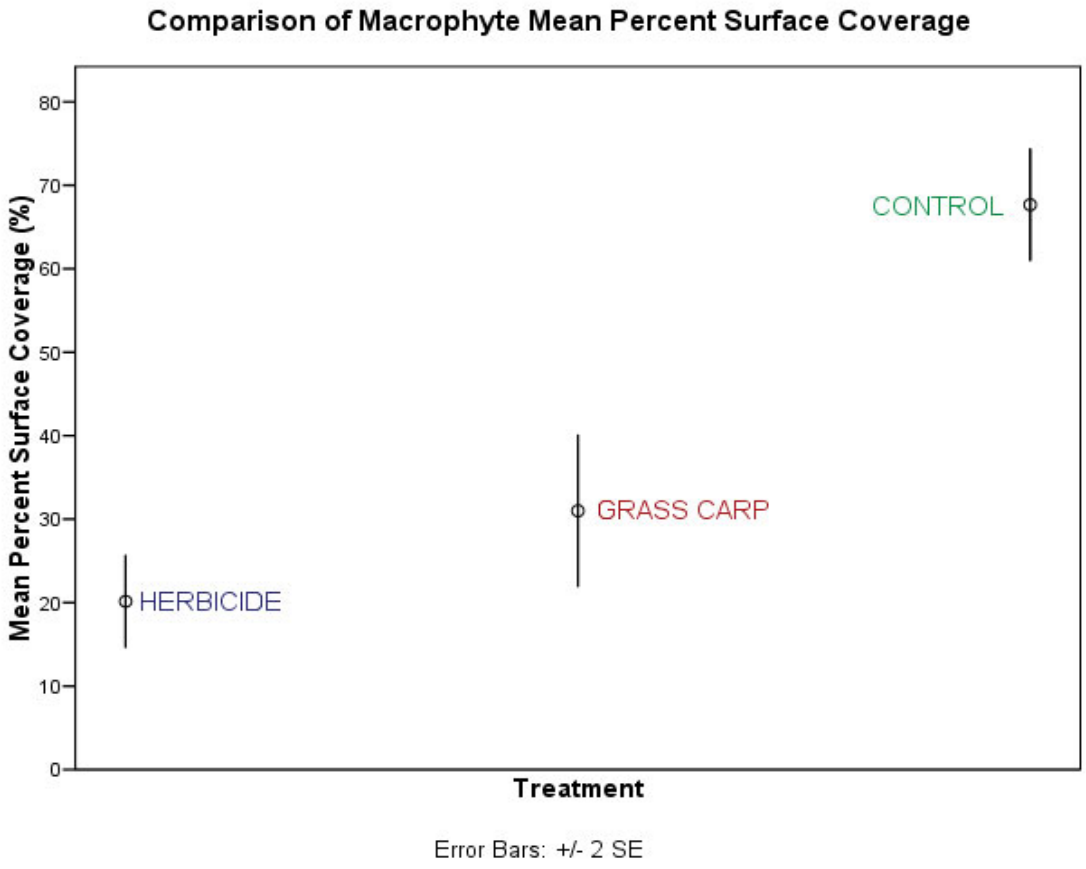

Figure 5. Comparison between treatments of mean percent coverage of macrophytes.

Table 7. Results of ANOVA and multiple comparison tests for percent coverage of macrophytes.

PercentCov

	Sum of Squares	df	Mean Square	F	Sig.
Between Groups	81836.121	2	40918.061	47.469	.000
Within Groups	168089.152	195	861.996		
Total	249925.273	197			

Dependent Variable: PercentCov
Bonferroni

(I) Treatment	(J) Treatment	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
					Upper Bound	Lower Bound
Herbicide	Grass Carp	-10.848	5.111	.105	-23.19	1.49
	Control	-47.515(*)	5.111	.000	-59.86	-35.17
Grass Carp	Herbicide	10.848	5.111	.105	-1.49	23.19
	Control	-36.667(*)	5.111	.000	-49.01	-24.32
Control	Herbicide	47.515(*)	5.111	.000	35.17	59.86
	Grass Carp	36.667(*)	5.111	.000	24.32	49.01

* The mean difference is significant at the .05 level.

3.4 Macroinvertebrates

Mean species richness of macroinvertebrates was analyzed using the one-way ANOVA test (Figure 6). The test resulted in an F value of 4.314 with degrees of freedom of 2 and 177. The Bonferroni adjustment resulted in mean difference values that were significant at the 0.05 level when the herbicide treatment was compared to the grass carp treatment as well as when the grass carp treatment was compared to the

control. There was no significant difference between the herbicide treatment and the control (Table 8).

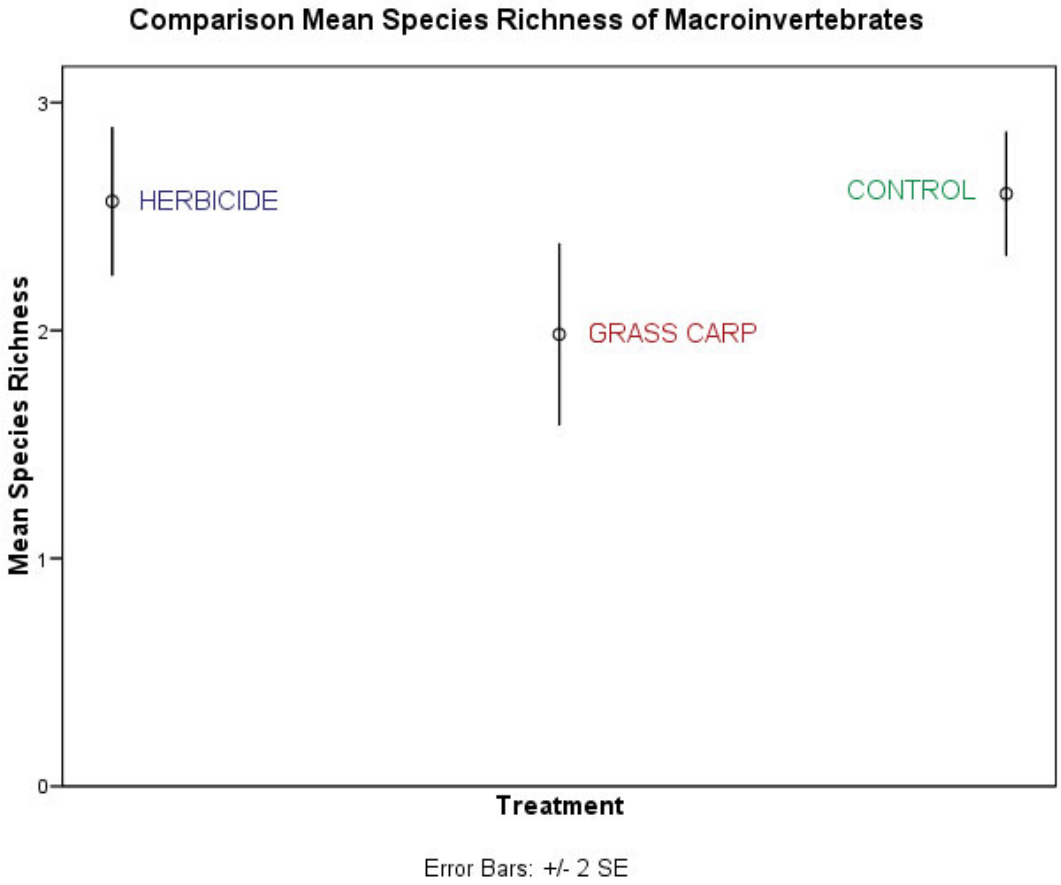

Figure 6. Comparison between treatments of mean species richness of macroinvertebrates.

Table 8. Results of ANOVA and multiple comparison tests for mean species richness of macroinvertebrates.

SpeciesRich

	Sum of Squares	df	Mean Square	F	Sig.
Between Groups	14.433	2	7.217	4.314	.015
Within Groups	296.117	177	1.673		
Total	310.550	179			

Dependent Variable: SpeciesRich
Bonferroni

(I) Treatment	(J) Treatment	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
					Upper Bound	Lower Bound
Herbicide	Grass Carp	.583(*)	.236	.043	.01	1.15
	Control	-.033	.236	1.000	-.60	.54
Grass Carp	Herbicide	-.583(*)	.236	.043	-1.15	-.01
	Control	-.617(*)	.236	.029	-1.19	-.05
Control	Herbicide	.033	.236	1.000	-.54	.60
	Grass Carp	.617(*)	.236	.029	.05	1.19

* The mean difference is significant at the .05 level.

3.5 Fish Parameters

The one-way ANOVA test was used to analyze largemouth bass mean weight between the treatments. The result was an F value of 3.743 with degrees of freedom of 2 and 51 (Figure 7). The Bonferroni adjustment indicated that there was a significant difference at the 0.05 level when the herbicide treatment was compared to the grass carp treatment (Table 9). No significant differences were found when the grass carp treatment was compared to the control or when the herbicide treatment was compared to the control. Largemouth bass mean length was also analyzed with ANOVA (Figure 8). The result was an F value of 3.842 with degrees of freedom of 2 and 51. The Bonferroni adjustment indicated that there was a significant difference at the 0.05 level when the herbicide treatment was compared to the grass carp treatment. No significant differences were found when the grass carp treatment was compared to the control or when the herbicide treatment was compared to the control (Table 10).

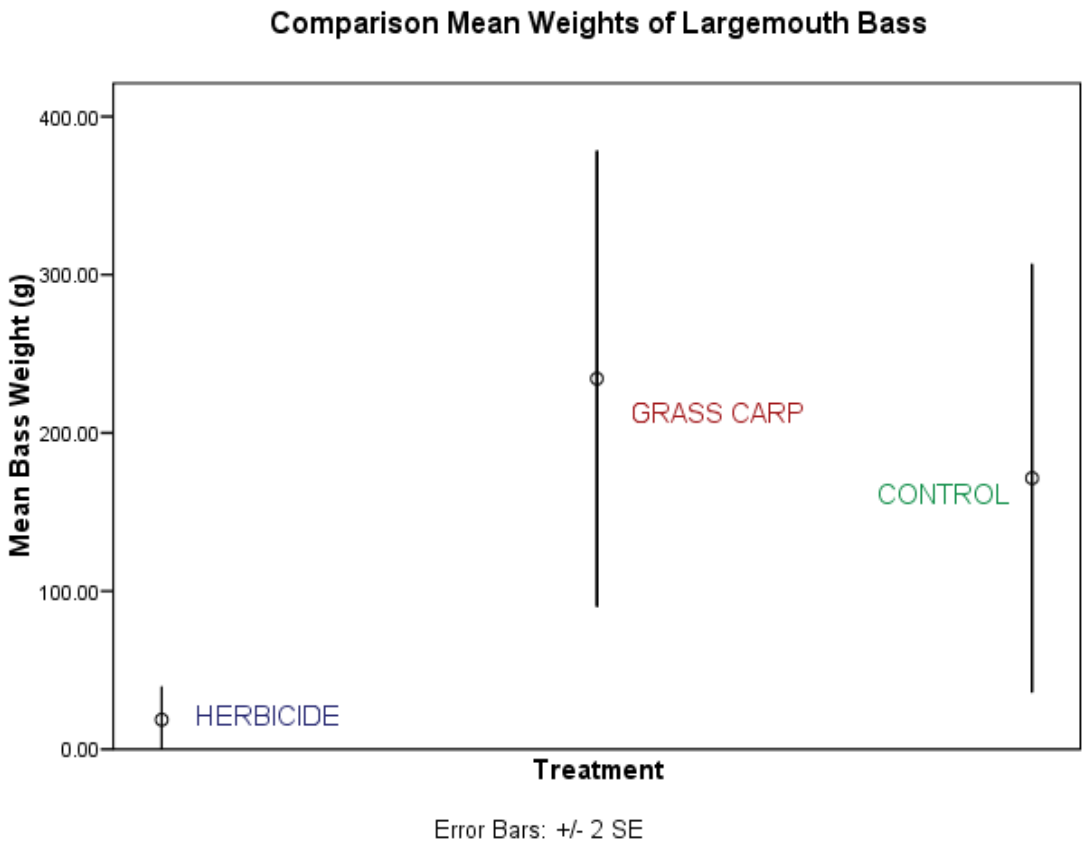

Figure 7. Comparison between treatments of largemouth bass mean weight.

Table 9. Results of ANOVA and multiple comparison tests for largemouth bass mean weight.

WeightLM

	Sum of Squares	df	Mean Square	F	Sig.
Between Groups	442471.940	2	221235.970	3.743	.030
Within Groups	3014562.852	51	59109.076		
Total	3457034.792	53			

Dependent Variable: WeightLM
Bonferroni

(I) Treatment	(J) Treatment	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
					Upper Bound	Lower Bound
Herbicide	Grass Carp	-215.58556(*)	81.04119	.031	-416.2042	-14.9669
	Control	-152.67944	81.04119	.196	-353.2981	47.9392
Grass Carp	Herbicide	215.58556(*)	81.04119	.031	14.9669	416.2042
	Control	62.90611	81.04119	1.000	-137.7125	263.5247
Control	Herbicide	152.67944	81.04119	.196	-47.9392	353.2981
	Grass Carp	-62.90611	81.04119	1.000	-263.5247	137.7125

* The mean difference is significant at the .05 level.

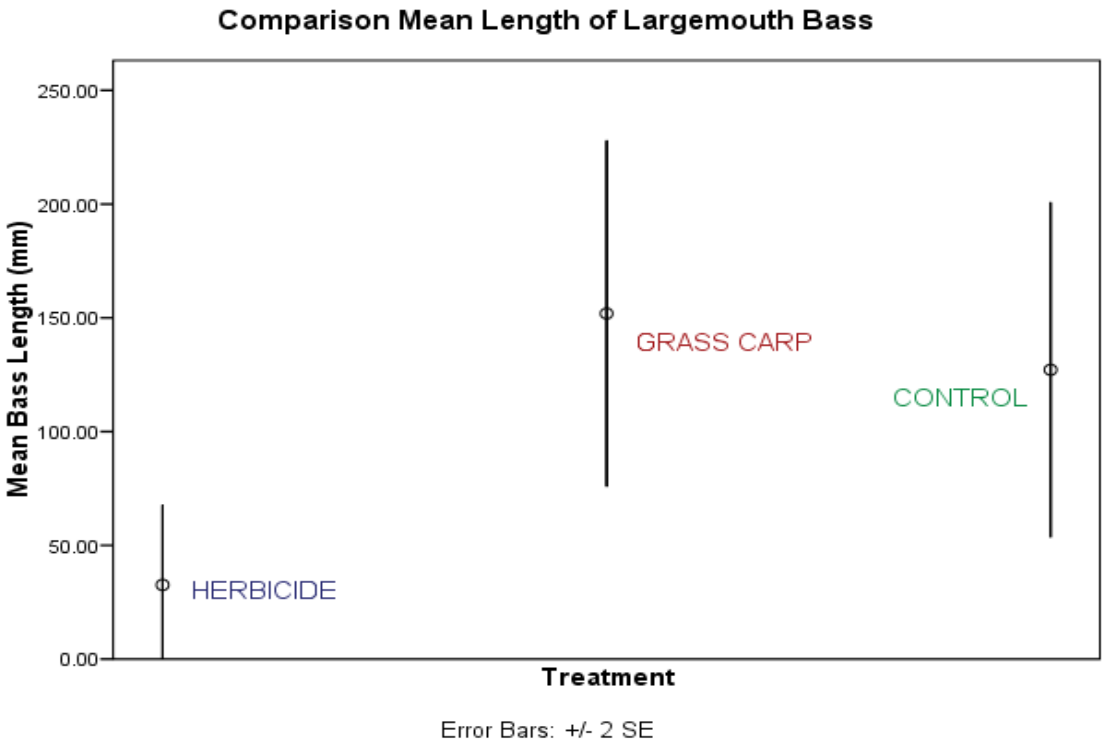

Figure 8. Comparison between treatments of largemouth bass mean length.

Table 10. Results of ANOVA and multiple comparison tests for largemouth bass mean length.

LengthLM

	Sum of Squares	df	Mean Square	F	Sig.
Between Groups	142742.106	2	71371.053	3.842	.028
Within Groups	947503.232	51	18578.495		
Total	1090245.338	53			

Dependent Variable: LengthLM
Bonferroni

(I) Treatment	(J) Treatment	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
					Upper Bound	Lower Bound
Herbicide	Grass Carp	-119.31000(*)	45.43432	.034	-231.7833	-6.8367
	Control	-94.56944	45.43432	.127	-207.0427	17.9039
Grass Carp	Herbicide	119.31000(*)	45.43432	.034	6.8367	231.7833
	Control	24.74056	45.43432	1.000	-87.7327	137.2139
Control	Herbicide	94.56944	45.43432	.127	-17.9039	207.0427
	Grass Carp	-24.74056	45.43432	1.000	-137.2139	87.7327

* The mean difference is significant at the .05 level.

The one-way ANOVA test was used to analyze juvenile largemouth bass mean weight between the treatments. The result was an F value of 1.000 with degrees of freedom of 2 and 51 (Figure 9). The Bonferroni adjustment indicated that there were no significant differences between the treatment means (Table 11). Total juvenile bass weight was also analyzed using ANOVA. The result was an F value of 1.000with degrees of freedom of 2 and 51 (Figure 10). The Bonferroni adjustment indicated that there were no significant differences between the treatment means (Table 12).

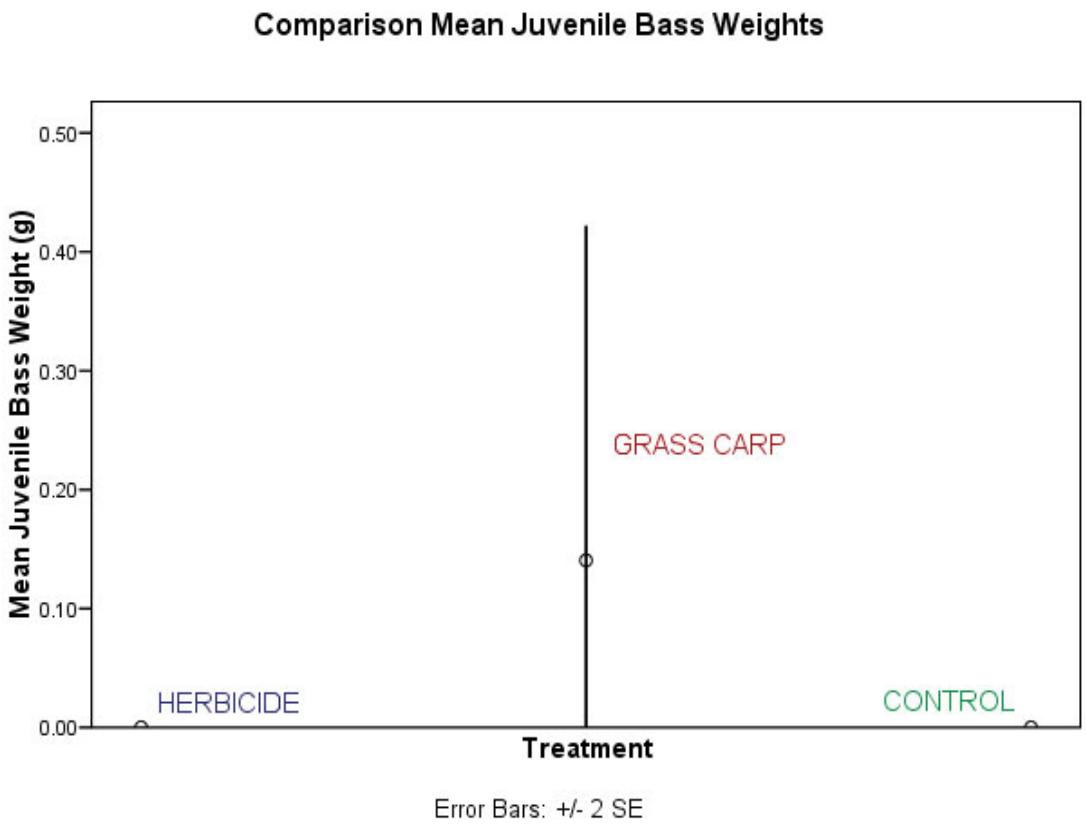

Figure 9. Comparison between treatments of juvenile largemouth bass mean weight.

Table 11. Results of ANOVA and multiple comparison tests for juvenile largemouth bass mean weight.

JuvBassWt

	Sum of Squares	df	Mean Square	F	Sig.
Between Groups	.237	2	.119	1.000	.375
Within Groups	6.045	51	.119		
Total	6.282	53			

Dependent Variable: JuvBassWt
Bonferroni

(I) Treatment	(J) Treatment	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
					Upper Bound	Lower Bound
Herbicide	Grass Carp	-.14056	.11476	.679	-.4247	.1435
	Control	.00000	.11476	1.000	-.2841	.2841
Grass Carp	Herbicide	.14056	.11476	.679	-.1435	.4247
	Control	.14056	.11476	.679	-.1435	.4247
Control	Herbicide	.00000	.11476	1.000	-.2841	.2841
	Grass Carp	-.14056	.11476	.679	-.4247	.1435

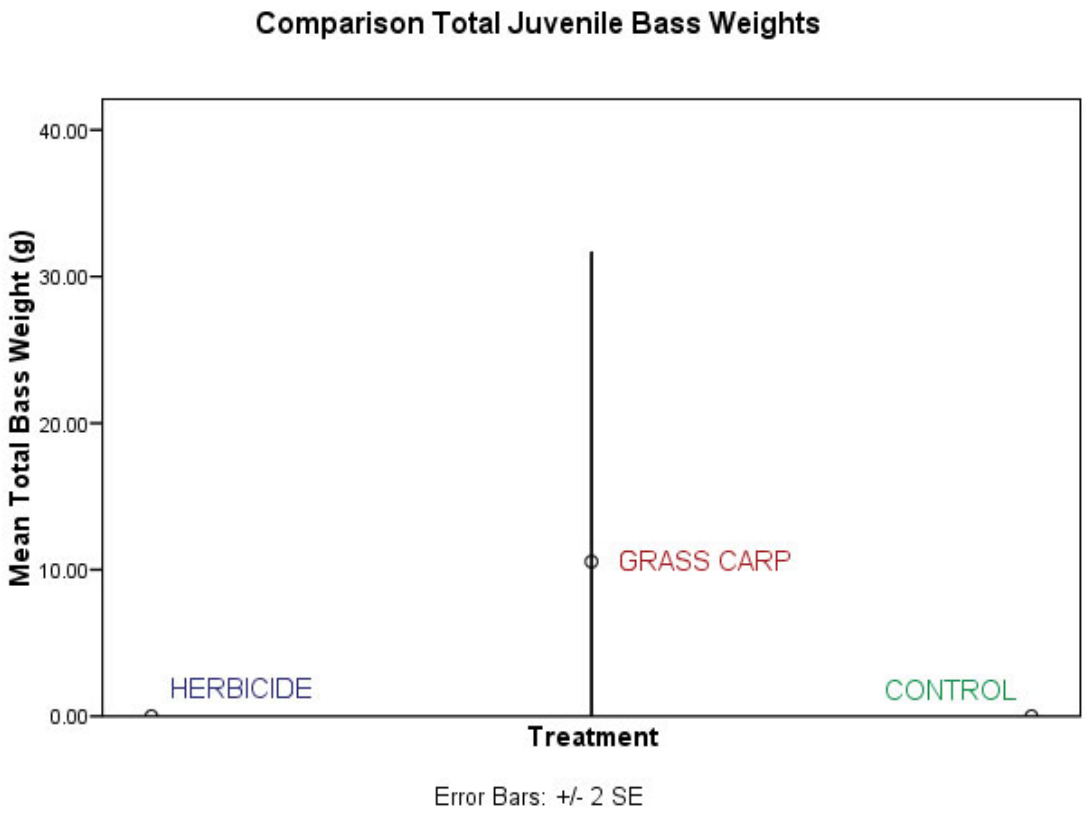

Figure 10. Comparison between treatments of mean juvenile largemouth bass total weight.

Table 12. Results of ANOVA and multiple comparison tests for mean juvenile bass total weight.

TotalBassWt

	Sum of Squares	df	Mean Square	F	Sig.
Between Groups	1337.037	2	668.519	1.000	.375
Within Groups	34094.444	51	668.519		
Total	35431.481	53			

Dependent Variable: TotalBassWt
Bonferroni

(I) Treatment	(J) Treatment	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
					Upper Bound	Lower Bound
Herbicide	Grass Carp	-10.55556	8.61858	.679	-31.8910	10.7799
	Control	.00000	8.61858	1.000	-21.3354	21.3354
Grass Carp	Herbicide	10.55556	8.61858	.679	-10.7799	31.8910
	Control	10.55556	8.61858	.679	-10.7799	31.8910
Control	Herbicide	.00000	8.61858	1.000	-21.3354	21.3354
	Grass Carp	-10.55556	8.61858	.679	-31.8910	10.7799

The one-way ANOVA test was used to analyze bluegill mean weight between the treatments. The result was an F value of 0.911 with degrees of freedom of 2 and 51 (Figure 11). The Bonferroni adjustment indicated that there were no significant differences between the treatment means (Table 13). Bluegill mean length was also analyzed using ANOVA. The result was an F value of 0.434 with degrees of freedom of 2 and 51 (Figure 12). The Bonferroni adjustment indicated that there were no significant differences between the treatment means (Table 14).

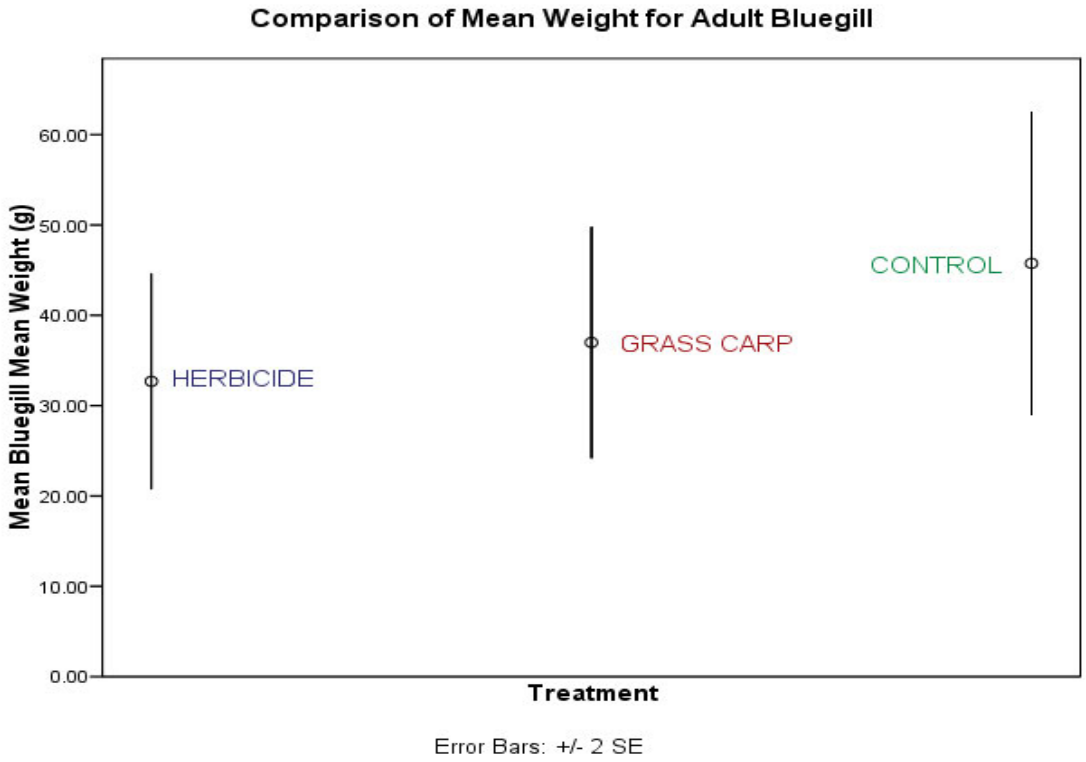

Figure 11. Comparison between treatments of bluegill mean weight.

Table 13. Results of ANOVA and multiple comparison tests for bluegill mean weight.

WeightBG

	Sum of Squares	df	Mean Square	F	Sig.
Between Groups	1594.631	2	797.316	.911	.408
Within Groups	44616.326	51	874.830		
Total	46210.957	53			

Dependent Variable: WeightBG
Bonferroni

(I) Treatment	(J) Treatment	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
					Upper Bound	Lower Bound
Herbicide	Grass Carp	-4.29944	9.85917	1.000	-28.7060	20.1071
	Control	-13.05944	9.85917	.574	-37.4660	11.3471
Grass Carp	Herbicide	4.29944	9.85917	1.000	-20.1071	28.7060
	Control	-8.76000	9.85917	1.000	-33.1665	15.6465
Control	Herbicide	13.05944	9.85917	.574	-11.3471	37.4660
	Grass Carp	8.76000	9.85917	1.000	-15.6465	33.1665

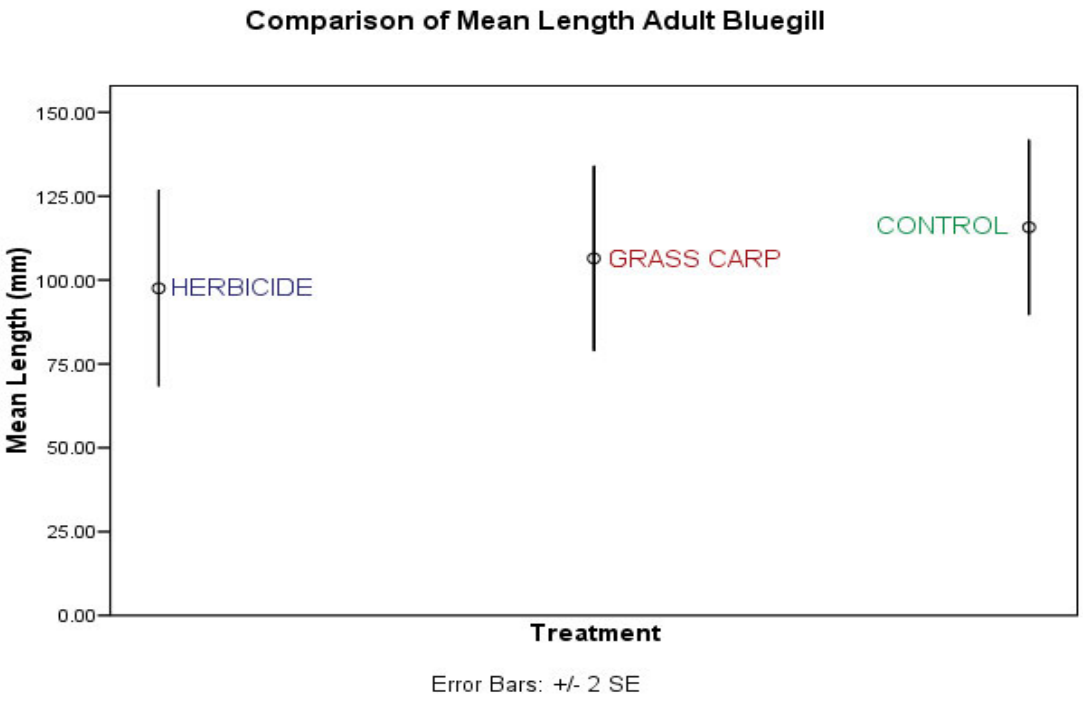

Figure 12. Comparison between treatments of bluegill mean length.

Table 14. Results of ANOVA and multiple comparison tests for bluegill mean length.

BGLength

	Sum of Squares	df	Mean Square	F	Sig.
Between Groups	2972.085	2	1486.043	.434	.650
Within Groups	174470.849	51	3420.997		
Total	177442.934	53			

Dependent Variable: BGLength
Bonferroni

(I) Treatment	(J) Treatment	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
					Upper Bound	Lower Bound
Herbicide	Grass Carp	-8.90722	19.49643	1.000	-57.1709	39.3565
	Control	-18.17111	19.49643	1.000	-66.4348	30.0926
Grass Carp	Herbicide	8.90722	19.49643	1.000	-39.3565	57.1709
	Control	-9.26389	19.49643	1.000	-57.5276	38.9998
Control	Herbicide	18.17111	19.49643	1.000	-30.0926	66.4348
	Grass Carp	9.26389	19.49643	1.000	-38.9998	57.5276

The final parameter analyzed using the one-way ANOVA was juvenile bluegill mean weight. The result was an F value of 3.221 with degrees of freedom of 2 and 6 (Figure 13). The Bonferroni adjustment indicated that there were no significant differences between the treatment means (Table 15).

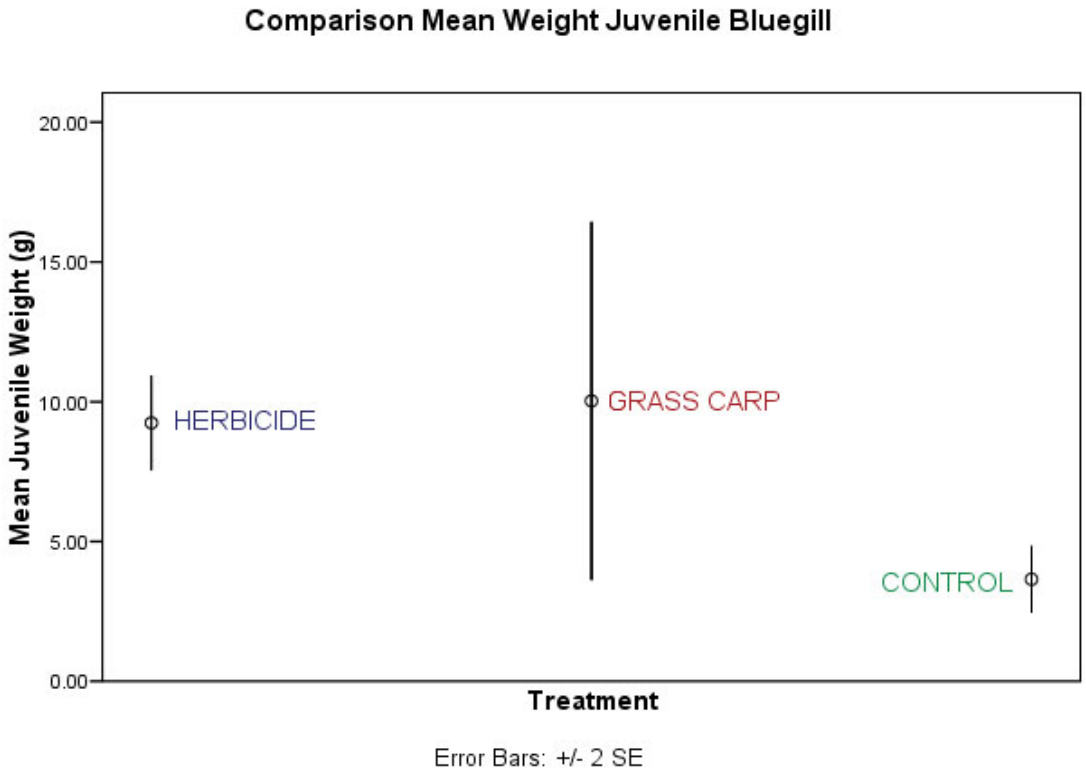

Figure 13. Comparison between treatments of juvenile bluegill mean weight.

Table 15. Results of ANOVA and multiple comparison tests for juvenile bluegill mean weight.

JuvenileWeight

	Sum of Squares	df	Mean Square	F	Sig.
Between Groups	72.609	2	36.304	3.221	.112
Within Groups	67.617	6	11.270		
Total	140.226	8			

Dependent Variable: JuvenileWeight
Bonferroni

(I) Treatment	(J) Treatment	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
					Upper Bound	Lower Bound
Herbicide	Grass Carp	-.78667	2.74099	1.000	-9.7976	8.2242
	Control	5.59333	2.74099	.262	-3.4176	14.6042
Grass Carp	Herbicide	.78667	2.74099	1.000	-8.2242	9.7976
	Control	6.38000	2.74099	.176	-2.6309	15.3909
Control	Herbicide	-5.59333	2.74099	.262	-14.6042	3.4176
	Grass Carp	-6.38000	2.74099	.176	-15.3909	2.6309

3.6 Multivariate Analysis

CANOCO version 4.5 was used to perform multivariate analysis of the data in several ways. The first was taking all the data and analyzing it at the same time with Redundancy Analysis. The result of the Monte Carlo simulations was an eigenvalue of 0.040, an F-ratio of 2.113, and a P-value of 0.2380 on the first canonical axis. The eigenvalue on all axes was 0.065, the F-ratio was 1.773, and the P-value was 0.0720. Therefore, there were no significant multivariate relationships between the data. Even though the relationships were not significant, generalizations could be made when the

results were graphed (Figure 14). Largemouth bass growth appeared to be most closely related to the grass carp treatment and least related to the herbicide treatment. Crustacean abundance and Chara were also most closely related to the grass carp treatment. Southern naiad, coleopteran abundance, adult bluegill abundance, total phosphorus, and ammonia were most closely related to the herbicide treatment. Percent coverage of macrophytes, coontail, low dissolved oxygen, and low turbidity, juvenile bluegill abundance, adult bluegill growth, and odonata abundance were most closely related to the control treatment. Chlorophyll *a* concentration was least related to the control treatment and more closely related to the herbicide and grass carp treatments. The fish data was then taken out of the analysis and the simulations were permutated again. The result was an eigenvalue of 0.052, an F-ratio of 2.776, and a P-value of 0.2240 on the first canonical axis. The eigenvalue on all axes was 0.078, the F-ratio was 2.158, and the P-value was 0.1400. There were still no significant relationships, although the same generalizations could be made graphically as in the first test (Figure 15). In the third analysis, only the fish data was analyzed to determine if there were significant relationships between the largemouth bass, bluegill, and treatments. The test for significance on the first canonical axis yielded an eigenvalue of 0.045, an F-ratio of 2.131, and a P-value of 0.0860. The eigenvalue on all axes was 0.053, the F-ratio was 1.265, and the P-value was 0.1000. There were no significant relationships. As in the previous two analyses, the same generalizations could be made from graphical analysis (Figure 16). The Principle Component Analysis resulted in 34.7 % of the total variance within the data being accounted for within two dimensional axes.

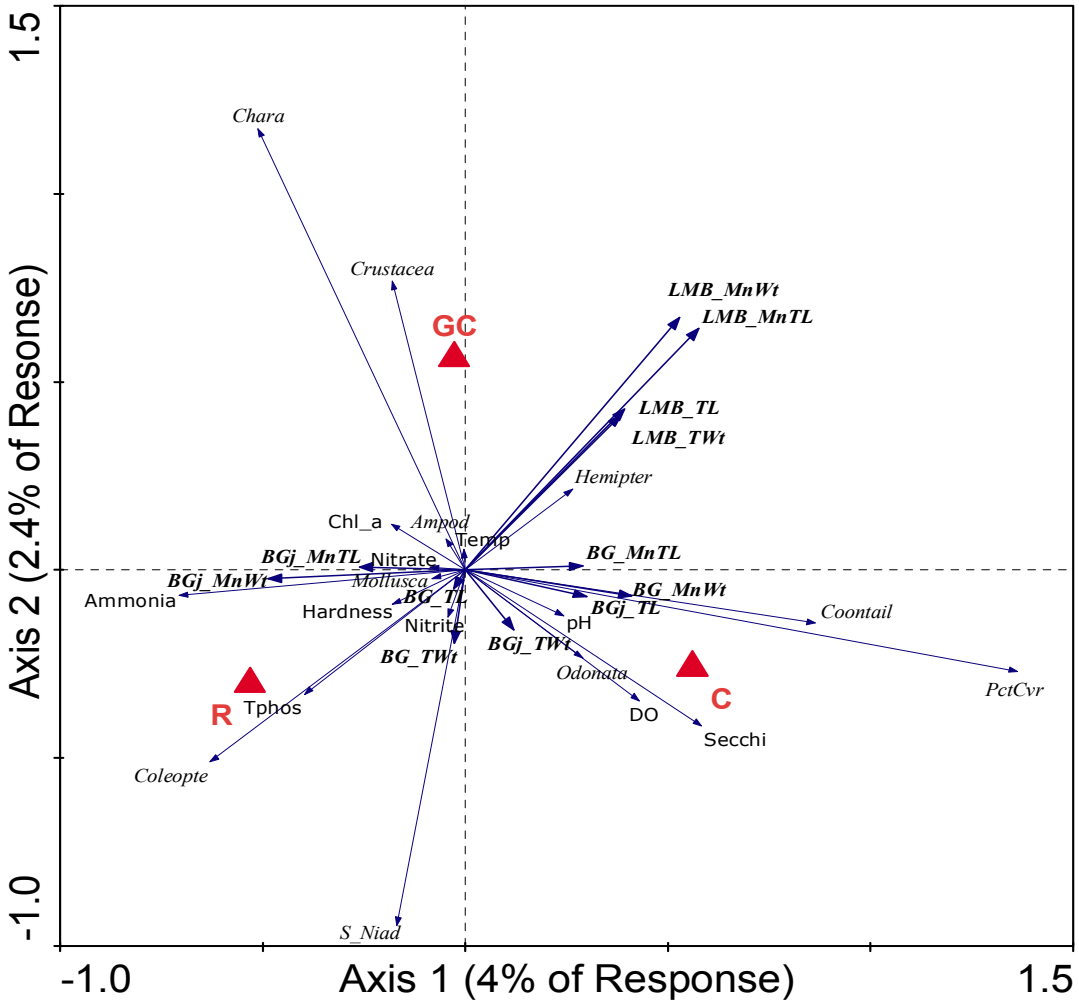

Figure 14. Multivariate analysis of all response variables using Monte Carlo simulations.

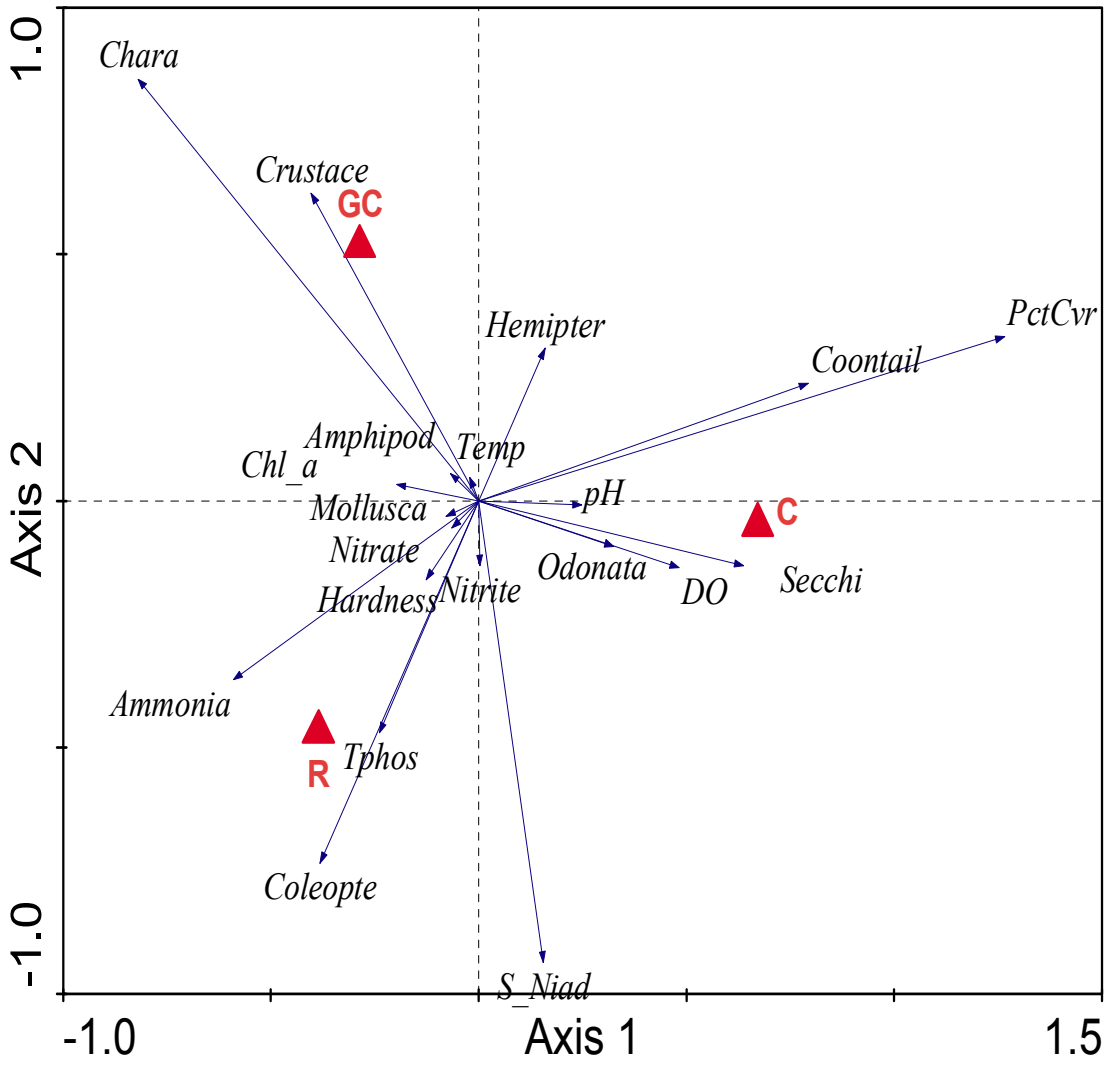

Figure 15. Multivariate analysis of macrophytes, macroinvertebrates, and water quality parameters using Monte Carlo simulations.

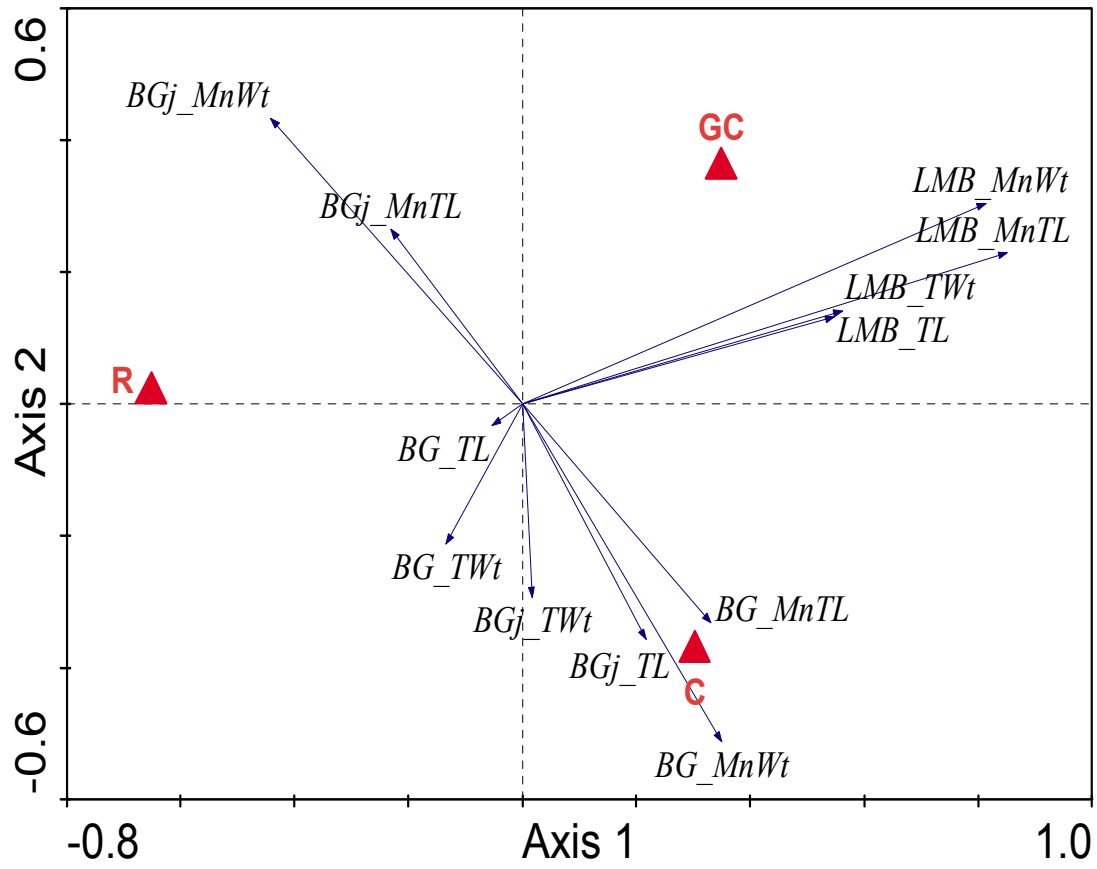

Figure 16. Multivariate analysis of largemouth bass and bluegill using Monte Carlo simulations.

4. DISCUSSION AND CONCLUSIONS

4.1 Discussion

Throughout the study, the most common macrophytes and macroalgae observed in the ponds were southern naiad, coontail, and chara. Occasionally, littoral plants such as primrose, *Ludwigia spp.*, and maidencane, *Panicum hemitomon*, established along the pond banks. Even though the ponds were sterilized prior to the study, coontail and chara were able to reestablish in some of the ponds, most likely due to seedbanks within the sediment. Although chara was common in the grass carp ponds during the beginning of the study, it was extirpated from those ponds within several months. Southern naiad was also quickly eliminated in the grass carp ponds. After southern naiad and chara were eliminated, coontail established itself in two of the grass carp ponds and was able to spread quickly in one of the ponds. Within several months, one grass carp pond was severely infested with coontail (>75% coverage). Even after additional grass carp were stocked to the three ponds in the treatment, the problem with coontail persisted in the one pond. The other two grass carp ponds remained devoid of aquatic macrophytes with occasional filamentous algal blooms throughout the study. The problem with coontail in the grass carp treatment may have been a result of coontail being one of the least preferred submerged macrophytes consumed by grass carp (Masser 2002). Grass carp were observed feeding on terrestrial plants along the bank even though coontail was abundant in the pond. Grass carp were ineffective at controlling coontail in this study. Therefore, it is important that pond owners and managers consider the macrophyte species present before deciding whether or not to stock grass carp.

In the herbicide treatment ponds, southern naiad was the most dominant macrophyte observed. After treatment of naiad with Reward, chara established itself in the ponds. Treatment with Cutrine Plus eliminated the chara. Typically, an algal bloom occurred after naiad and chara were eliminated and would persist for a period of two to four weeks. After several weeks, naiad would reestablish in the shallow areas and the cycle would repeat. Occasionally, coontail would grow in the herbicide ponds, but treatment with Reward eliminated any coontail present. Re-growth in the herbicides ponds consisted mainly of naiad and chara. Reward was effective at controlling southern naiad and coontail, and supplemental applications with Cutrine plus was effective at controlling chara. The only problem encountered was the fact that five subsequent treatments with herbicides were required due to reestablishment of macrophytes.

Typically, shallow aquatic ecosystems exist in one of two states, a vegetated state with clear water or a turbid state as the result of phytoplankton or suspended solids. The state of the system is typically dictated by nutrient concentration. At low nutrient concentrations, a system dominated by aquatic macrophytes exists. At high nutrient concentrations, a system dominated by phytoplankton exists. At intermediate nutrient concentrations, either state can occur. Unless an extreme change in nutrient concentration occurs, the system typically stays in the state it currently is in despite changes in external conditions. This is known as hysteresis (Scheffer 1998). A major disturbance, such as the application of herbicides, will change the system from a vegetated state to a turbid phytoplankton state. The system should stay in that state

unless another disturbance occurs. The ponds in this study reverted back to the vegetated state, which goes against the typical pattern observed. A possible explanation is that after the phytoplankton were able to establish a bloom, the macrophytes were able to recolonize the extreme shallow areas of the pond even with the turbid conditions and slowly out-compete the plankton for nutrients, which would allow the macrophytes to expand and cause a decline in the plankton population (Scheffer 1998).

The control ponds were dominated by southern naiad and coontail at the beginning of the study. Coontail became the dominant macrophyte in one of the control ponds. Southern naiad was the dominant macrophyte in one of the control ponds, and the third control pond experienced a bloom of cyanobacteria that eliminated the macrophytes in this pond. Small amounts of coontail and naiad were able to recolonize the shallow areas of the pond. Chara was present in small amounts in the control ponds but never became a dominant species. In the two ponds that did not have a cyanobacteria bloom, the percent coverage of macrophytes was between 70 and 100 percent for the majority of the study. This demonstrates that the absence of vegetation management in small impoundments typically leads to severe infestations of aquatic macrophytes.

Aquatic vegetation management with the use of either herbicides or grass carp both had significant impacts on the percent coverage of macrophytes in the small impoundments. The statistical analysis showed that both treatments resulted in a significantly lower percent coverage of macrophytes than the control treatment. The herbicide treatment had the lowest percent coverage of macrophytes, although it was not

significantly different from the grass carp treatment. It should be noted that if coontail had not rapidly established itself in the one grass carp pond, it is likely that all three grass carp ponds would have had zero percent coverage of macrophytes at the end of the study and would have been significantly lower than the herbicide treatment. The mean percent coverage in the herbicide treatment ponds was between 15-25 percent coverage, which is considered the desirable range of aquatic macrophytes for a healthy warmwater fishery (Bettoli et al. 1993; Brown and Maceina 2002; Engel 1995; Henderson 1996; Hoyer and Canfield 2001; Maceina and Slipke 2004; Miranda and Pugh 1997).

Mean chlorophyll *a* concentrations were highest in the herbicide treatment ponds. The grass carp ponds were slightly higher than the control ponds. However, there were no significant differences in mean chlorophyll *a* concentrations. There was no significant difference because the chlorophyll *a* concentrations in the herbicide ponds radically fluctuated due to crashes in phytoplankton populations. Klussmann et al. (1988) also found increased chlorophyll *a* concentrations in Lake Conroe after macrophyte removal.

A reason for this may have been that there was not enough phosphorus in the water column to sustain a phytoplankton bloom for an extended period of time. A surge in phosphorus as a result of the decaying macrophytes after herbicide application allowed phytoplankton populations to increase rapidly. After several weeks, the phytoplankton used up all the available phosphorus and the population declined. Similar trends were found by Carter and Hestand (1977b), in their study on the relationship between macrophytes and plankton after herbicide treatment. Macrophytes would then

reestablish because of their ability to utilize phosphorus sequestered in the sediment as well as out-compete plankton for light (Carter and Hestand 1977b).

In the grass carp ponds, phytoplankton populations were limited by the lack of light penetration as a result of increased turbidity. The increase in turbidity was the result of grass carp stirring up the sediment causing silt and clay particles to become suspended in the water column. Even though nutrients were available for the phytoplankton, they were limited by the lack of sunlight penetration. Even though turbidity was low in one of the grass carp ponds, a plankton bloom never occurred as a result of the severe coontail infestation in the pond. Chlorophyll *a* was lowest in the control ponds as would be expected due to the high percent coverage of macrophytes.

Turbidity was highest in the grass carp treatment, and both the herbicide and grass carp treatments had significantly lower secchi depths than the control treatment. As hypothesized, the high turbidity in the herbicide ponds was due to phytoplankton blooms, which occurred after the elimination of macrophytes, and the high turbidity in the grass carp ponds was the result of grass carp activity stirring up the sediment. An increase in turbidity as a result of grass carp activity has been found in several studies (Kirkagac and Demir 2004; Klussmann et al. 1988). If not for the coontail problem in the one grass carp pond, it is likely that the grass carp treatment would have had significantly lower turbidity than the herbicide treatment. The turbidity of the control treatment would also have been higher if there had not been a cyanobacteria bloom in one of the ponds. Unfortunately, the stochastic nature of ponds makes conducting truly controlled experiments difficult at times.

Even though ammonia and nitrite levels were higher in the herbicide treatment ponds than in the grass carp treatment and the control, there were no significant differences. The slightly higher levels of ammonia and nitrite may have been caused by the rapid decay of plant material after herbicide applications. There were no significant differences in any of the other water quality parameters measured. Although, the differences were not significant, there were several instances in the herbicide and control ponds where the dissolved oxygen concentration was below 2.00 mg/L, which can be lethal to fish (Smale and Rabeni 1995). In the herbicide ponds it was most likely due to the decomposition of plants after herbicide application. In the control ponds it was most likely caused by the overabundance of macrophytes and their elevated oxygen demand at night (Carter et al. 1991; Kilgore and Hoover 2001).

The most common macroinvertebrates collected during sampling were dragonfly and damselfly nymphs, *Odonata*, snails, *Mollusca*, amphipods, *amphipoda*, and predacious diving beetles, *Coleoptera*. Other macroinvertebrates collected were water boatmen and backswimmers, *Hemiptera*, grass shrimp, *Crustacea*, mosquito larvae, *Diptera*, and some trematodes, *Trematoda*. The herbicide treatment ponds were dominated mostly by dragonfly nymphs, damselfly nymphs, snails, and amphipods. The control ponds were dominated by species similar to the herbicide ponds. The grass carp ponds were dominated by snails and grass shrimp. It appeared that dragonfly nymphs, damselfly nymphs, snails, and amphipods were most abundant when aquatic macrophytes were present in intermediate to high amounts regardless of treatment. Predacious diving beetles, grass shrimp, water boatmen, and backswimmers were most

abundant when aquatic macrophytes were present in low amounts regardless of treatment, although it was not statistically significant. The abundance of grass shrimp in two of the grass carp ponds may have occurred because the lack of macrophytes caused the grass shrimp to seek refuge in littoral vegetation along the pond banks making them more susceptible to capture during sampling. Although the relationship was not significant, the multivariate analysis showed a possible interaction between the percent coverage of macrophytes and the number of *Odonata* present.

Statistical analysis indicated that the mean macroinvertebrate species richness was significantly lower in the grass carp treatment when compared to the control and the herbicide treatment. Previous studies found an increase in macroinvertebrate abundance and species richness after grass carp stocking (Cassani and Caton 1986; Kirkagac and Demir 2004). There was no significant difference between the herbicide treatment and the control. The lower species richness could be attributed to lower percent coverage of macrophytes. Most macroinvertebrates rely on macrophytes for shelter from predators such as bluegill and fathead minnows. The absence of macrophytes in two of the grass carp ponds left the macroinvertebrates vulnerable to predation resulting in fewer species with the ability to adapt to the conditions. Intermediate levels of macrophytes provided macroinvertebrates with sufficient shelter in the herbicide ponds. The use of herbicides appeared to have little impact on macroinvertebrate species richness when compared to the control. Similar results have been found in other studies (Cassani and Caton 1985; Harman et al. 2005). The grass carp pond infested with coontail had similar macroinvertebrate counts as those of the herbicide and control ponds.

It was difficult to analyze the largemouth bass data collected due to their avoidance of the electroshocking gear during sampling and the low survival in the majority of the ponds. Five ponds had zero percent survival. One grass carp pond and one control pond were near 50 percent survival. No bass were harvested from the herbicide ponds, one of the grass carp ponds, or one of the control ponds. The grass carp pond with zero percent survival was the pond with the coontail infestation. There are several factors that may have caused the low survival of bass in the ponds. During the beginning of the study we had fish kills in two of the herbicide ponds. One of the fish kills was most likely related to low dissolved oxygen concentrations after herbicide application, even though an aerator was placed in the pond to provide fish with an area of oxygenated water. Several bass were part of that fish kill. The cause of the other fish kill was difficult to determine. All water quality parameters were within normal ranges for largemouth bass and bluegill at all times measured. A large amount of fluke parasites were found on the dying fish, and one bluegill was shocked during a later fish sampling had severe exophthalmia. The fish kill consisted mainly of bluegill, but some bass were also killed. It is possible that an outbreak of columnaris occurred since these ponds are typically used for aquaculture. Another possible factor that may have contributed to low survival of bass was the presence of predators. Several fish-eating birds were occasionally observed around the ponds. Egrets, herons, and even a cormorant were present. The cormorant was only present for a short time but may have had severe impacts on the small ponds. Herons may have been able to capture small

bass during the morning hours when oxygen levels were low and fish were near the surface (Wywialowski 1999).

Largemouth bass reproduction occurred in only one pond. One of the grass carp treatment ponds yielded 75 fingerlings and one small adult bass during pond harvest. Six of the originally stocked bass were also harvested. Therefore, the pond experienced two spawning events in the second year of the study. One spawn occurred in the early spring resulting in one bass surviving to sexual maturation. Another spawn must have occurred in the fall just before the harvest of fish in November resulting in the numerous fingerlings collected. Multiple spawning events within the same year are possible, especially in warmer climates with adequate forage (Davis and Lock 1997). Due to the stress of spawning and guarding fry, largemouth must be in excellent condition in order to perform multiple spawns within a year. Spawning may have occurred in other ponds, but recruitment in those ponds was zero. Several studies have shown a decrease in the survival of young largemouth bass following macrophyte removal as a result of increased predation by piscivores (Bettoli et al. 1993; Klussmann et al. 1988; Sammons et al. 2005). Sammons et al. (2005) found that removal of macrophytes led to an increase in egg production in largemouth bass, due to improved condition as a result of greater prey vulnerability, but reduced survival of young bass.

Statistical analysis showed that there were significant differences between the grass carp treatment ponds and the herbicides treatment ponds with regards to both largemouth bass mean weight and mean length. Largemouth bass were significantly longer and heavier in the grass carp ponds. There were no significant differences

between the control ponds and the other treatments. Juvenile mean weight and total weight was significantly higher in the grass carp ponds than in the other treatments since no juveniles were collected in the herbicide or control ponds. It was expected that largemouth bass would be larger in the herbicide and grass carp treatments since it has been observed in several past studies (Klussmann et al. 1988; Luedke 1987; Pothoven et al. 1999; Sammons et al. 2005). The multivariate analysis showed a weak relationship between largemouth bass production and grass carp ponds. This is due to the fact that the most bass were collected in the grass carp ponds.

Bluegill appeared to be slightly longer and heavier in the control ponds when compared to the herbicide and grass carp ponds. However, the difference was not significant. It was hypothesized that adult bluegill would be larger in the herbicide and grass carp ponds. Pothoven et al. (1999), observed an increase in bluegill growth rates after macrophyte removal. It may be possible that the early fish kills had an impact on adult bluegill populations in the herbicide ponds. The differences may also be the result of a few large bluegill from the control ponds. Klussmann et al. (1988) and Luedke (1987) also observed a decrease in bluegill size and decline in population after grass carp stocking and macrophyte removal. It is possible that the decrease in macrophytes may have resulted in increased predation on bluegills by largemouth bass and other predators (Bettoli et al. 1993; Klussmann et al. 1988; Luedke 1987). The decrease in macrophytes may have decreased the macroinvertebrate population, which serves as the main food source for bluegills.

Analysis of juvenile bluegill collected indicated that the control ponds had higher total weights of juvenile bluegill than the herbicide and grass carp ponds although it was not significantly higher. Mean weight of juvenile bluegill was lowest in the control ponds and relatively similar in the herbicide and grass carp ponds. This supports the hypothesis that juvenile bluegill would have lower survival in grass carp and herbicide ponds than in the controls. The higher percent coverage of macrophytes in the control ponds provided juvenile bluegill with protection from predation (Savino and Stein 1982). The increased number of juvenile bluegills led to an increase in total juvenile bluegill weight. However, more juvenile bluegills caused greater competition amongst them for food. This may explain the decreased individual weight of juveniles in the control ponds when compared to the herbicide and grass carp ponds where fewer juveniles survived predation. Several studies indicated that removal of macrophytes led to decreased survival of juvenile bluegill as a result of predation and decreased macroinvertebrate population (Bettoli et al. 1993; Klussmann et al. 1988; Luedke 1987).

It is possible that macrophyte coverage could be indirectly impacting the bluegill population and the macroinvertebrate population by directly impacting the other entity. In other words, macrophyte removal may decrease the bluegill population, which then increases the macroinvertebrate population. At the same time, macrophyte removal may decrease the macroinvertebrate population, which also decreases the bluegill population. Although the multivariate analysis did not reveal a significant relationship between bluegill, macroinvertebrates, and macrophytes, it is a topic which deserves further research.

4.2 Conclusions

Pond ecology has long been a neglected topic in the field of aquatic vegetation management. As problems with aquatic vegetation become more prolific, it will be vital that the ecological impacts of vegetation management be understood. The impacts of aquatic vegetation and aquatic vegetation management are often more pronounced in private impoundments due to their small size relative to lakes and reservoirs. The results of this study will be useful in helping pond owners and managers in making management decisions with regards to aquatic vegetation problems they encounter in their small ponds in Texas. This study showed that there are potential interactions between various biotic and abiotic factors when aquatic vegetation management is implemented. The use of aquatic herbicides and grass carp both have significant impacts on the ecology of small impoundments that have experienced problems with aquatic vegetation. The type of aquatic vegetation present also determines the magnitude of the impacts that various management techniques have on pond ecology.

Future research should be conducted focusing on the ecological impacts of aquatic vegetation management on small impoundments. Studies should be conducted in a variety of climates, soils, and watersheds. Further research will help determine if the trends observed in this study are valid and if other trends exist. Continued research will help pond managers and private pond owners conduct better management practices resulting in healthier and more productive impoundments.

REFERENCES

Allen, S.K., and R.J. Wattendorf. 1987. Triploid grass carp: status and management implications. Fisheries 12(4):20-24.

AquaPlant. 2008. Texas AgriLife Extension. Available: www.aquaplant.tamu.edu. (August 2008).

Ashton, F.M., and A.S. Crafts. 1973. Mode of action of herbicides. Wiley, New York.

Bartell, S.M., K.R. Campbell, C.M. Lovelock, S.K. Nair, and J.L. Shaw. 2000. Characterizing aquatic ecological risks from pesticides using a diquat dibromide case study III. Ecological process models. Environmental Toxicology and Chemistry 19(5):1441-1453.

Bettoli, P., M. J. Maceina, R. Noble, and R. Betsill. 1993. Response of a reservoir fish community to aquatic vegetation removal. North American Journal Fisheries Management 13:110–124.

Brown, S. J., and M. J. Maceina. 2002. The influence of disparate levels of submersed aquatic vegetation on largemouth bass population characteristics in a Georgia reservoir. Journal of Aquatic Plant Management 40:28–35.

Carter, C.C., and R.S. Hestand. 1977a. The effects of selected herbicides on phytoplankton and sulfur bacteria populations. Journal of Aquatic Plant Management 15:47-56.

Carter, C.C., and R.S. Hestand. 1977b. Relationship of regrowth of aquatic macrophytes after treatment with herbicides to water quality and phytoplankton populations. Journal of Aquatic Plant Management 15:65-69.

Carter, V., N. B. Rybicki, and R. Hammerschlag. 1991. Effects of submersed macrophytes on dissolved oxygen, pH, and temperature under different conditions of wind, tide, and bed structure. Journal Freshwater Ecology 6:121-133.

Cassani, J.R., and W.E. Caton. 1986. Efficient production of grass carp (Ctenopharyngodon idella) utilizing hydrostatic pressure. Aquaculture 55(1):43-50.

Chu, A., T.A. Thayer, B.W. Ford, D.F. Unites, and J.F. Roetzer. 1978. Copper in the aquatic environment: a literature review. Washington Public Power Supply. Envirosphere Company. Bellevue, Washington.

Crowder, L. B., and W. E. Cooper. 1979. Structural complexity and fish prey interactions in ponds: a point of view. Pages 2–10 *in* D.L. Johnson and R.A. Stein, editors. Response of fish to habitat structure in standing water. North Central Division, American Fisheries Society, Special Publication 6, Bethesda, Maryland.

Emmett, K. 2002. Final supplemental environmental impact statement for diquat dibromide. Washington State Department of Ecology. Water Quality Program. Publication No. 02-10-052. Olympia, Washington.

Glomski, L. A. M., J. G. Skogerboe, and K. D. Getsinger. 2005. Comparative efficacy of diquat for control of two members of the *Hydrocharitaceae*: elodea and hydrilla. Journal of Aquatic Plant Management 43:103–105.

Hallingse, M.E., and E.J. Phlips. 1996. Effects of Cutrine-Plus and Cide-kick II on the growth of algae and cyanobacteria. Journal of Aquatic Plant Management 34:39-40.

Harman, W. H., L. P. Hingula, and C. E. MacNamara. 2005. Does long-term macrophyte management in lakes affect biotic richness and diversity? Journal of Aquatic Plant Management 43:57–64.

Hasle, G. R. 1978. The inverted-microscope method. Pages 88–96 *in* A. Sournia, editor. Phytoplankton manual. UNESCO, Paris, France.

Hestand, R.S., and C.C. Carter. 1978. Comparative effects of grass carp and selected herbicides on macrophyte and phytoplankton communities. Journal of Aquatic Plant Management 16:43-50.

Hiltibran, R. C., D. L. Underwood, and J. S. Fickle. 1972. Fate of diquat in the aquatic environment. University of Illinois, Water Resource Center. Final Report to Office of Water Resources Research, Dept. of the Interior. Project No. A-035-ILL. Urbana, Illinois.

Hoyer, M. V., and D. E. Canfield, Jr. 1996. Largemouth bass abundance and aquatic vegetation in Florida lakes: an empirical analysis. Journal of Aquatic Plant Management 34:23–32.

Kilgore, K. J., and J. J. Hoover. 2001. Effects of hypoxia on fish assemblages in a vegetated waterbody. Journal of Aquatic Plant Management 39:40–44.

Kirkagac, M., and N. Demir. 2004. The effects of grass carp on aquatic plants, plankton, and benthos in ponds. Journal of Aquatic Plant Management 42:32–39.

Klussmann, W. G., R. L. Noble, R. D. Martyn, W. J. Clark, R. K. Betsill, P. W. Bettoli, M. F. Cichra, and J. M. Campbell. 1988. Control of aquatic macrophytes by grass carp in Lake Conroe, Texas, and the effects on the reservoir ecosystem. Texas Agricultural Experiment Station, Department of Wildlife and Fisheries Sciences, Texas A&M University. MP-1664, College Station, Texas.

Langeland, K.A., O.N. Hill, T.J. Koschnick, and W.T. Haller. 2002. Evaluation of a new formulation of Reward landscape and aquatic herbicide for control of duckweed, water hyacinth, water lettuce, and hydrilla. Journal of Aquatic Plant Management 40:51-53.

Lenz, J. 2000. Introduction. Pages 1–32 *in* R.P. Harris, P.H. Wiebe, J. Lenz, H.R. Skjoldal, and M. Huntley, editors. Zooplankton methodology manual. Academic Press, San Diego, California.

Lepš, J, and P. Šmilauer. 2003. Multivariate analysis of ecological data using CANOCO. Cambridge University Press, Cambridge, United Kingdom.

Luedke, M. W. 1987. Impacts of grass carp and hybrid carp on aquatic macrophytes and endemic fish populations in a cooling reservoir. Dissertation. Department of Wildlife and Fisheries Sciences, Texas A&M University, College Station, Texas.

Maceina, M. J. 1996. Largemouth bass abundance and aquatic vegetation in Florida lakes: an alternate interpretation. Journal of Aquatic Plant Management 34:43–47.

Maceina, M. J., and J. W. Slipke. 2004. The use of herbicides to control hydrilla and the effects on young largemouth bass population characteristics and aquatic vegetation in Lake Seminole, Georgia. Journal of Aquatic Plant Management 42:5–11.

Masser, M.P. 1996. Management of recreational fish ponds in Alabama. ANR-577. Alabama Cooperative Extension, Auburn, Alabama.

Masser, M. P. 2002. Using grass carp in aquaculture and private impoundments. Southern Regional Aquaculture Center. SRAC Publication No. 3600, Stoneville, Mississippi.

Mastin, B.J., and J.H. Rodgers. 2000. Toxicity and bioavailability of copper herbicides (clearigate, cutrine-plus, and copper sulfate) to freshwater animals. Archives of Environmental Contamination and Toxicology 39: 445-451.

Miranda, L. E., and L. L. Pugh. 1997. Relationship between vegetation coverage and abundance, size, and diet of juvenile largemouth bass during winter. North American Journal of Fisheries Management 17:601–610.

Mulrennan, J.A. 1962. The relationship of mosquito breeding to aquatic plant production. Journal of Aquatic Plant Management 1:6-7.

National Science and Technology Council. 2007. A strategy for federal science and technology to support water availability and quality in the United States. Report of the Committee on Environment and Natural Resources. Subcommittee on Water Availability and Quality. Executive Office of the President. Washington D.C.

Nelson, L. S., and J. F. Shearer. 2005. 2,4-D and *Mycoleptodiscus terrestris* for control of Eurasian watermilfoil. Journal of Aquatic Plant Management 43:29–34.

Noble, R. L. 1980. Management of lakes, reservoirs, and ponds. Pages 265–295 *in* R.T. Lackey and L.A. Nielsen, editors. Fisheries management. John and Wiley Sons, New York.

Palmer, M.W. 1993. Putting things in even better order: the advantages of canonical correspondence analysis. Ecology 74(8):2215-2230.

Peterson, H.G., C. Boutin, K.E. Freemark, and P.A. Martin, 1997. Toxicity of hexazinone and diquat to green algae, diatoms, cyanobacteria, and duckweed. Aquatic Toxicology 39(2):111-134.

Pieterse, A. H. 1990. Introduction. Pages 3–16 *in* A.H. Pieterse and K.J. Murphy, editors. Aquatic weeds: the ecology and management of nuisance aquatic vegetation. Oxford University Press, New York.

Poovey, A.G., and K.D. Getsinger. 2002. Impacts of inorganic turbidity on diquat efficacy against Egeria densa. Journal of Aquatic Plant Management 40:6-10.

Pothoven, S., A. B. Vondracek, and D. L. Pereira. 1999. Effects of vegetation removal on bluegill and largemouth bass in two Minnesota lakes. North American Journal of Fisheries Management 19:748–757.

Rabeni, C. F. 1996. Invertebrates. Pages 335–352 *in* B.R. Murphy and D.W. Willis, editors. Fisheries techniques. Second Edition. American Fisheries Society, Bethesda, Maryland.

Sammons, S. M., and M. J. Maceina. 2005. Effect of Aquathol K treatments on activity patterns of largemouth bass in two coves of Lake Seminole, Georgia. Journal of Aquatic Plant Management 43:17–24.

Sammons, S. M., M. J. Maceina, and D. G. Partridge. 2005. Population characteristics of largemouth bass associated with changes in abundance of submerged aquatic vegetation in Lake Seminole, Georgia. Journal of Aquatic Plant Management 43:9–16.

Savino, J. F., and R. A. Stein. 1982. Predator-prey interaction between largemouth bass and bluegills as influenced by simulated, submerged vegetation. Transactions of American Fisheries Society 111:255–266.

Scheffer, M. 1998. Ecology of shallow lakes. Chapman & Hall, London, United Kingdom.

Schonrock, A.E. 2005. An internet survey of private pond owners and managers in Texas. Master's Thesis. Department of Wildlife & Fisheries Sciences, Texas A&M University, College Station, Texas.

Shelton, J.L., and T.R. Murphy. 1989. Aquatic weed management: control methods. Publication No. 360. Southern Regional Aquaculture Center. Texas A&M University, College Station, Texas.

Shireman, J.V., and C.R. Smith. 1983. Synopsis of biological data on the grass carp Ctenopharynogodon idella (Cuvier and Valenciennes, 1844), Volume 135. FAO Fisheries Synopsis. Food and Agriculture Organization of the United Nations, Rome, Italy.

Smale, M.A., and C.F. Rabeni. 1995. Hypoxia and hypothermia tolerances of headwater stream fishes. Transactions of American Fisheries Society 124(5):698-710.

Ter Braak, C.J.F. and P.F.M. Verdonschot. 1995. Canonical correspondence analysis and related multivariate methods in aquatic ecology. Aquatic Sciences 57(3):255-289.

Ter Braak, C. J. F. and P. Smilauer. 2002. CANOCO Reference manual and CanoDraw for Windows User's guide: software for Canonical Community Ordination (version 4.5). Microcomputer Power. Ithaca, New York.

Texas Agricultural Extension Service. 1996. Texas farm ponds: stocking, assessment, and management recommendations. Special Publication No 1, Texas Chapter American Fisheries Society, Texas A&M University, College Station, Texas.

Texas Chapter of American Fisheries Society. 2005. Texas farm ponds: stocking, assessment, and management recommendations. Special Publication No. 1. Texas Chapter of American Fisheries Society. College Station, Texas.

Texas State Data Center. 2006. Summary of trends in Texas. Texas State Data Center. Office of the State Demographer. Available: http://txsdc.utsa.edu/tpepp/2006projections/summary/. (August 2008).

Texas Water Development Board. 2004. A Texan's guide to water and water rights marketing. Texas Water Development Board. Austin, Texas.

U.S. Census Bureau. 2008. International database: country summary: United States. U.S. Census Bureau. Available: http://www.census.gov/ipc/www/idb/country/usportal.html. (August 2008).

USDA (United States Department of Agriculture), Natural Resources Conservation Service. 1997. Ponds-planning, design, construction. Agriculture Handbook 590. Washington, D.C.

Watson, C. 1989. Use of copper in aquaculture and farm ponds. IFAS Fact Sheet, FA-13. University of Florida, Gainesville, Florida.

Wetzel, R. G., and G. E. Likens. 2000. Limnological analyses, third edition. Springer Science Business Media, Inc. New York.

Wetzel, R.G. 2001. Limnology: lake and river ecosystems, third edition. Academic Press, San Diego, California.

Washington State Department of Ecology. 1992. Aquatic plants management program for Washington State. Vol. 1: Final Supplemental Environmental Impact Statement and Responsiveness Summary. Olympia, Washington.

Weed Science Society of America. 1994. Herbicide handbook, seventh edition. Champagne, Illinois.

Wywialowski, A.P. 1999. Wildlife-caused losses for producers of channel catfish *Ictalurus punctatus* in 1996. Journal World Aquaculture Society 30(4):461-472.

APPENDIX A

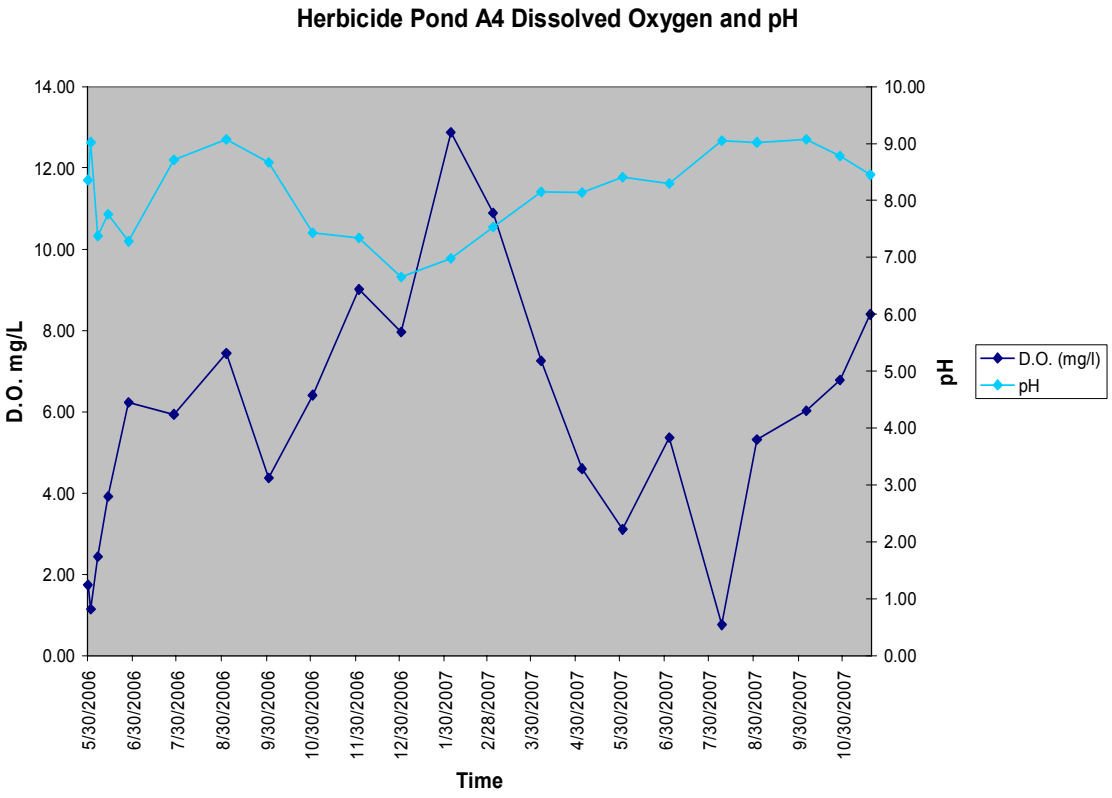

Figure A-1. Dissovled Oxygen and pH for herbicide pond A4

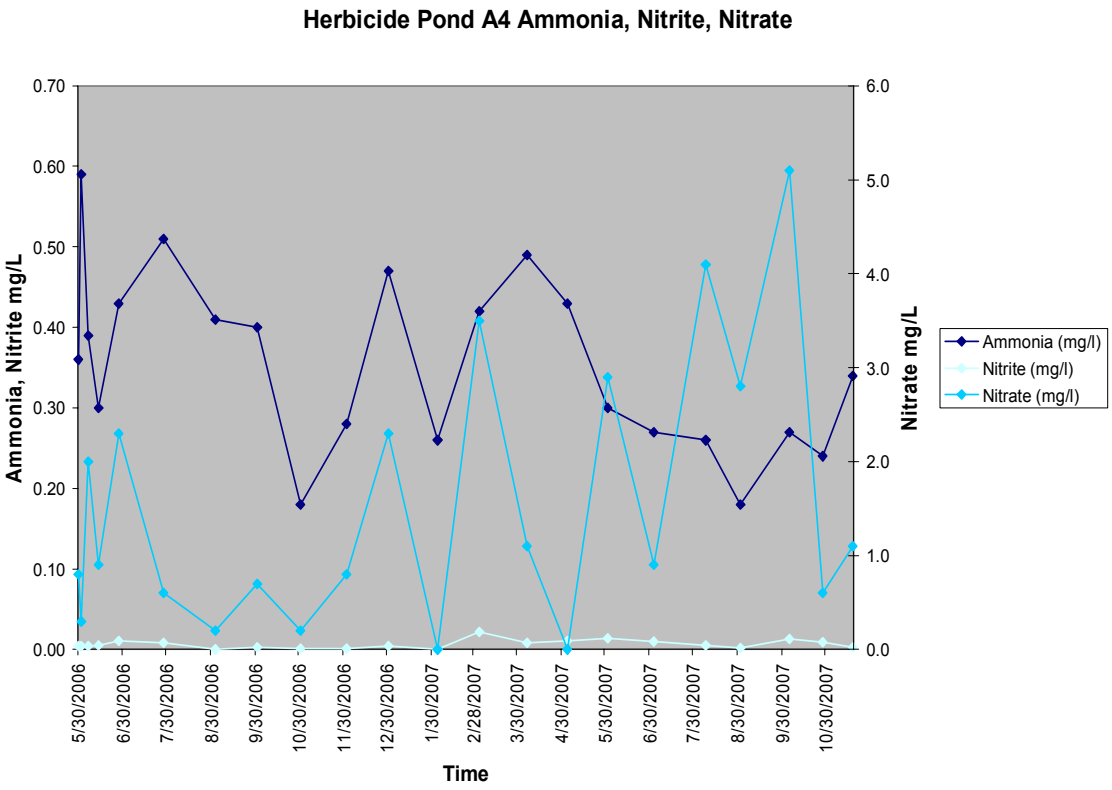

Figure A-2. Ammonia, nitrite, and nitrate for herbicide pond A4

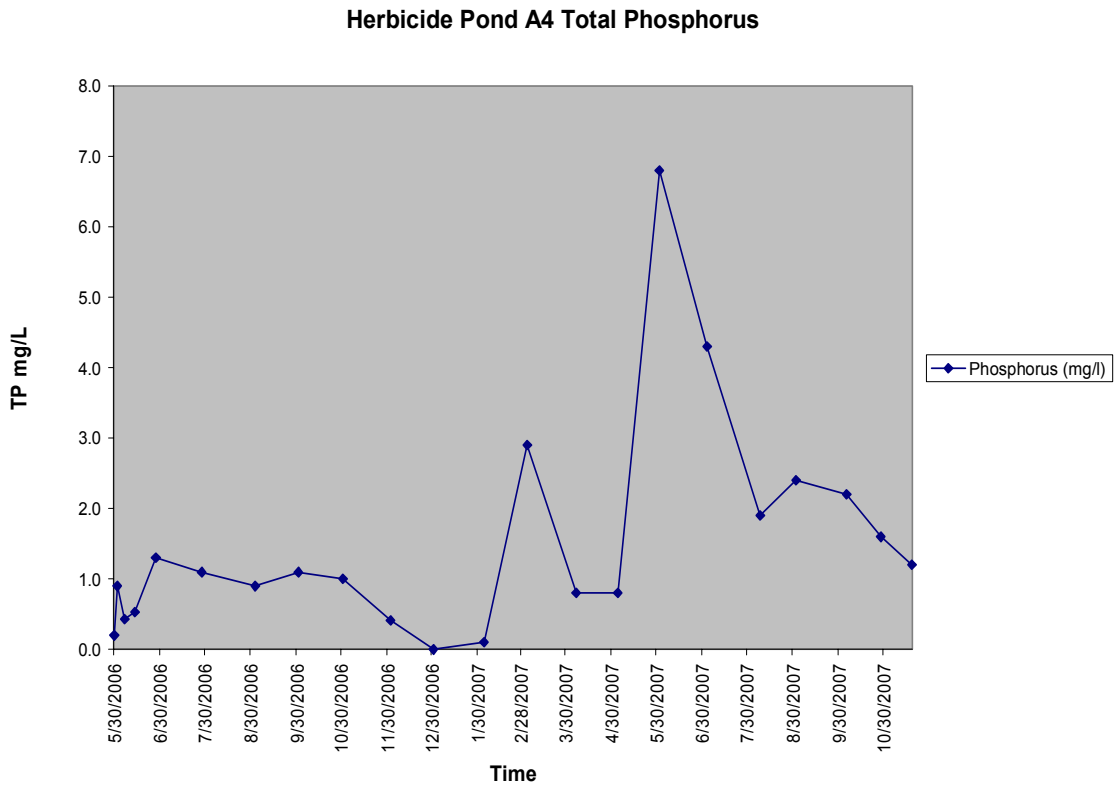

Figure A-3. Total phosphorus for herbicide pond A4

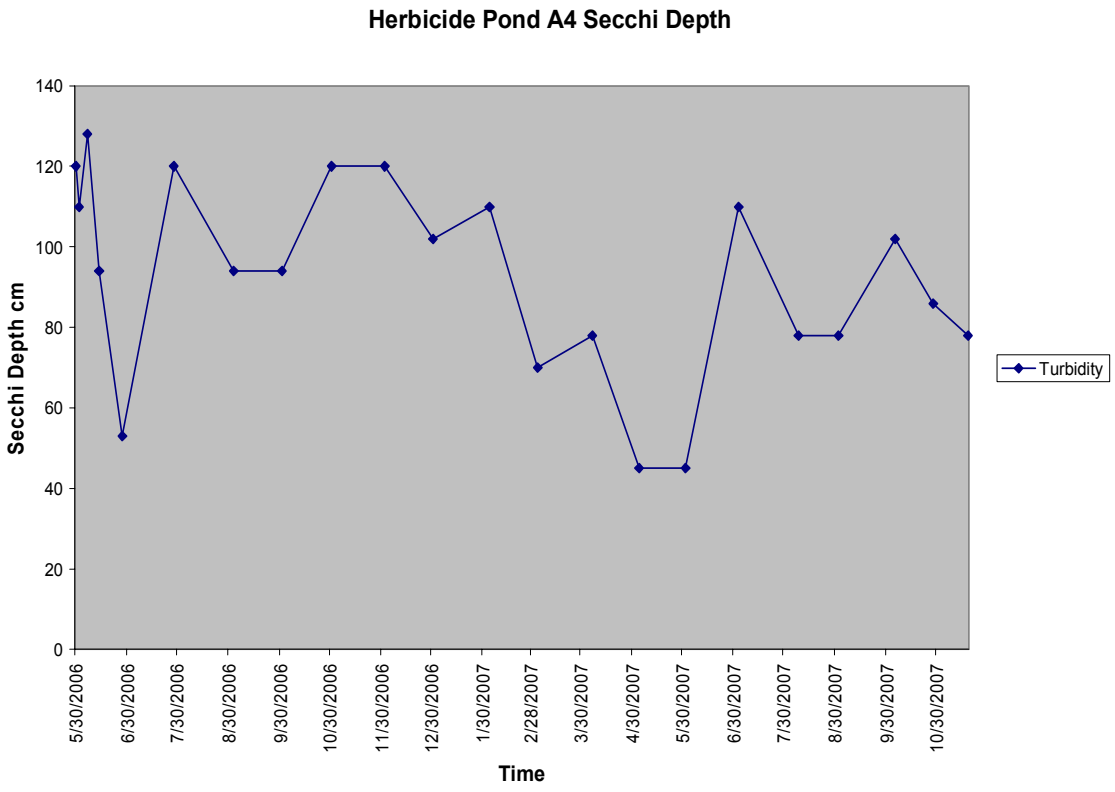

Figure A-4. Secchi depth for herbicide pond A4

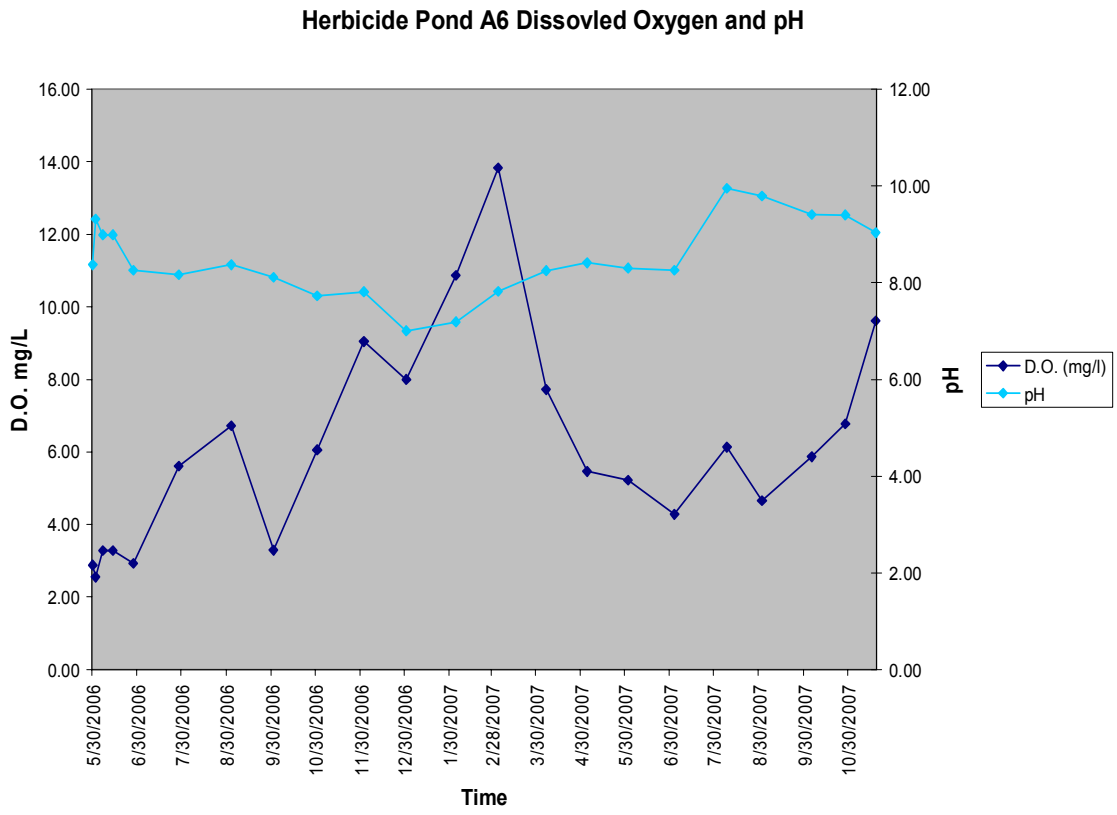

Figure A-5. Dissolved oxygen and pH for herbicide pond A6

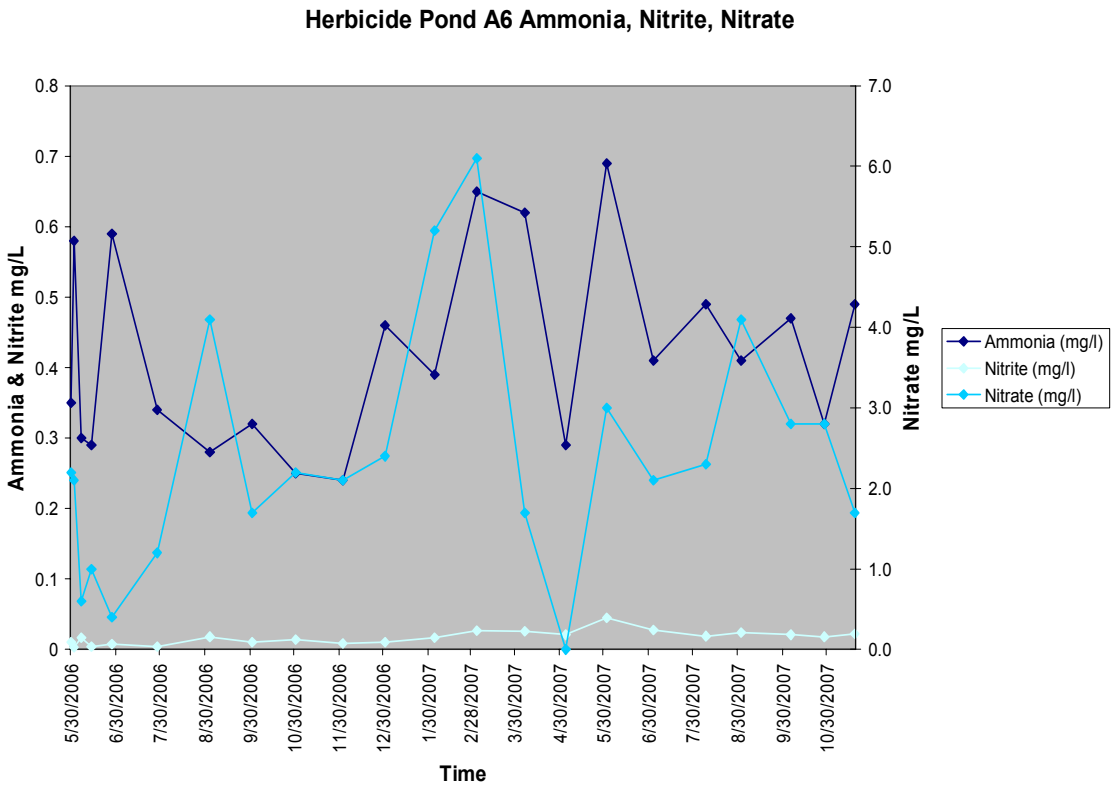

Figure A-6. Ammonia, nitrite, and nitrate for herbicide pond A6

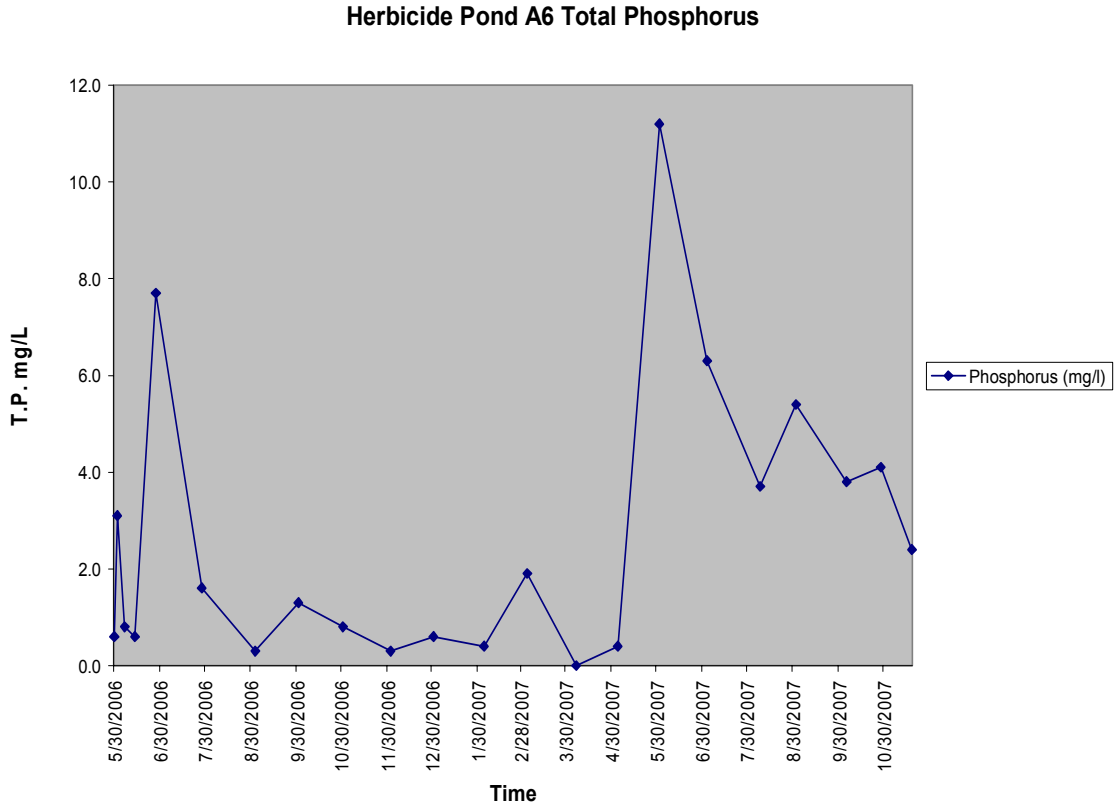

Figure A-7. Total phosphorus for herbicide pond A6

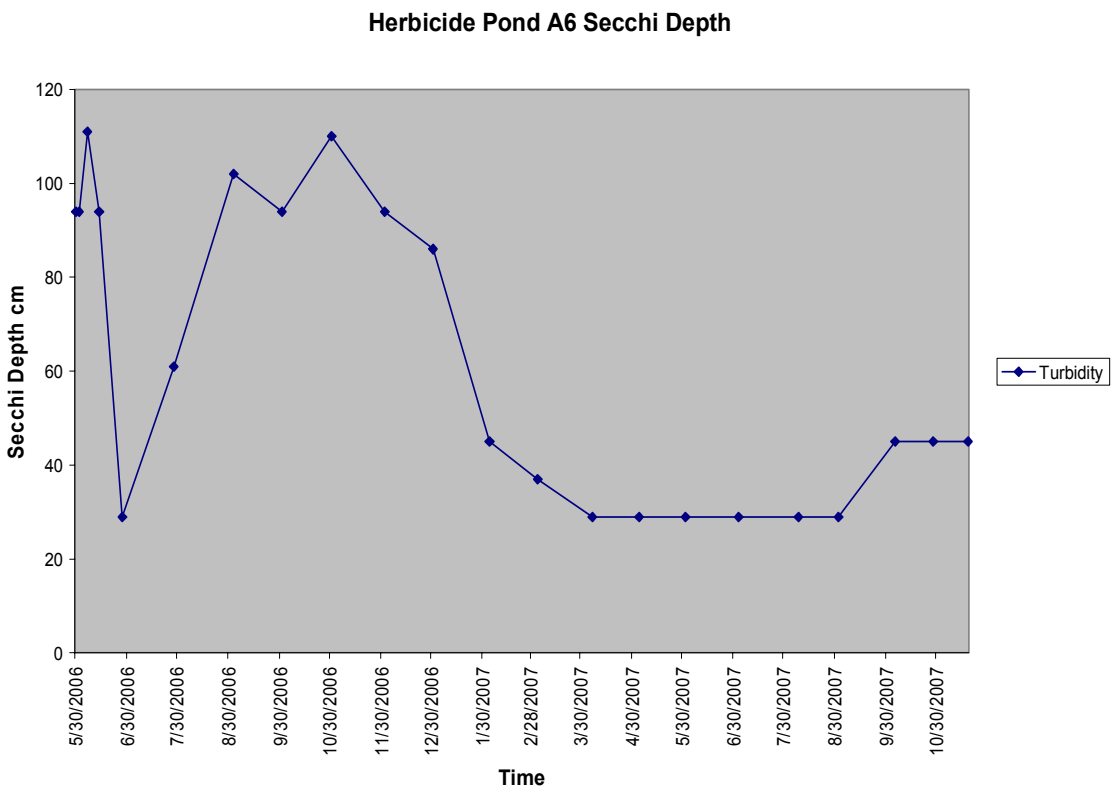

Figure A-8. Secchi depth for herbicide pond A6

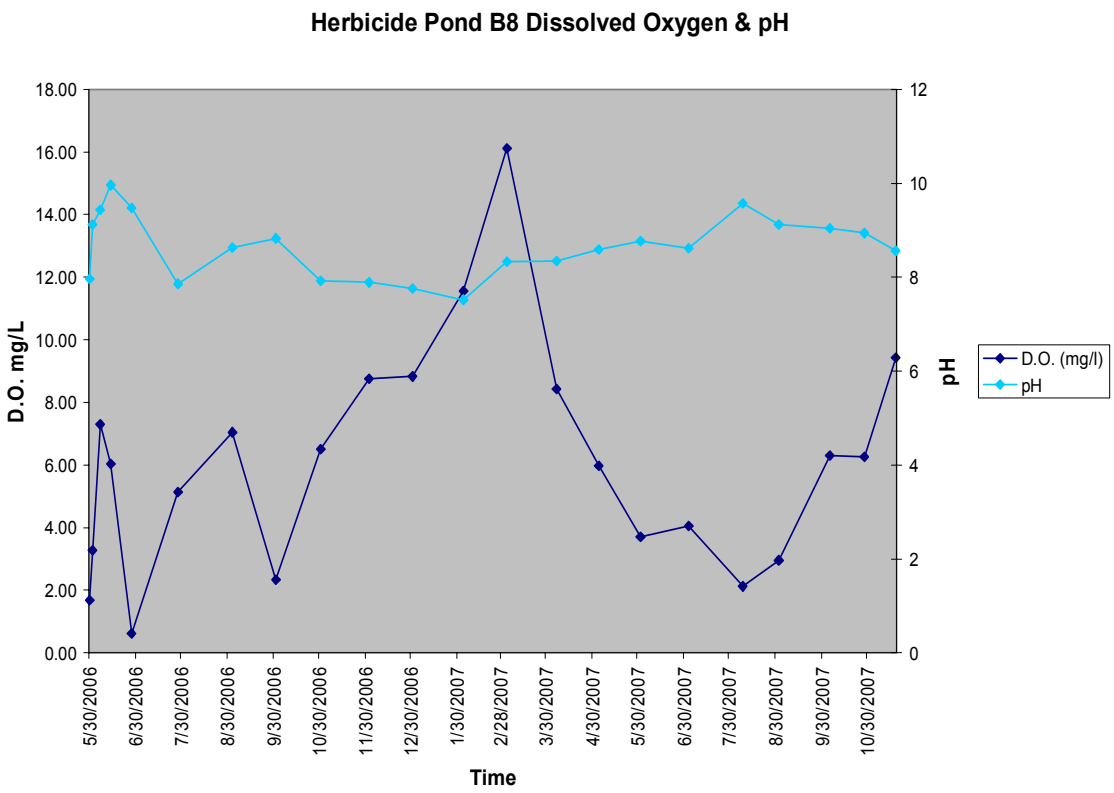

Figure A-9. Dissolved oxygen and pH for herbicide pond B8

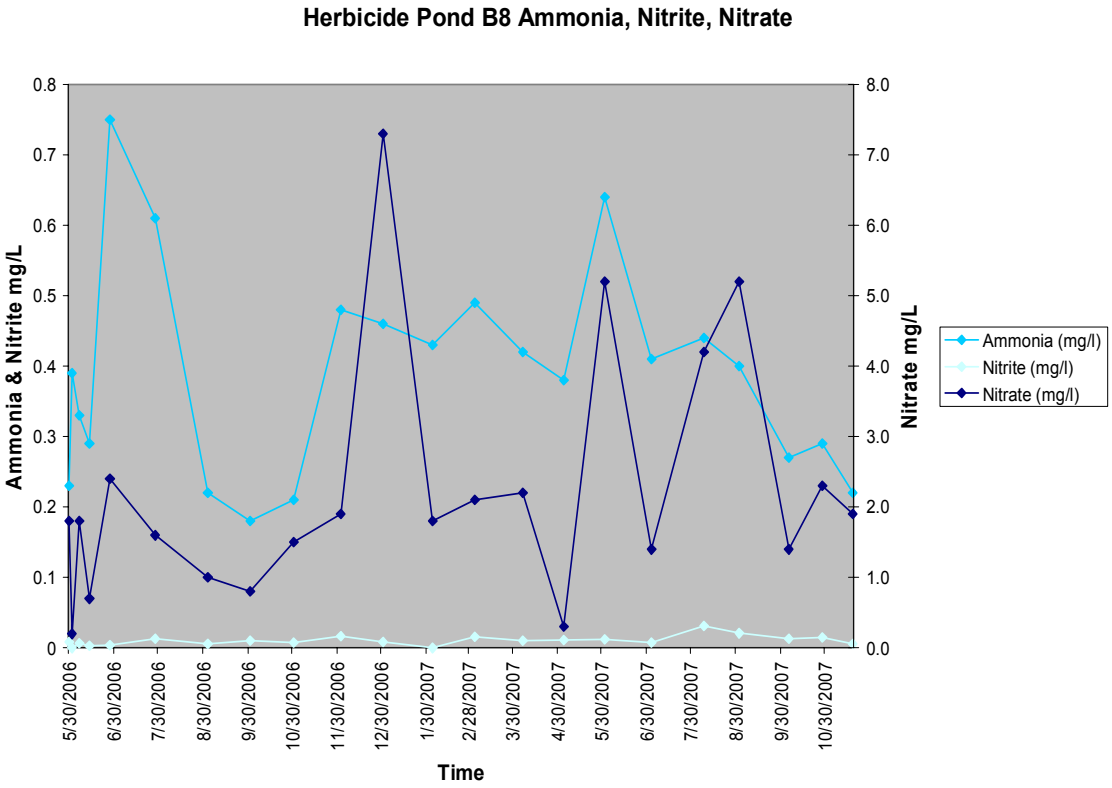

Figure A-10. Ammonia, nitrite, and nitrate for herbicide pond B8

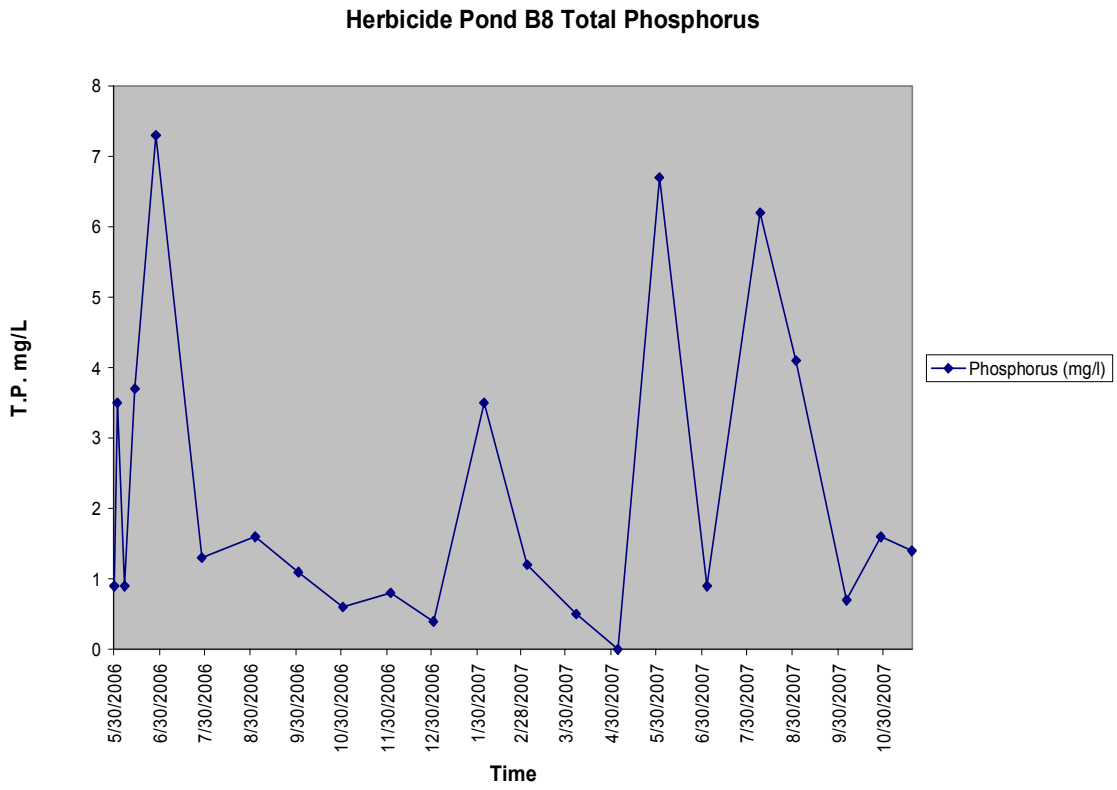

Figure A-11. Total phosphorus for herbicide pond B8

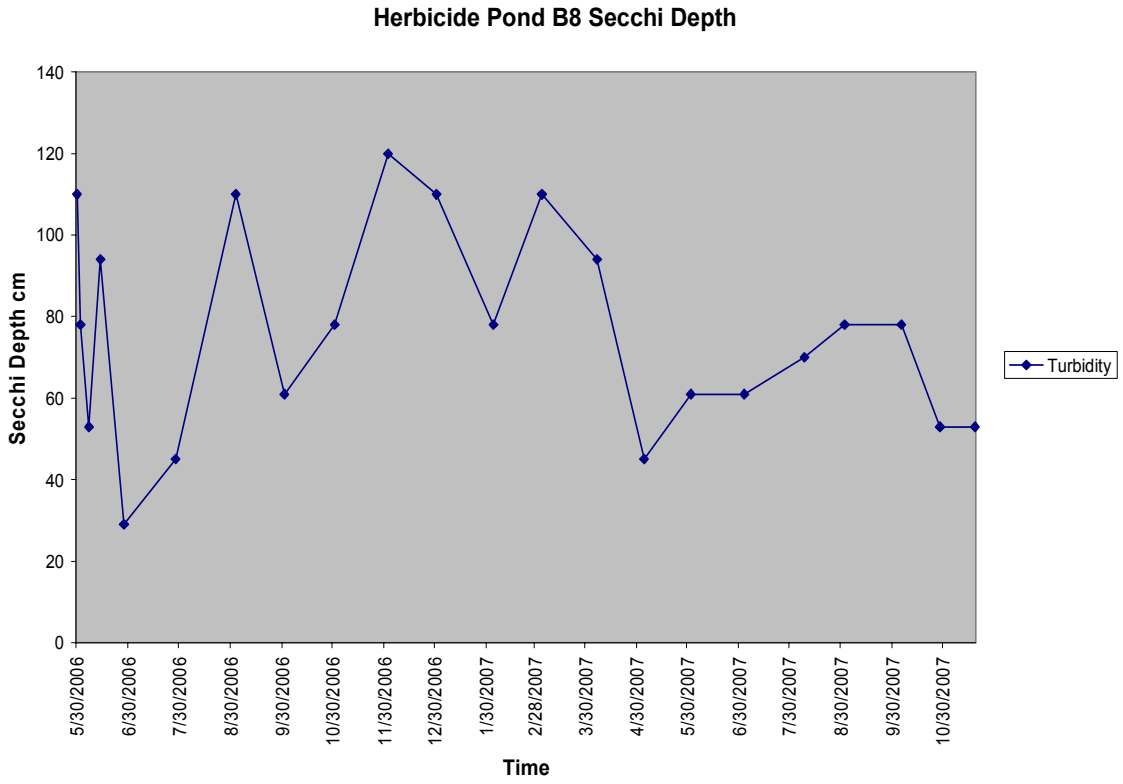

Figure A-12. Secchi depth for herbicide pond B8

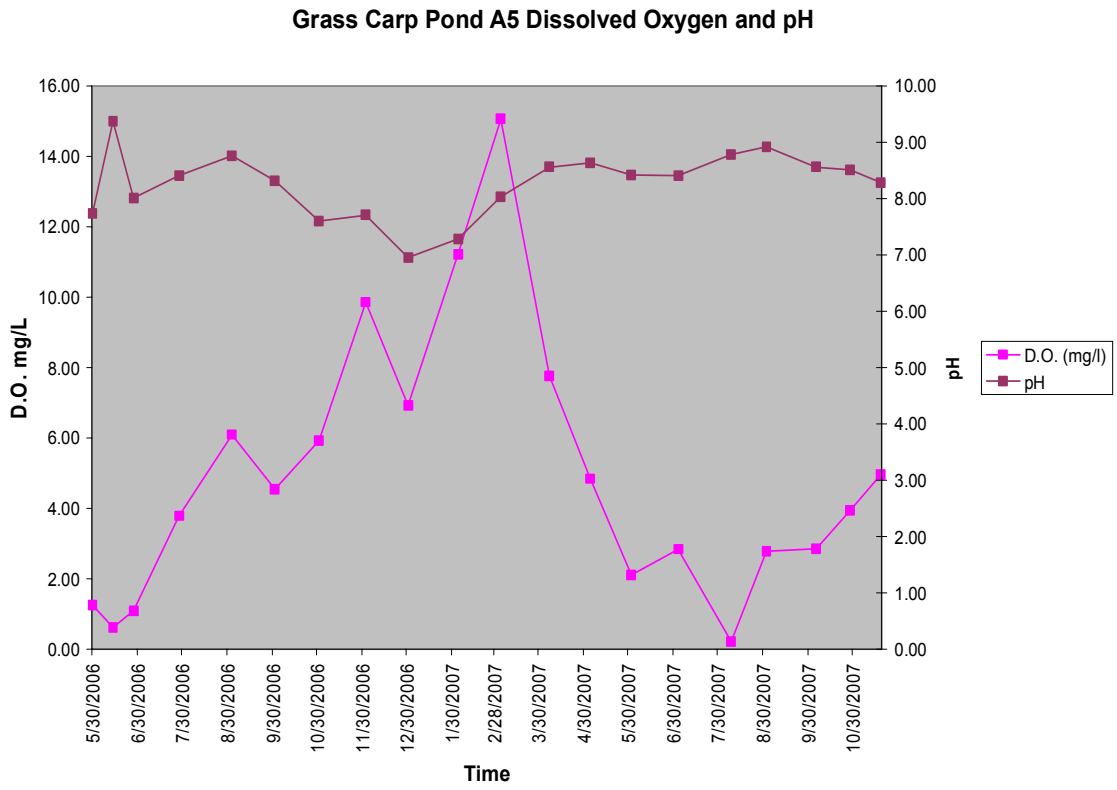

Figure A-13. Dissolved oxygen and pH for grass carp pond A5

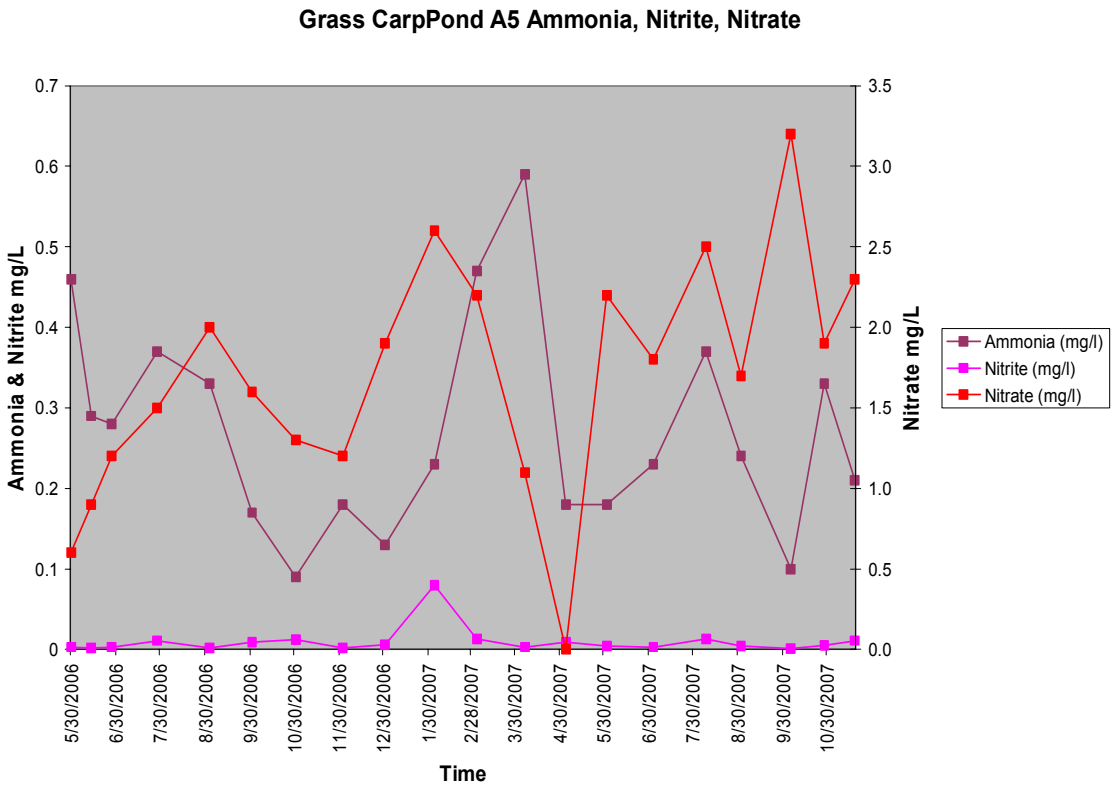

Figure A-14. Ammonia, nitrite, and nitrate for grass carp pond A5

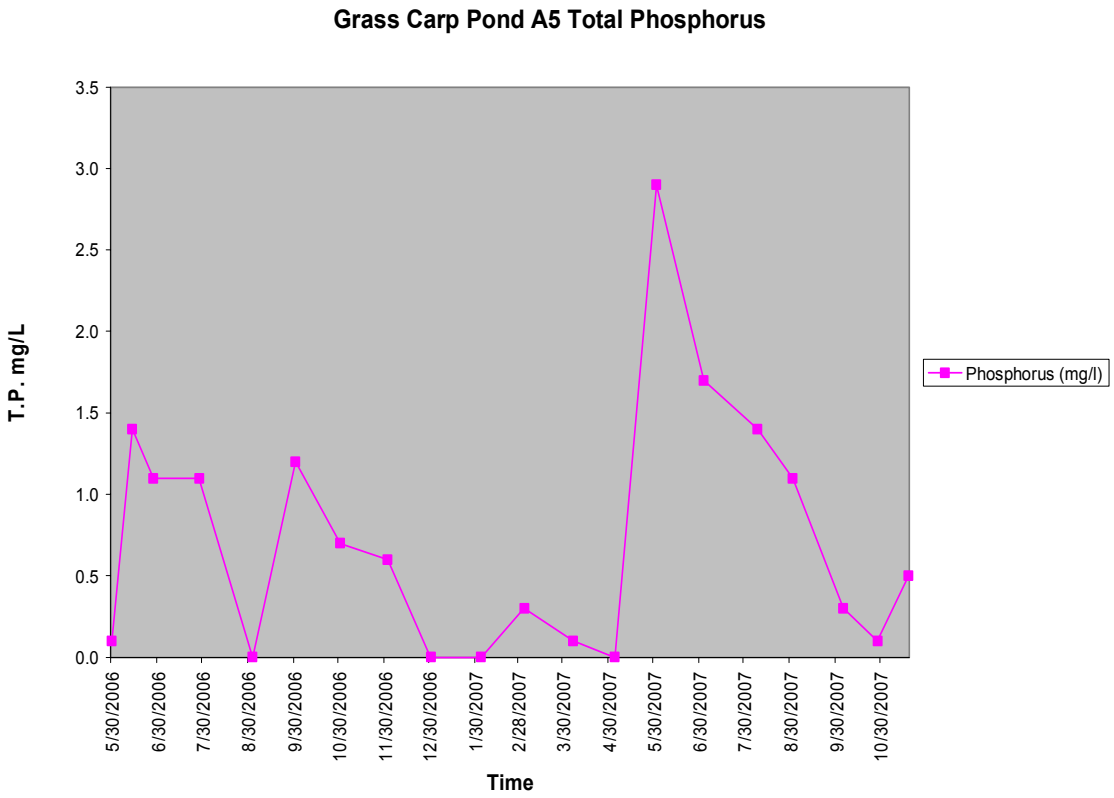

Figure A-15. Total phosphorus for grass carp pond A5

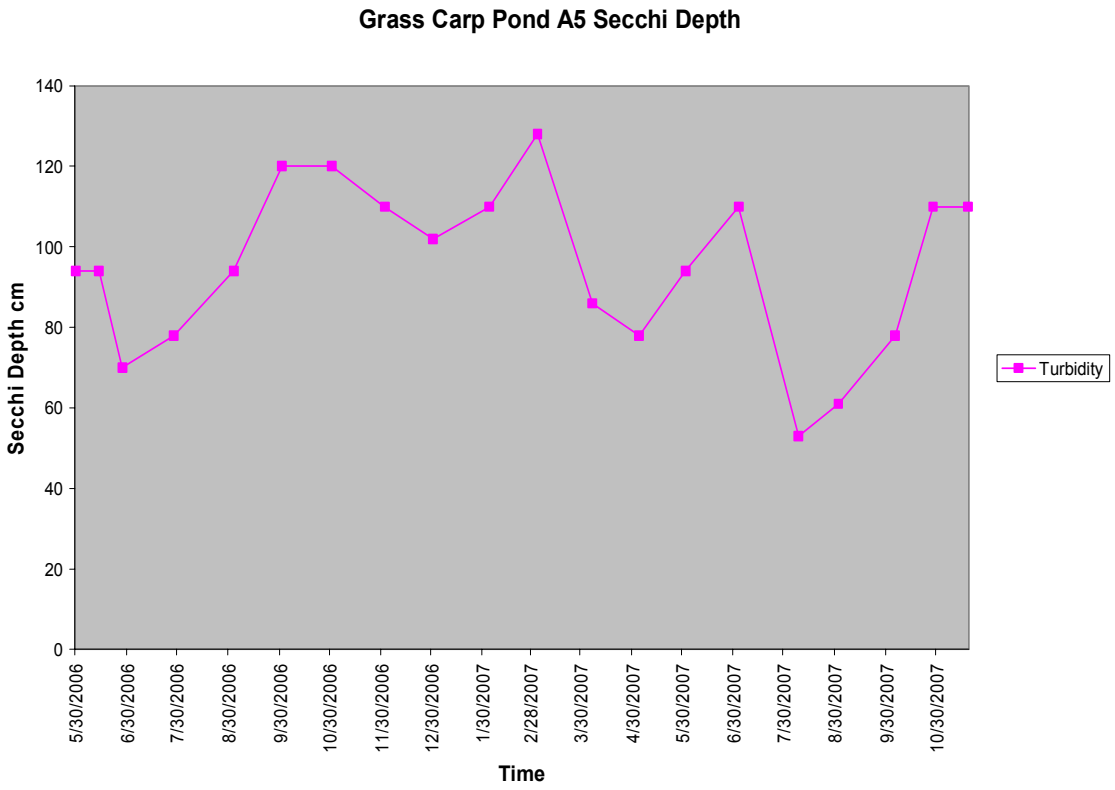

Figure A-16. Secchi depth for grass carp pond A5

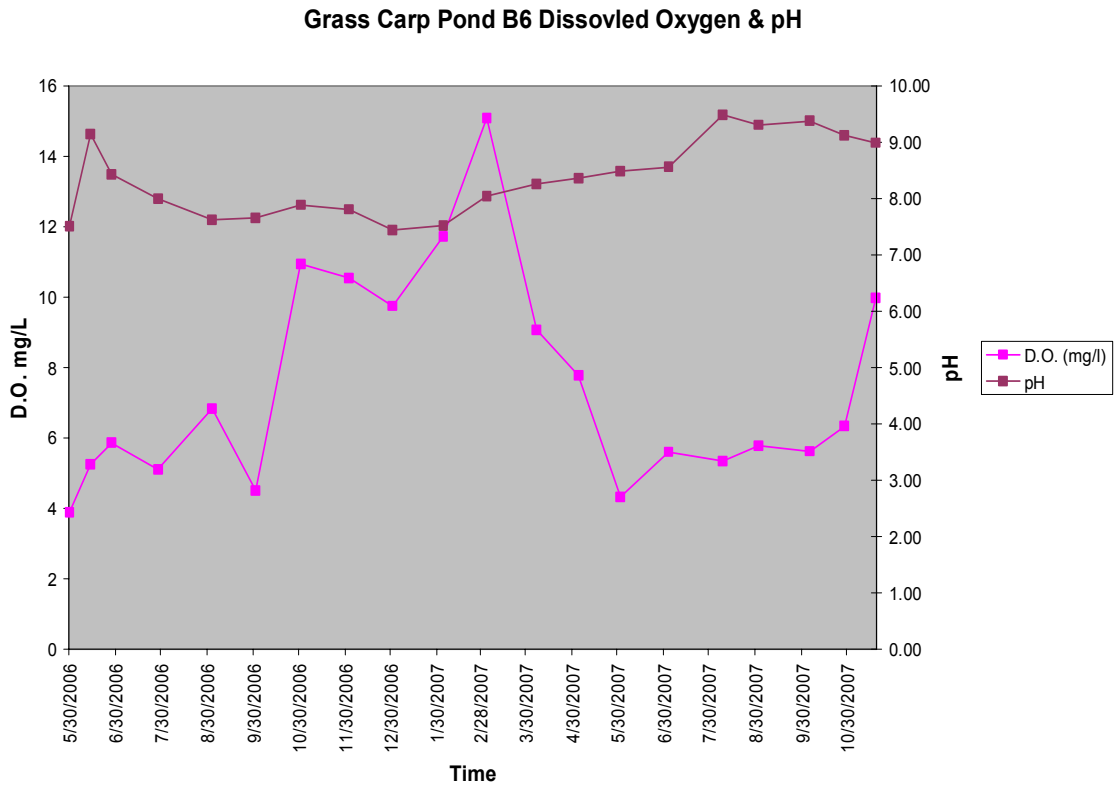

Figure A-17. Dissolved oxygen and pH for grass carp pond B6

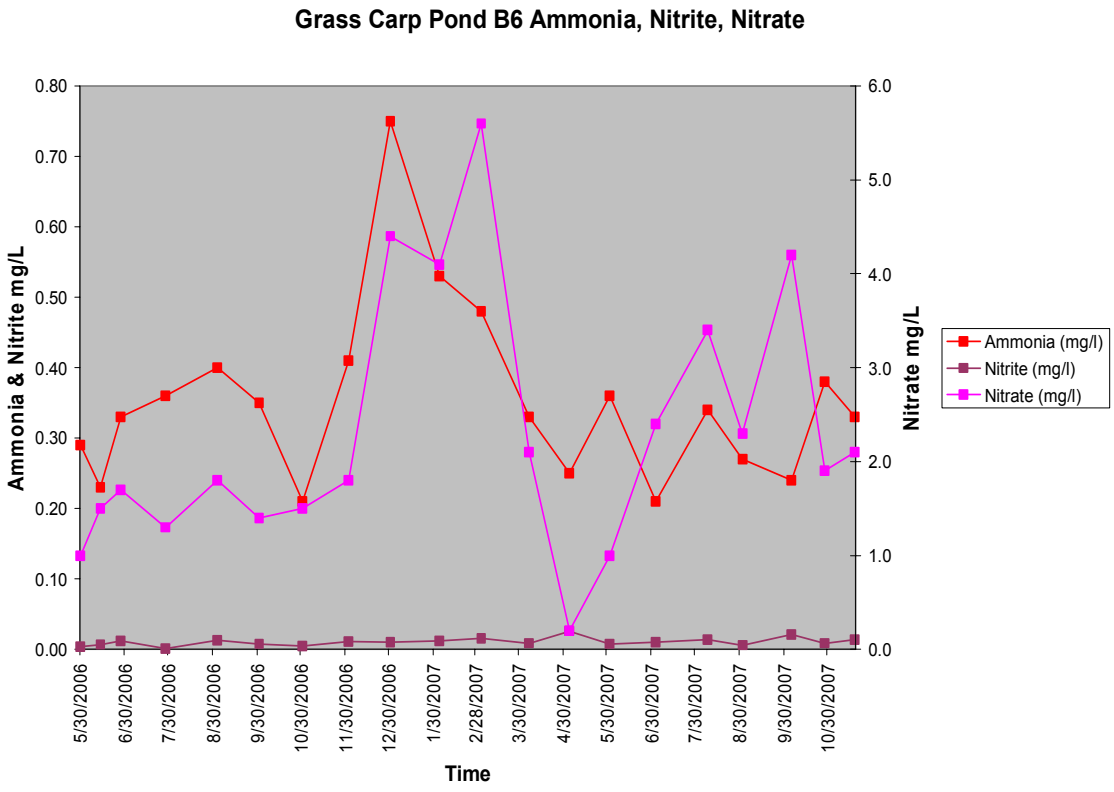

Figure A-18. Ammonia, nitrite, and nitrate for grass carp pond B6

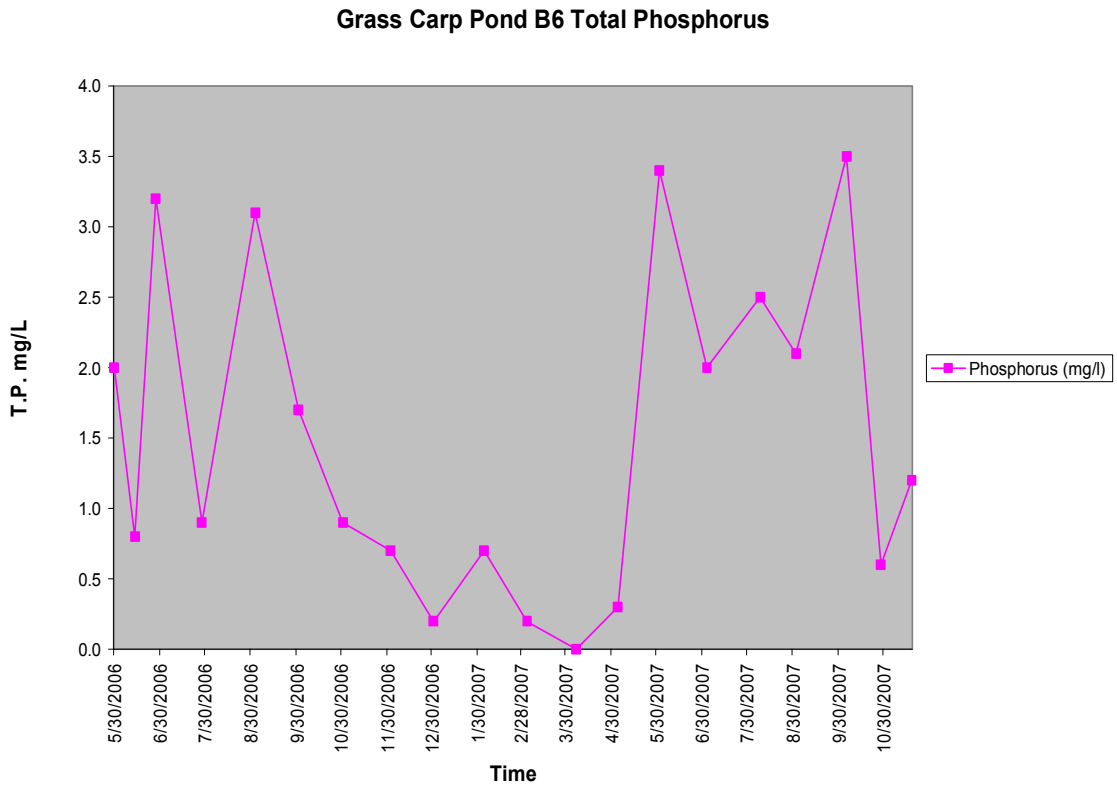

Figure A-19. Total phosphorus for grass carp pond B6

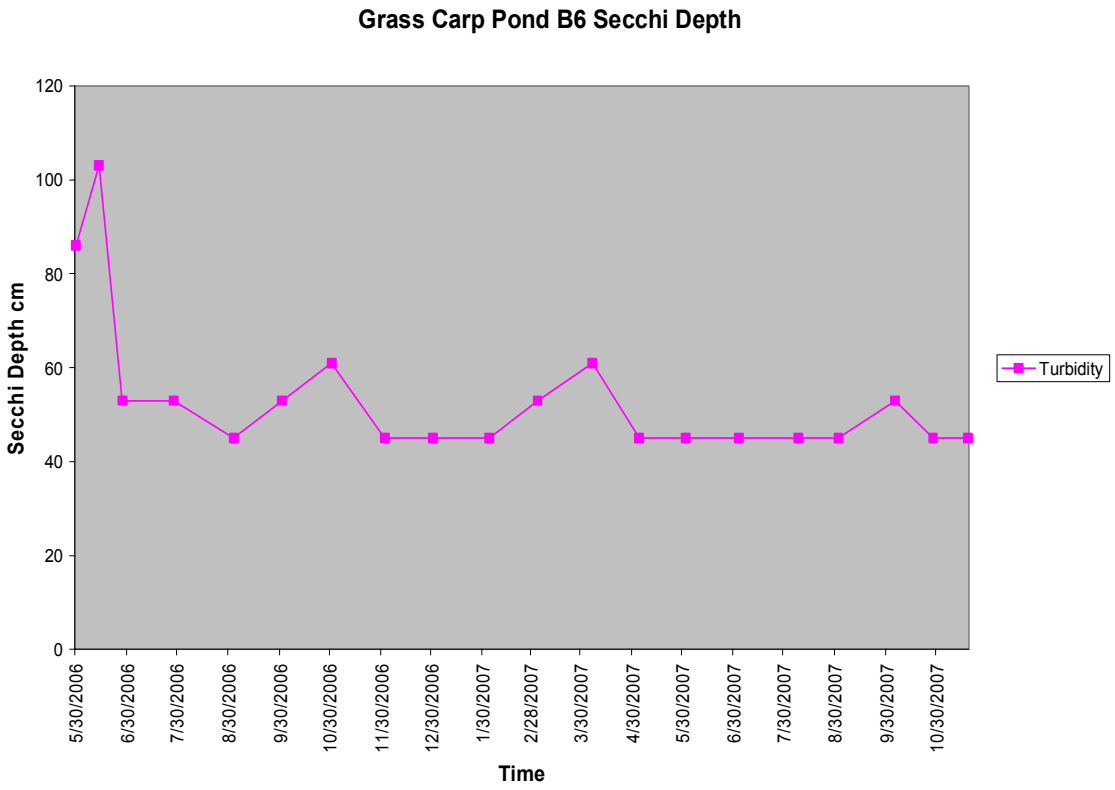

Figure A-20. Secchi depth for grass carp pond B6

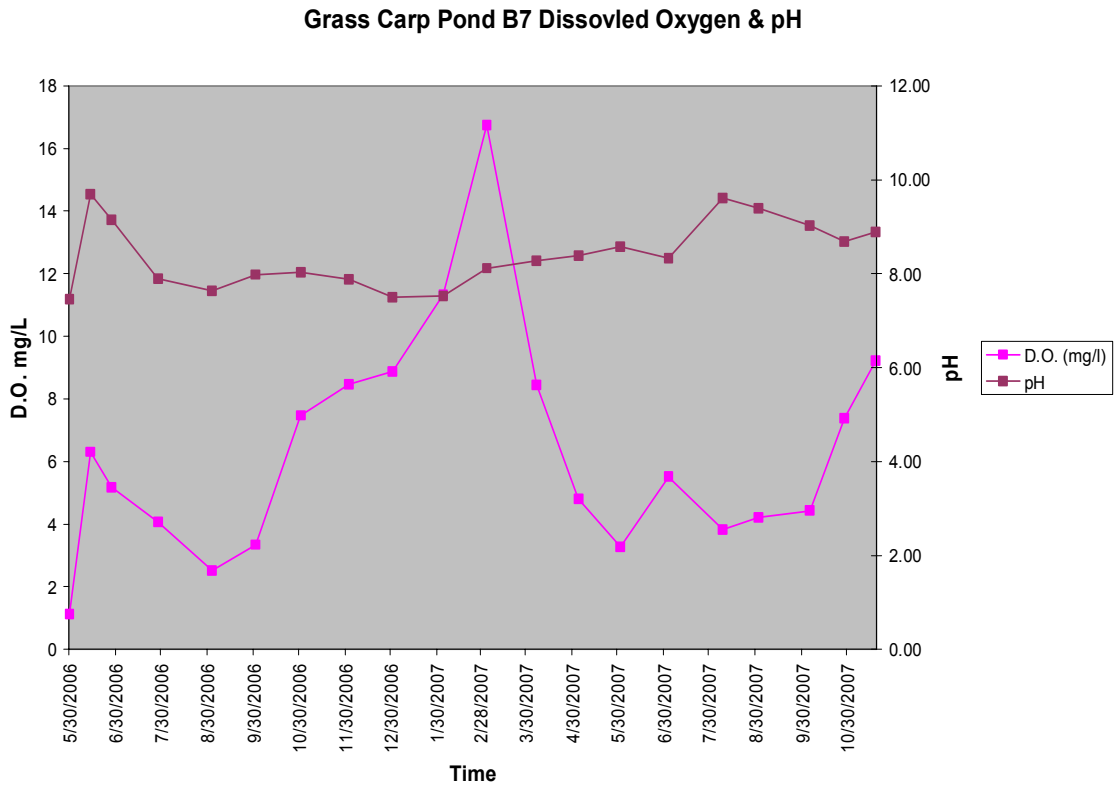

Figure A-21. Dissolved oxygen and pH for grass carp pond B7

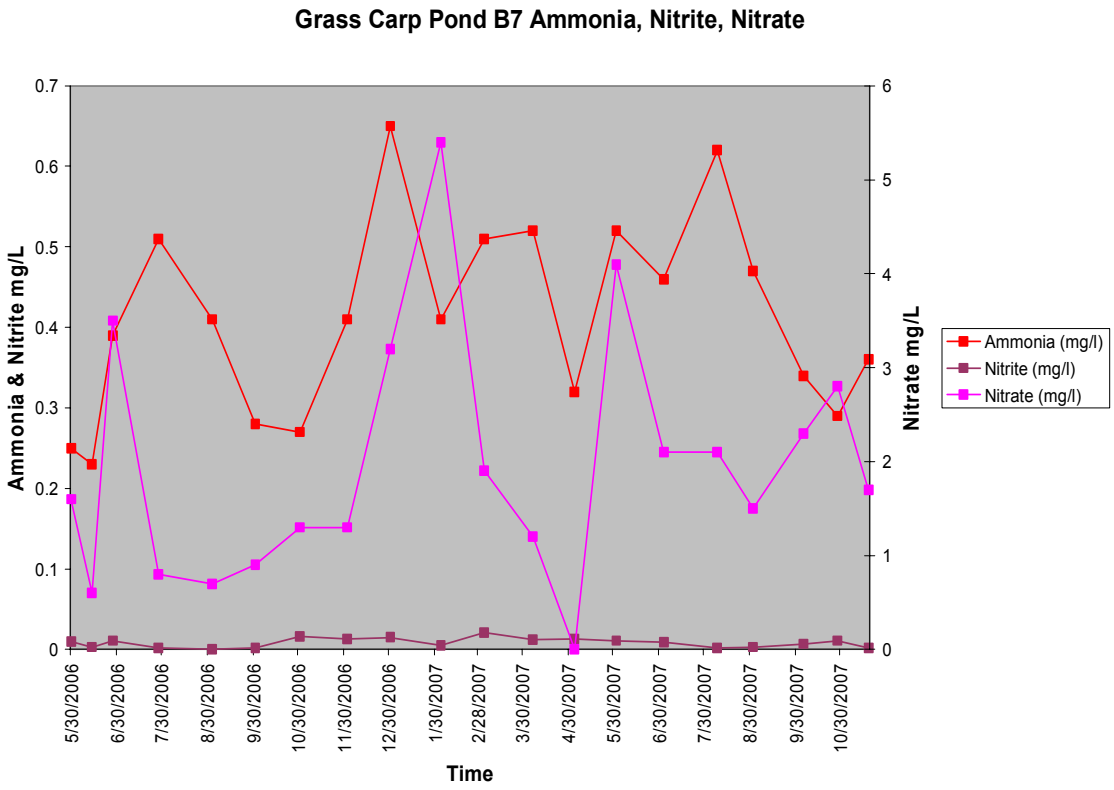

Figure A-22. Ammonia, nitrite, and nitrate for grass carp pond B7

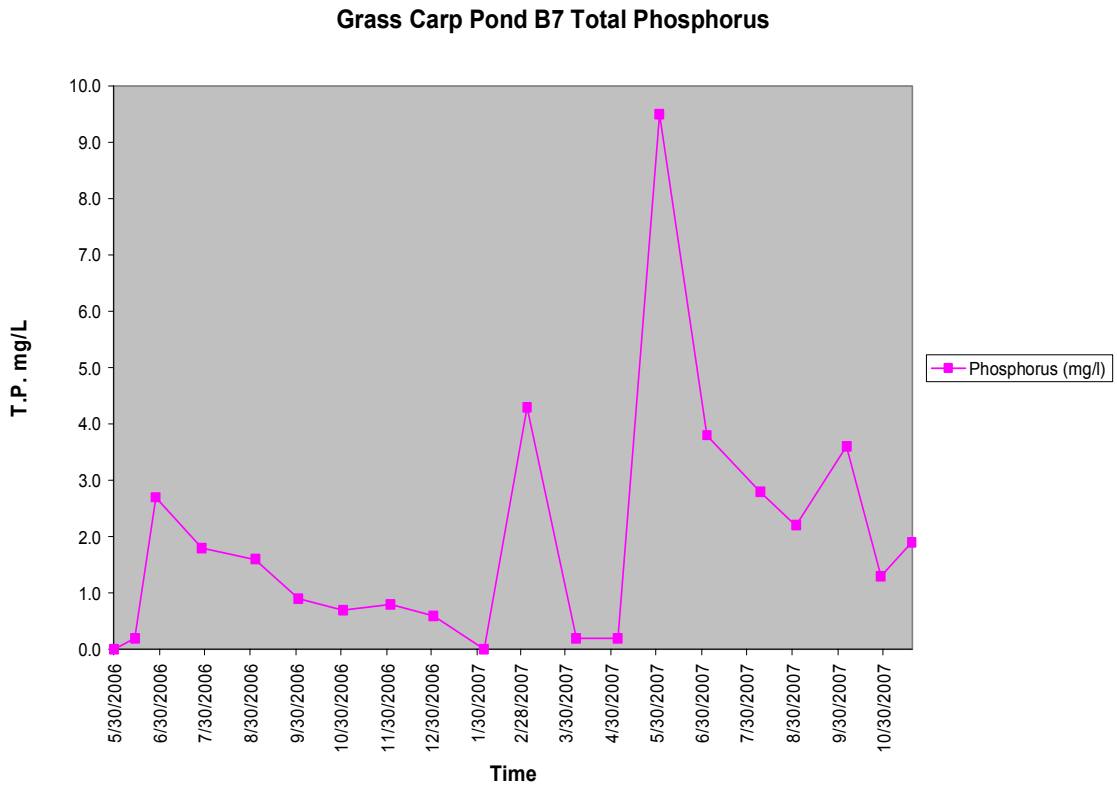

Figure A-23. Total phosphorus for grass carp pond B7

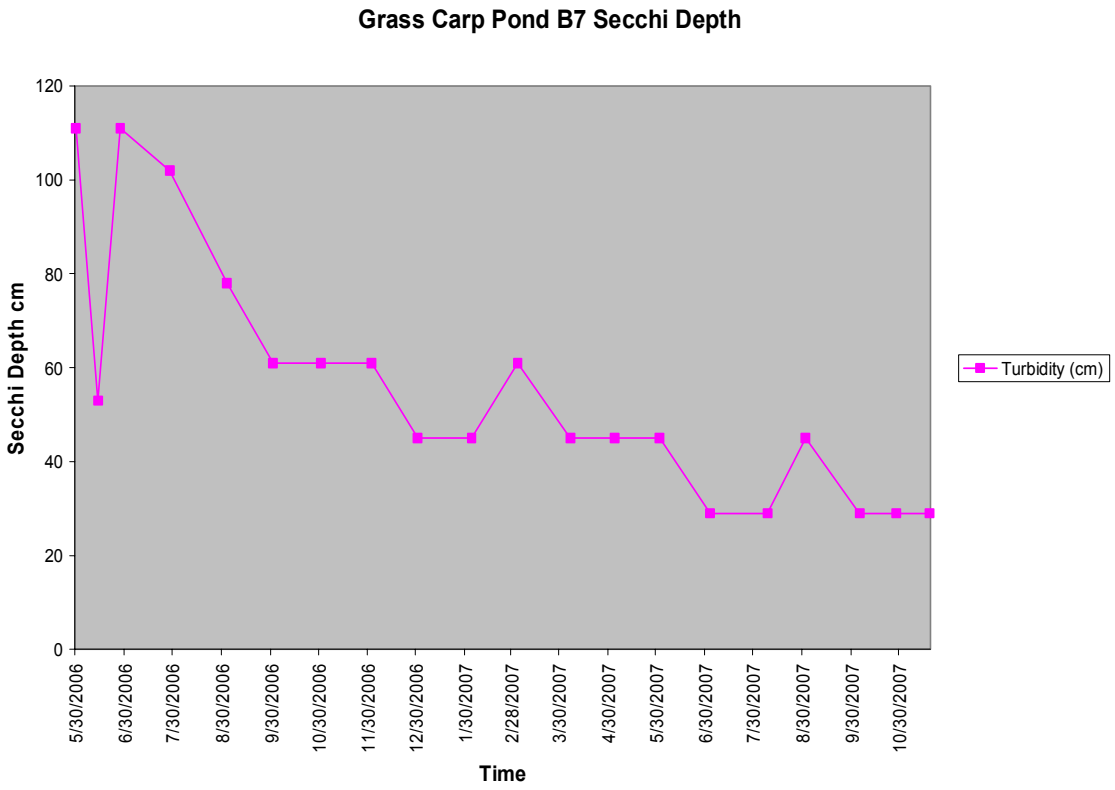

Figure A-24. Secchi depth for grass carp pond B7

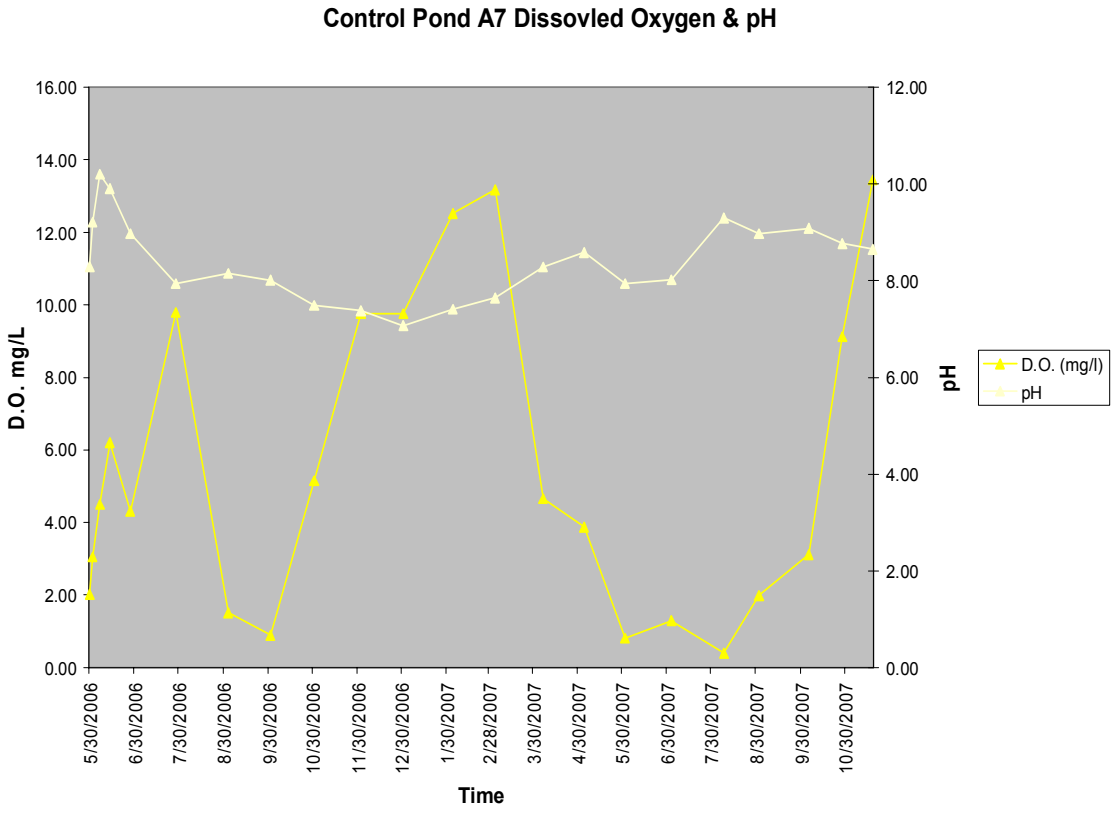

Figure A-25. Dissolved oxygen and pH for control pond A7

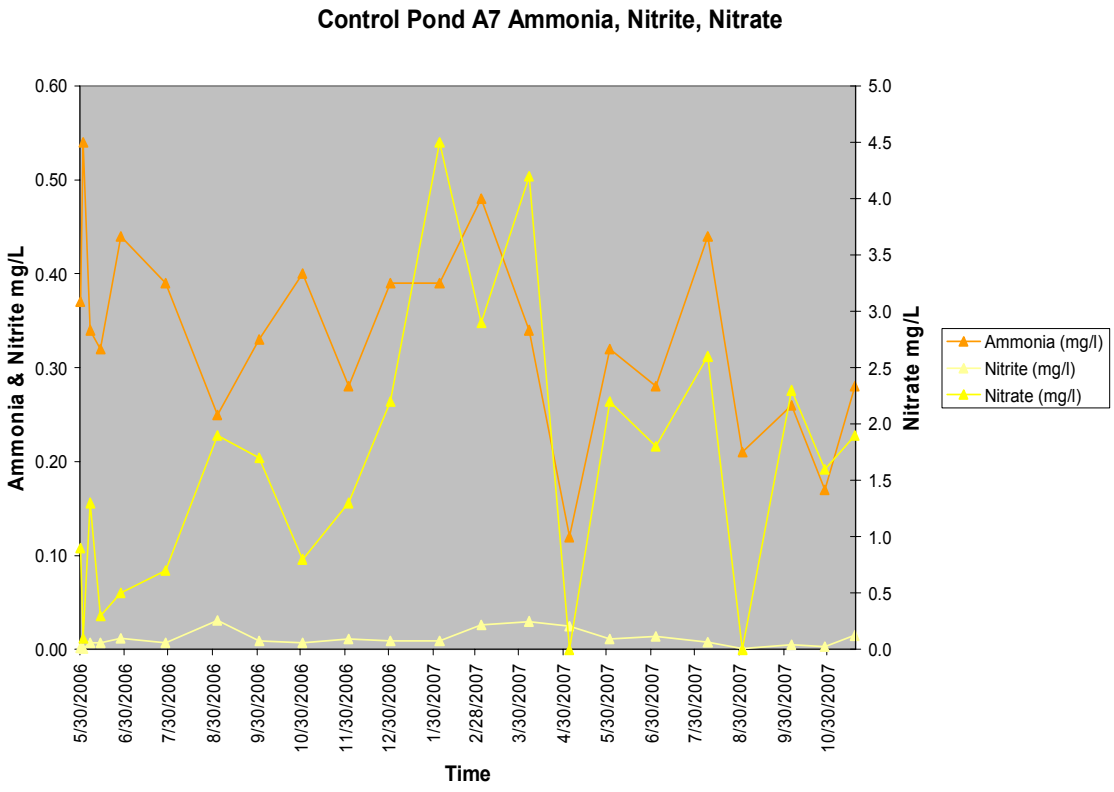

Figure A-26. Ammonia, nitrite, and nitrate for control pond A7

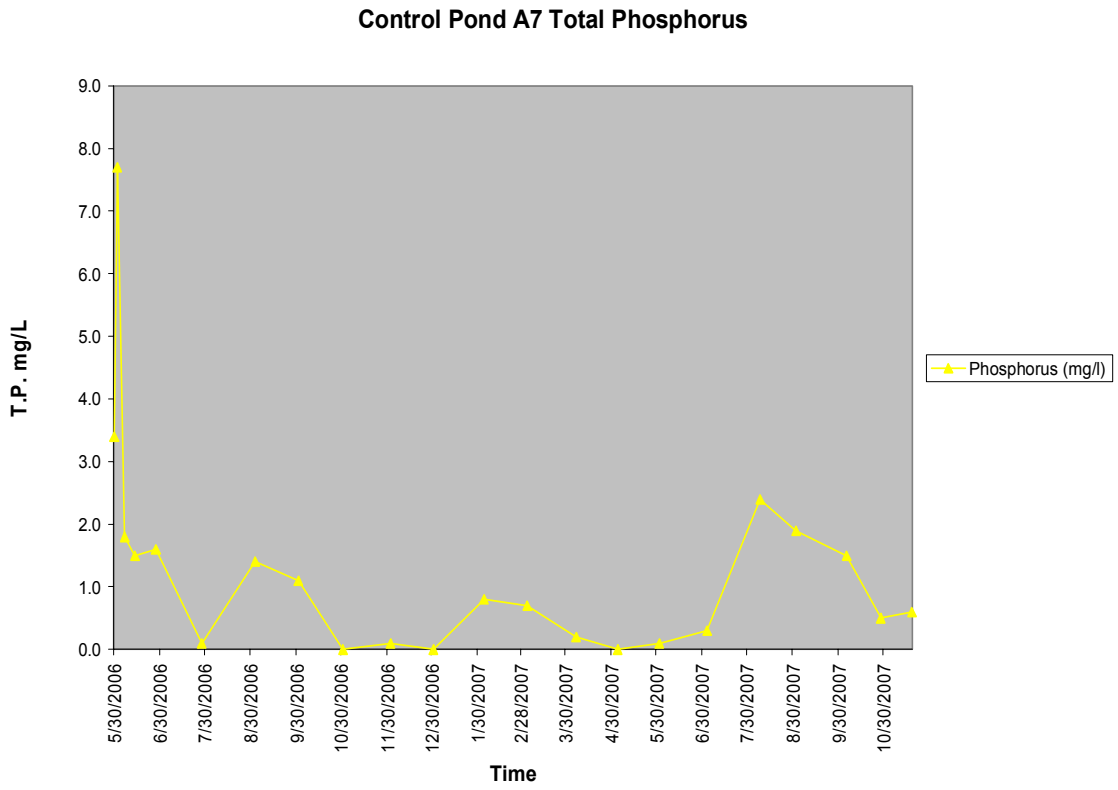

Figure A-27. Total phosphorus for control pond A7

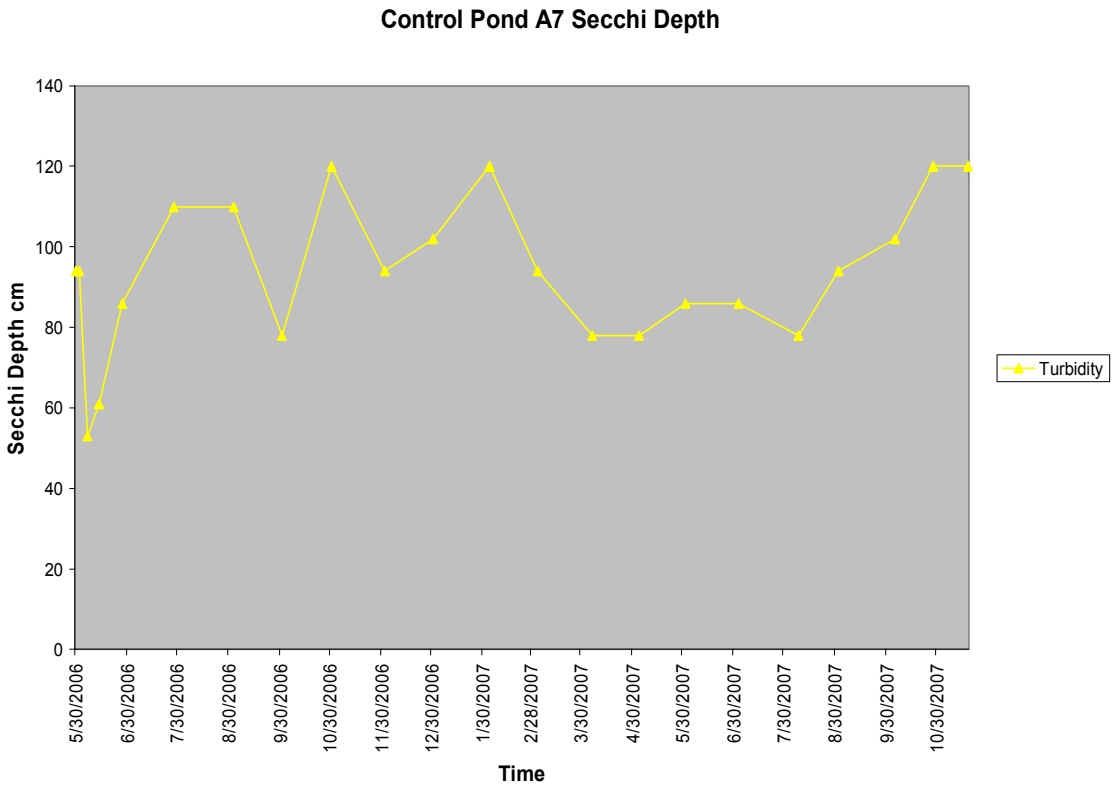

Figure A-28. Secchi depth for control pond A7

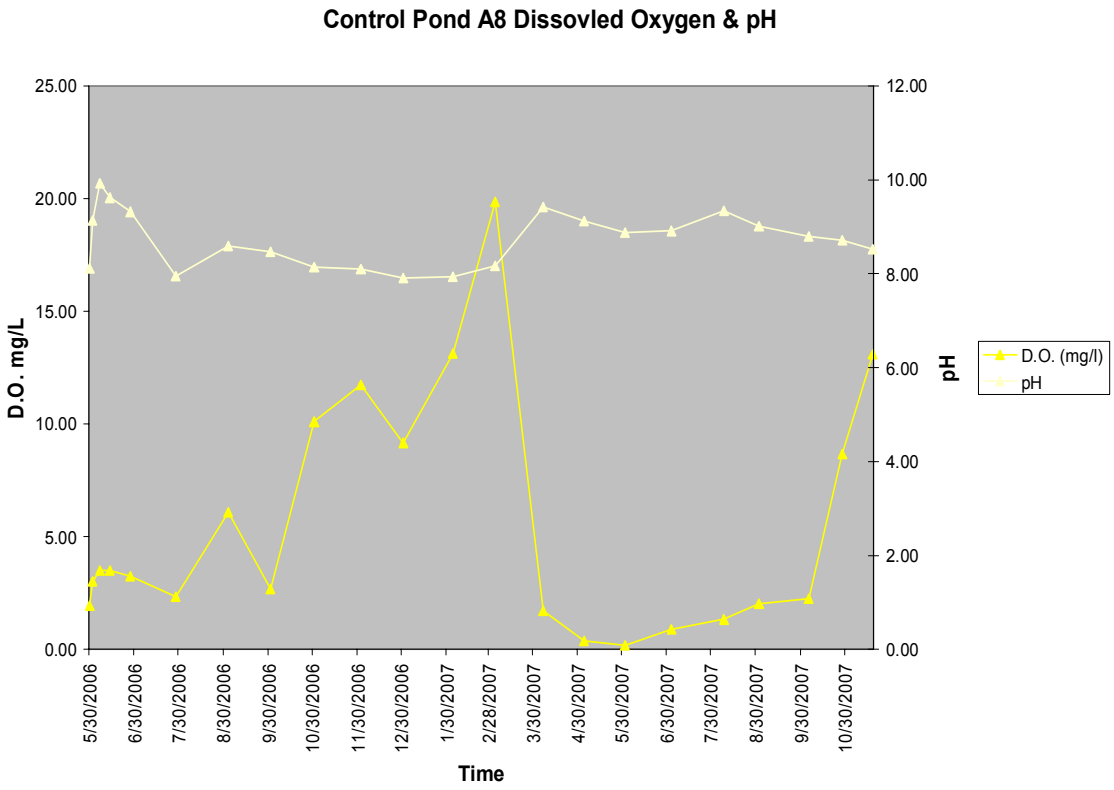

Figure A-29. Dissolved oxygen and pH for control pond A8

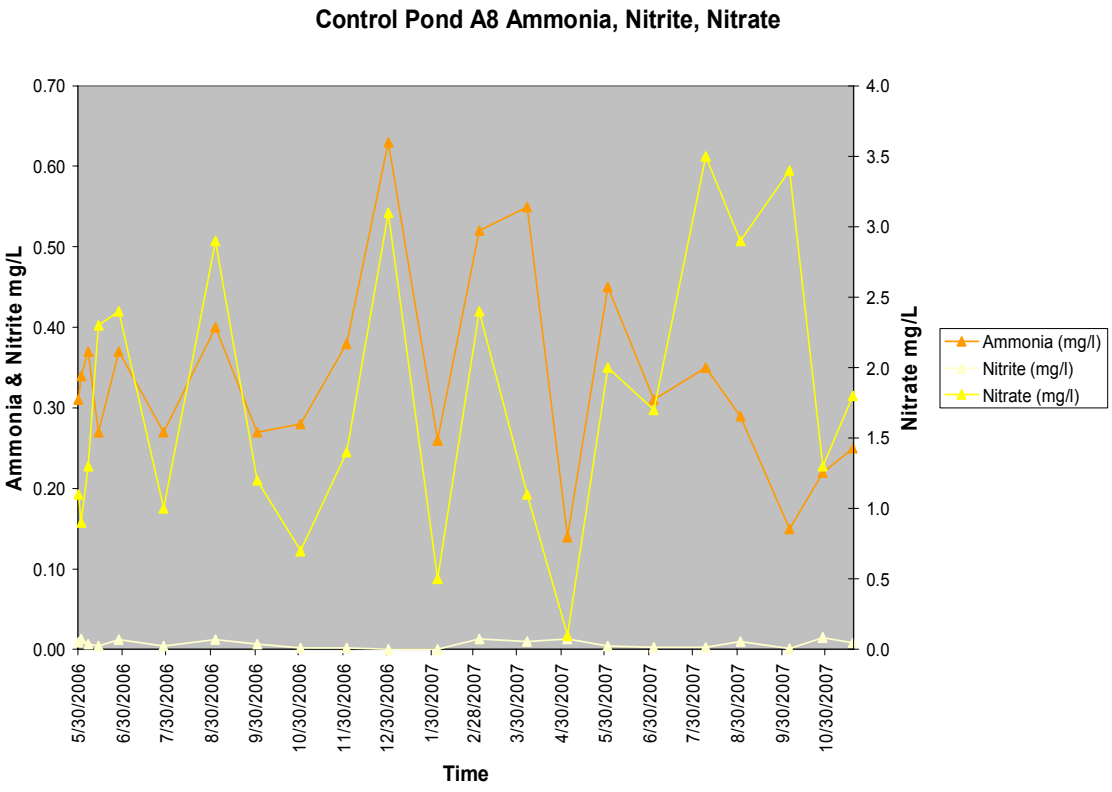

Figure A-30. Ammonia, nitrite, and nitrate for control pond A8

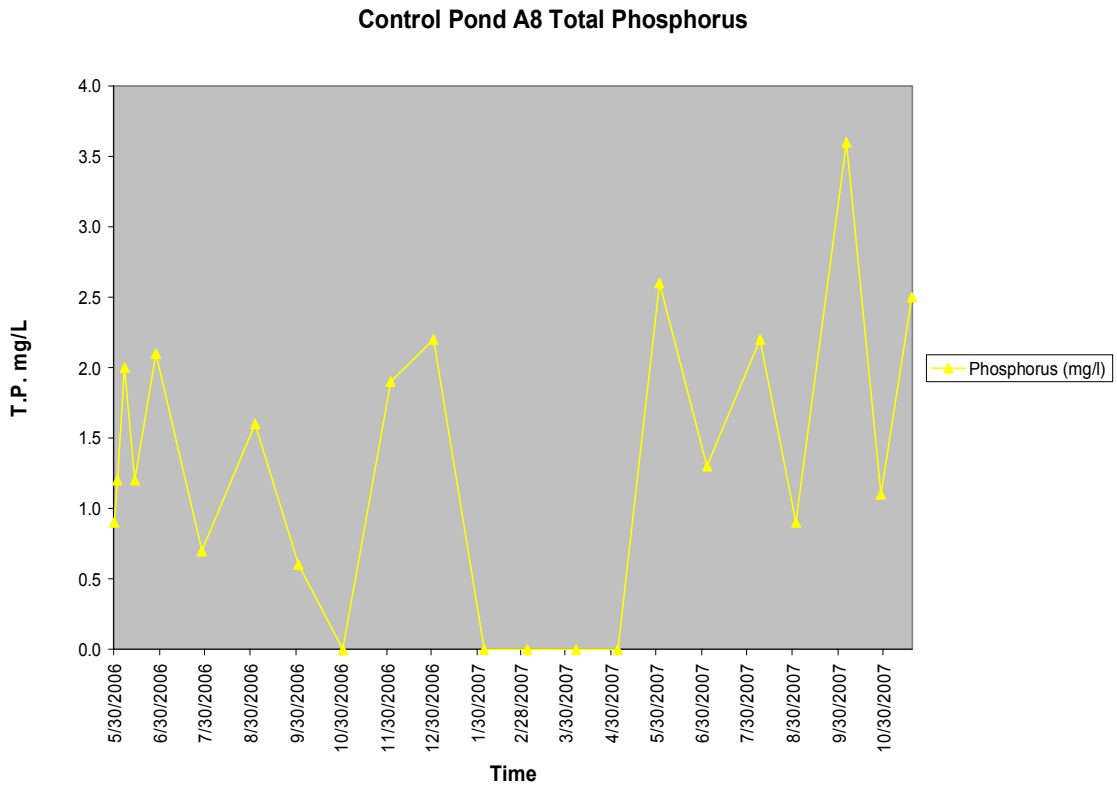

Figure A-31. Total phosphorus for control pond A8

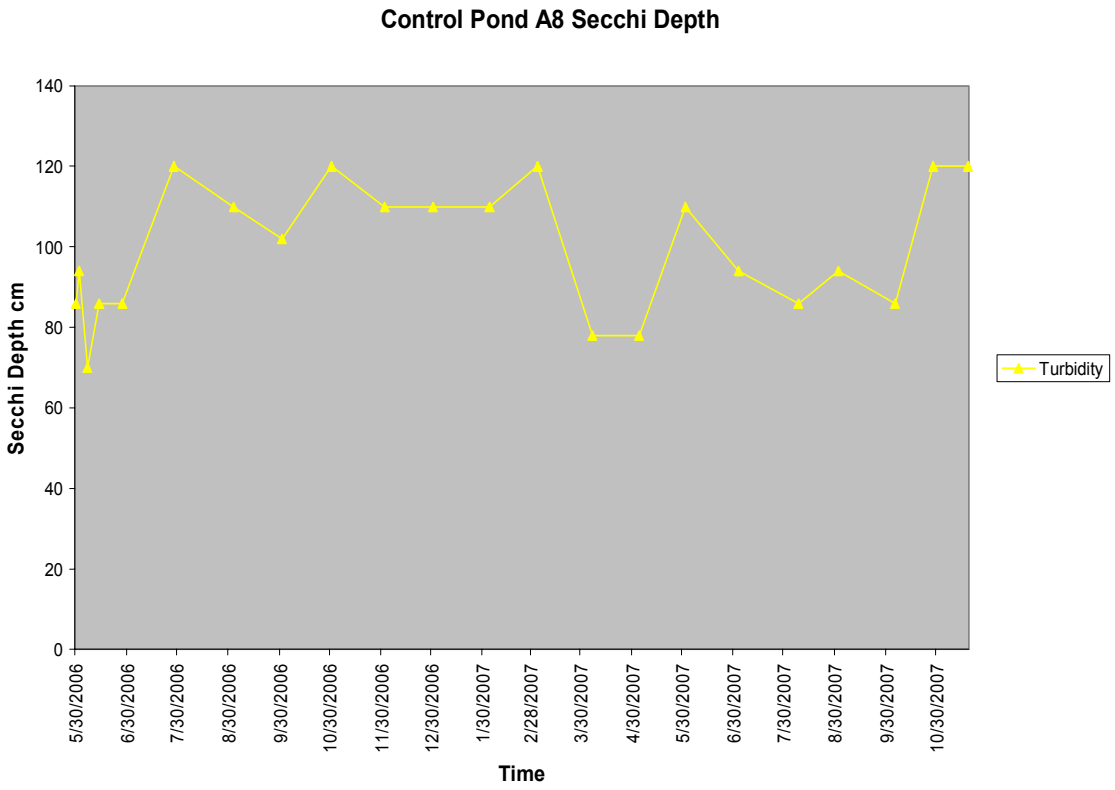

Figure A-32. Secchi depth for control pond A8

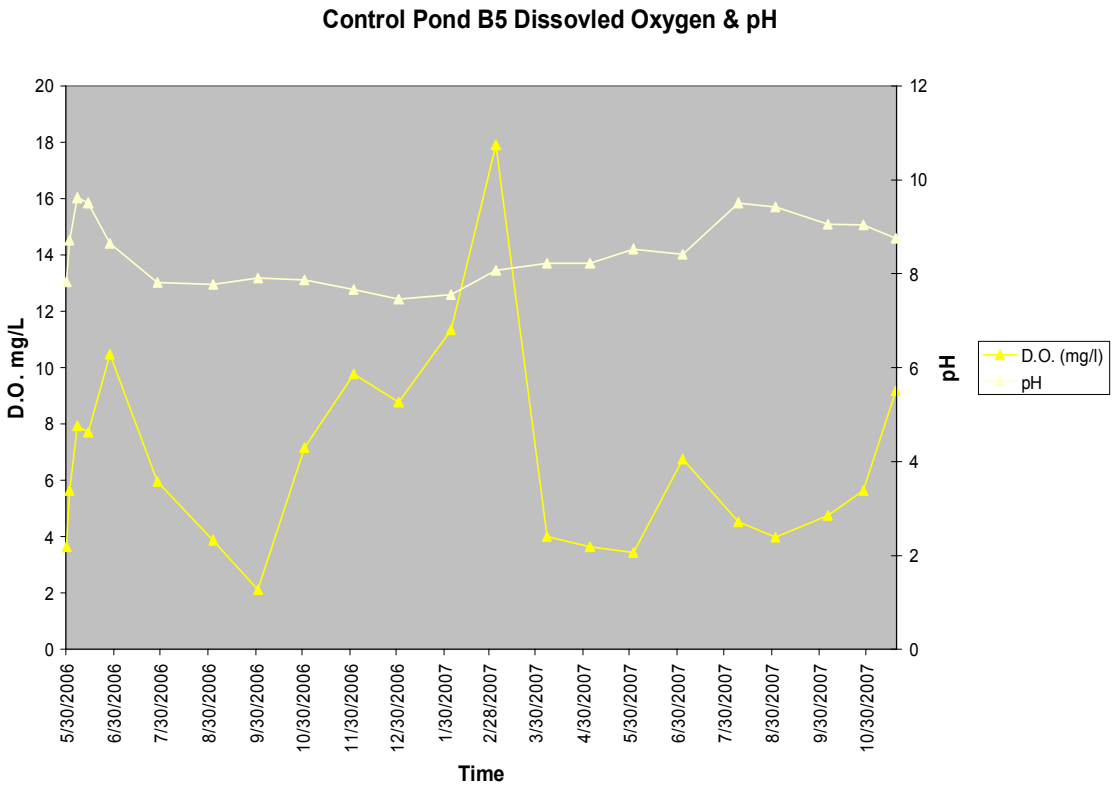

Figure A-33. Dissolved oxygen and pH for control pond B5

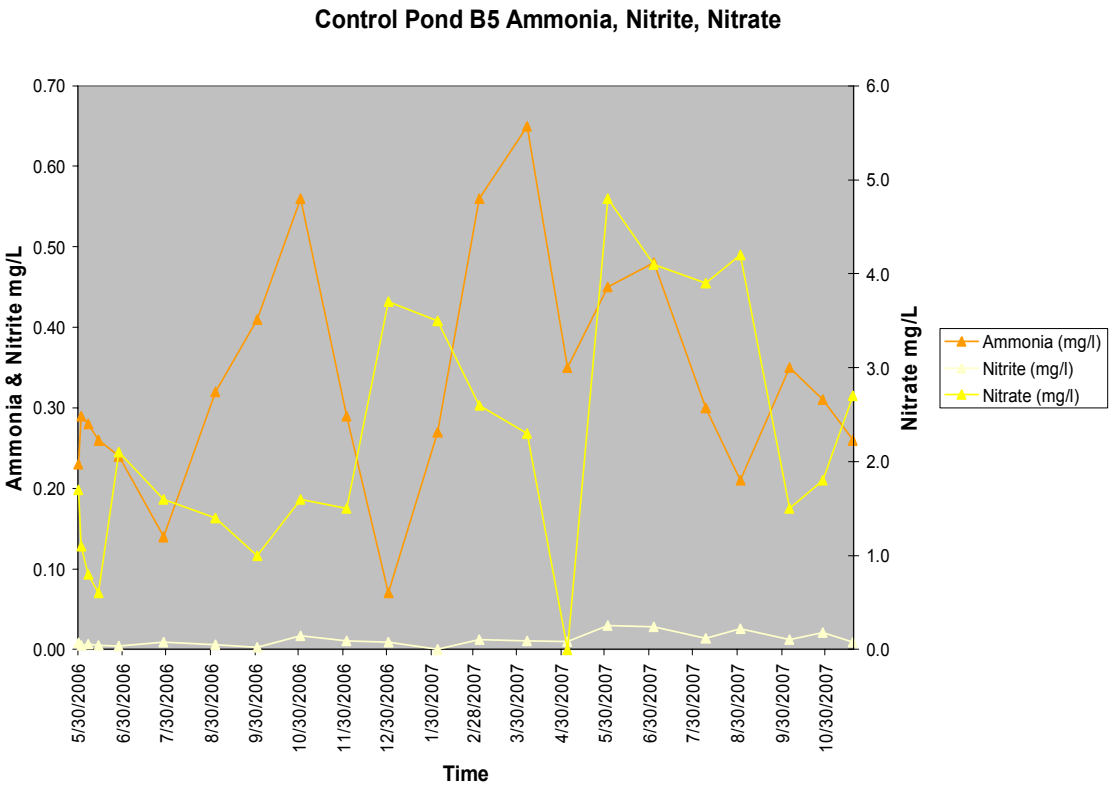

Figure A-34. Ammonia, nitrite, and nitrate for control pond B5

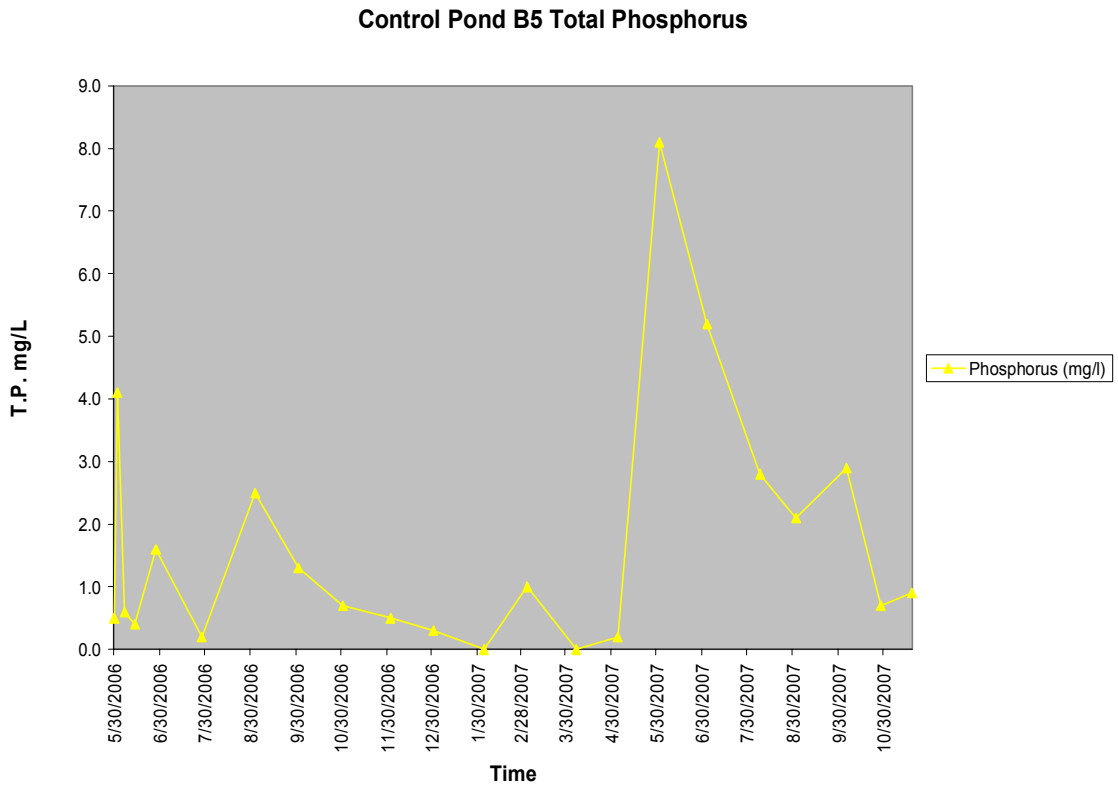

Figure A-35. Total phosphorus for control pond B5

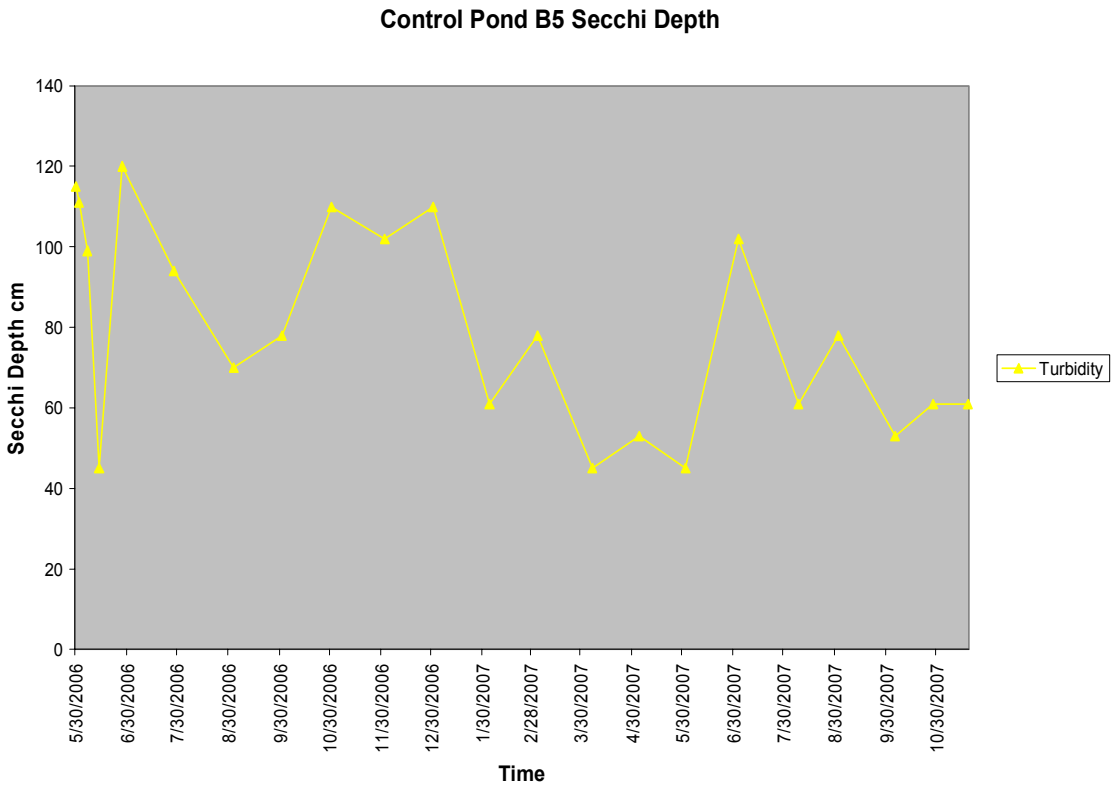

Figure A-36. Secchi depth for control pond B5

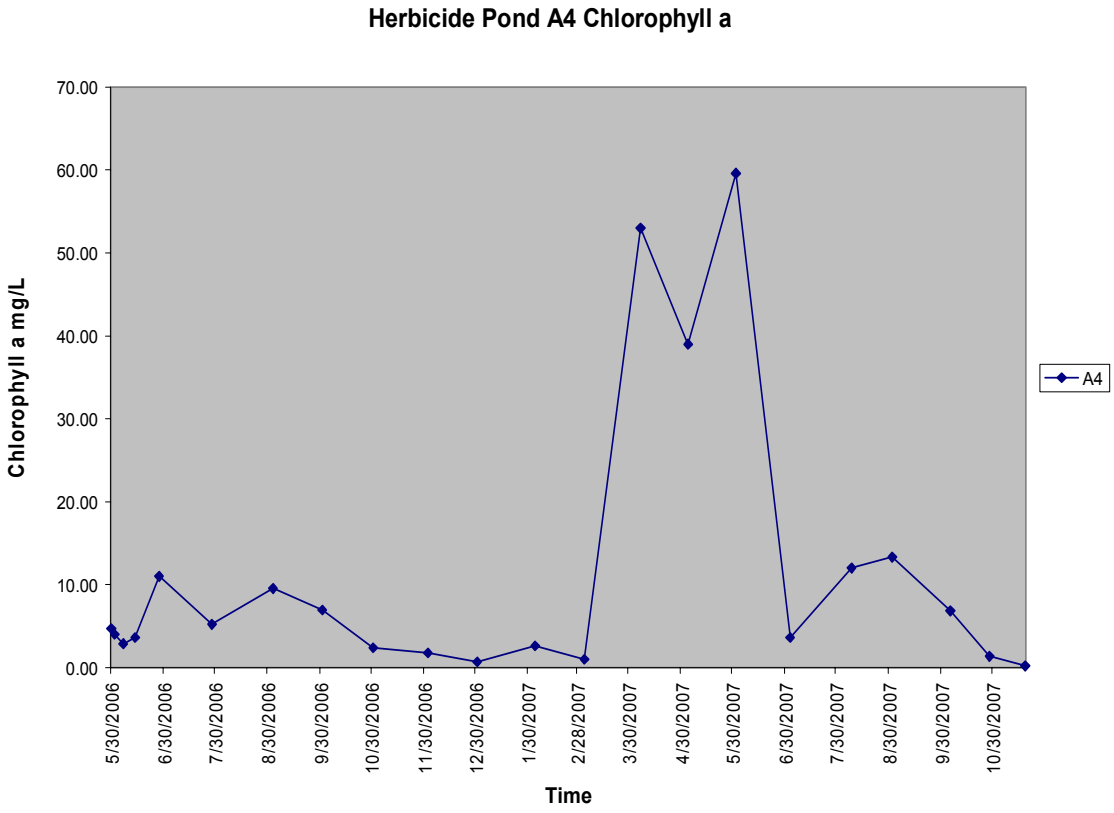

Figure A-37. Chlorophyll *a* for herbicide pond A4

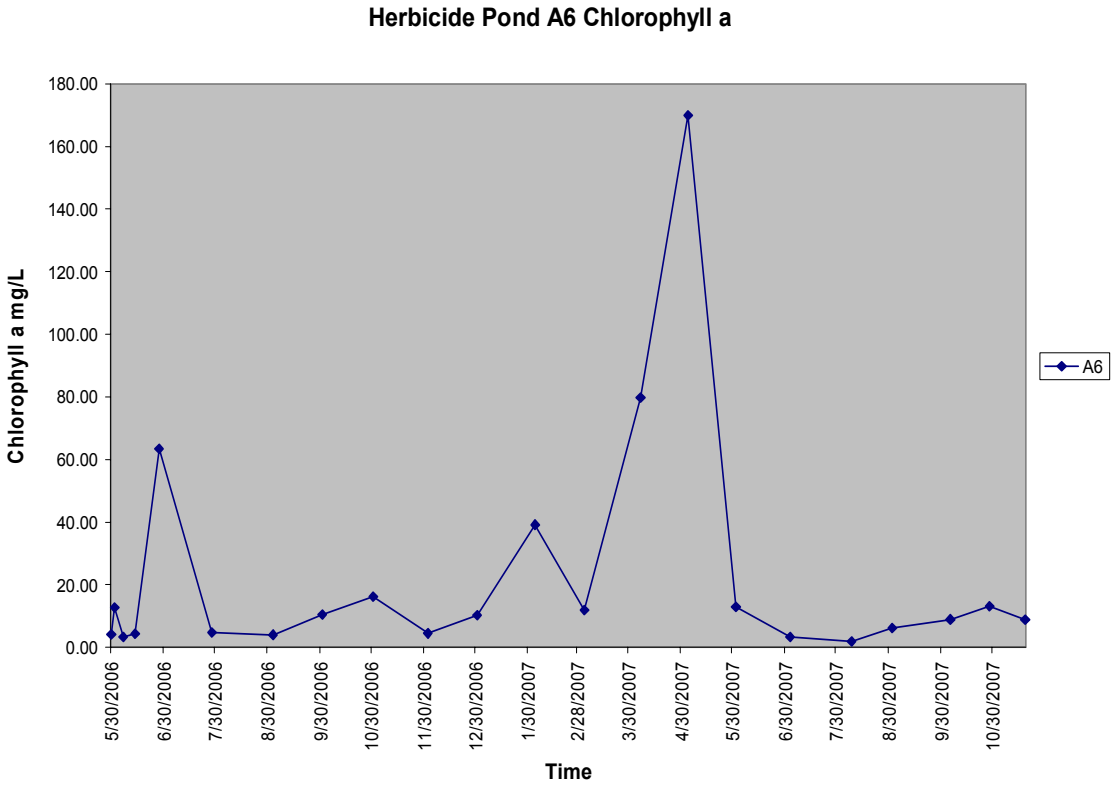

Figure A-38. Chlorophyll *a* for herbicide pond A6

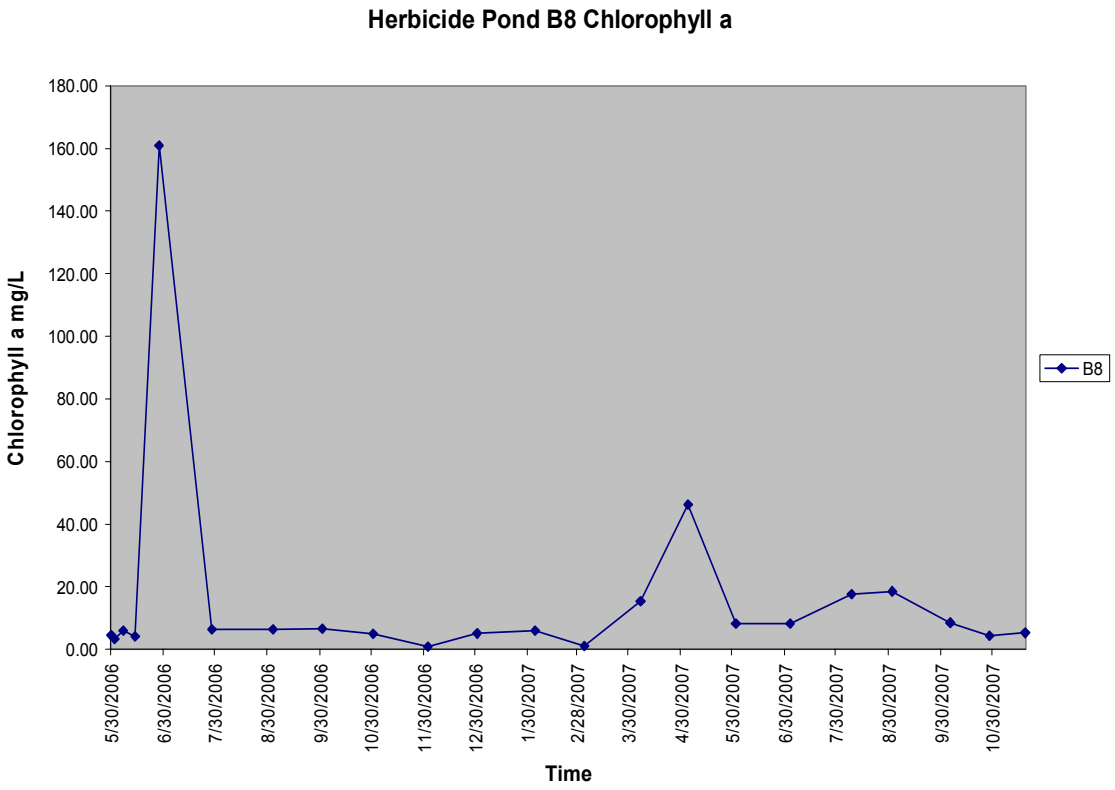

Figure A-39. Chlorophyll *a* for herbicide pond B8

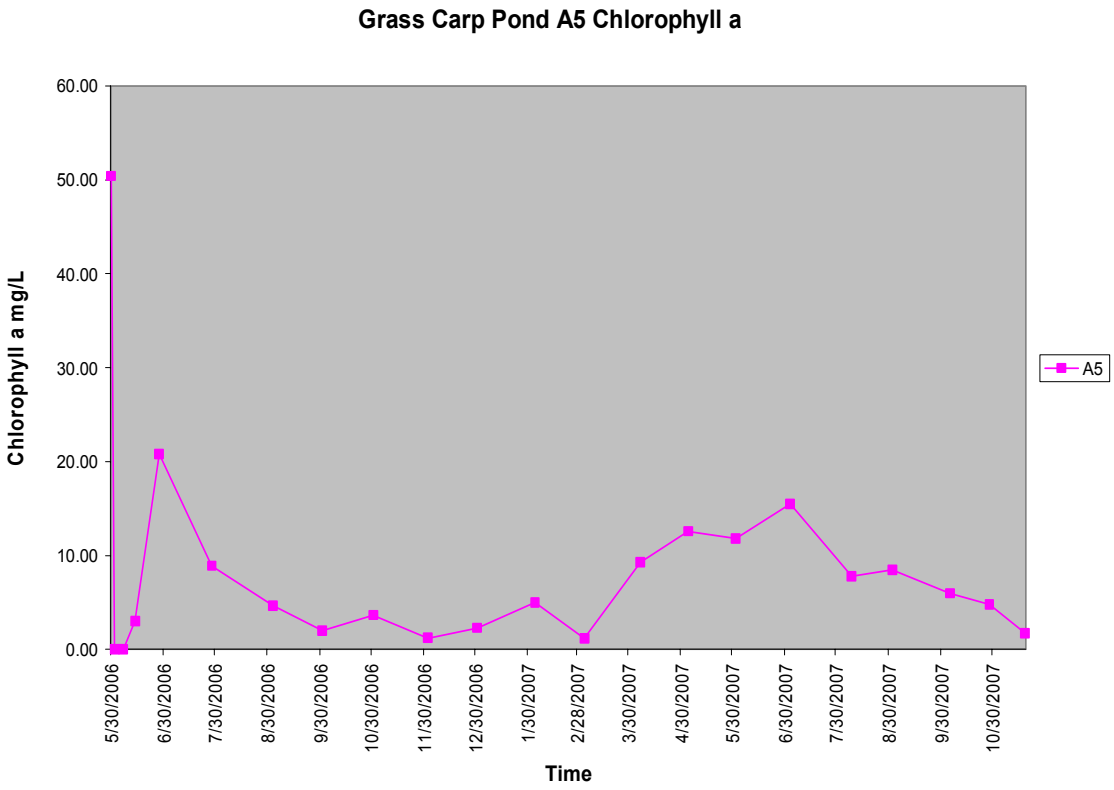

Figure A-40. Chlorophyll *a* for grass carp pond A5

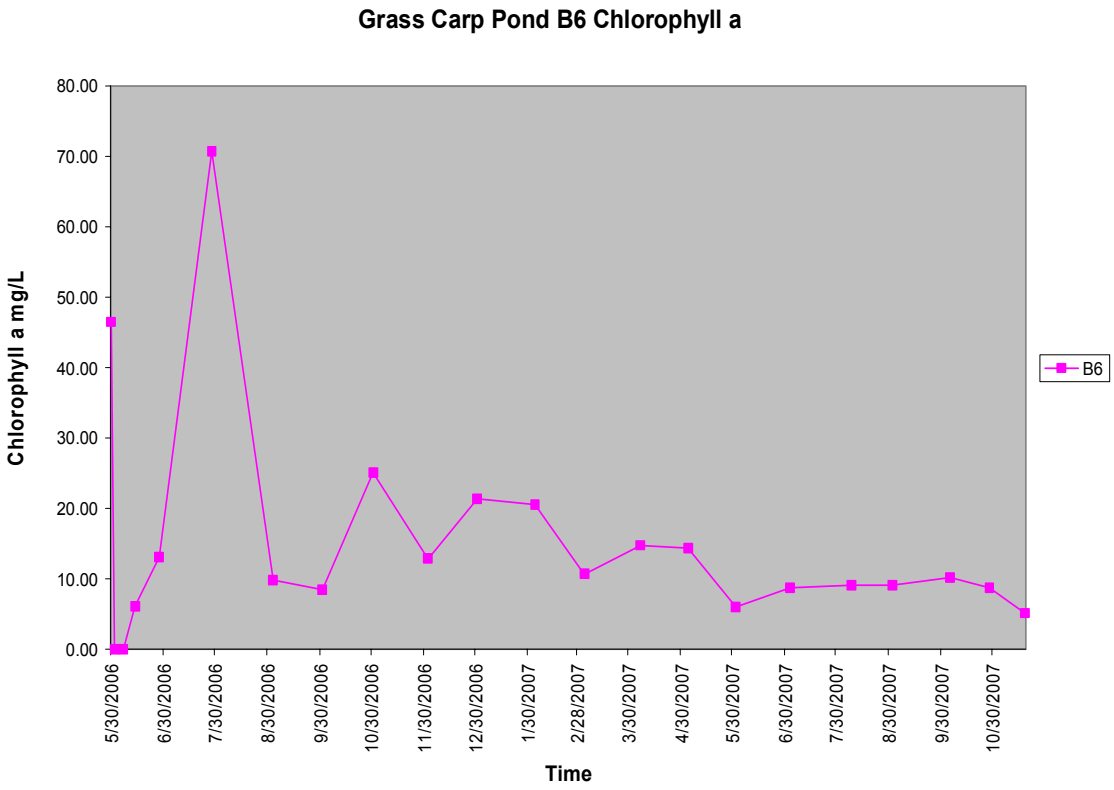

Figure A-41. Chlorophyll a for grass carp pond B6

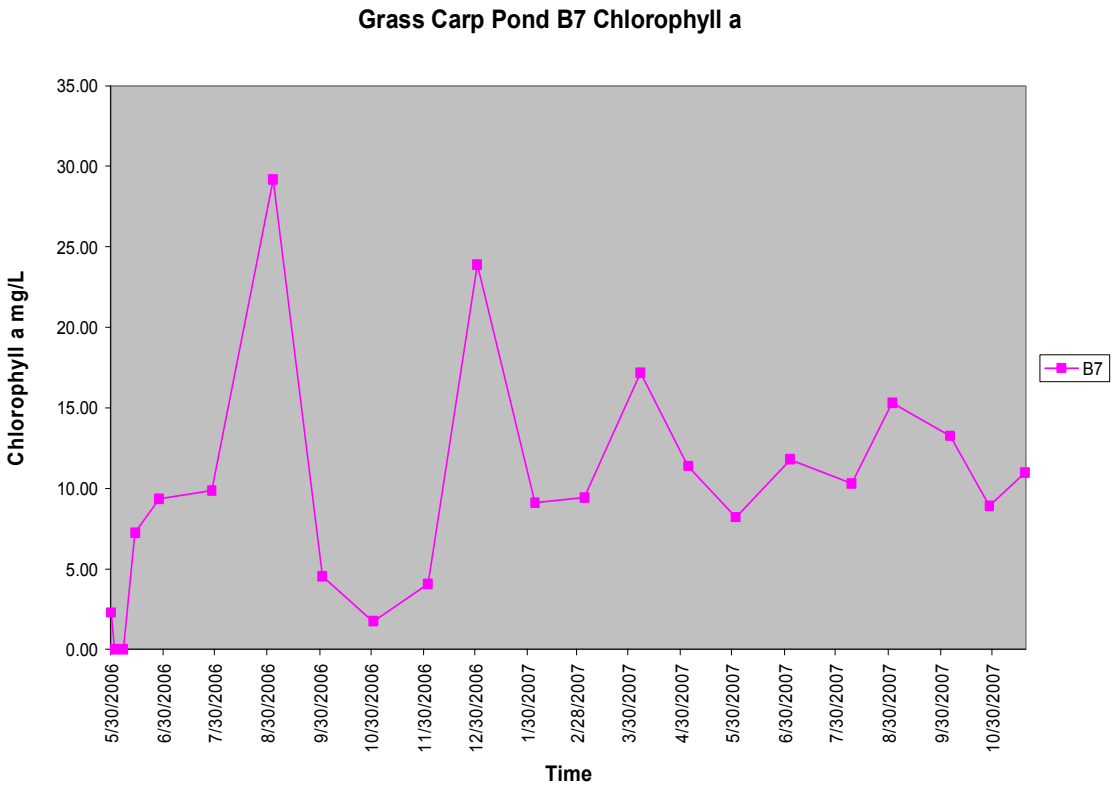

Figure A-42. Chlorophyll *a* for grass carp pond B7

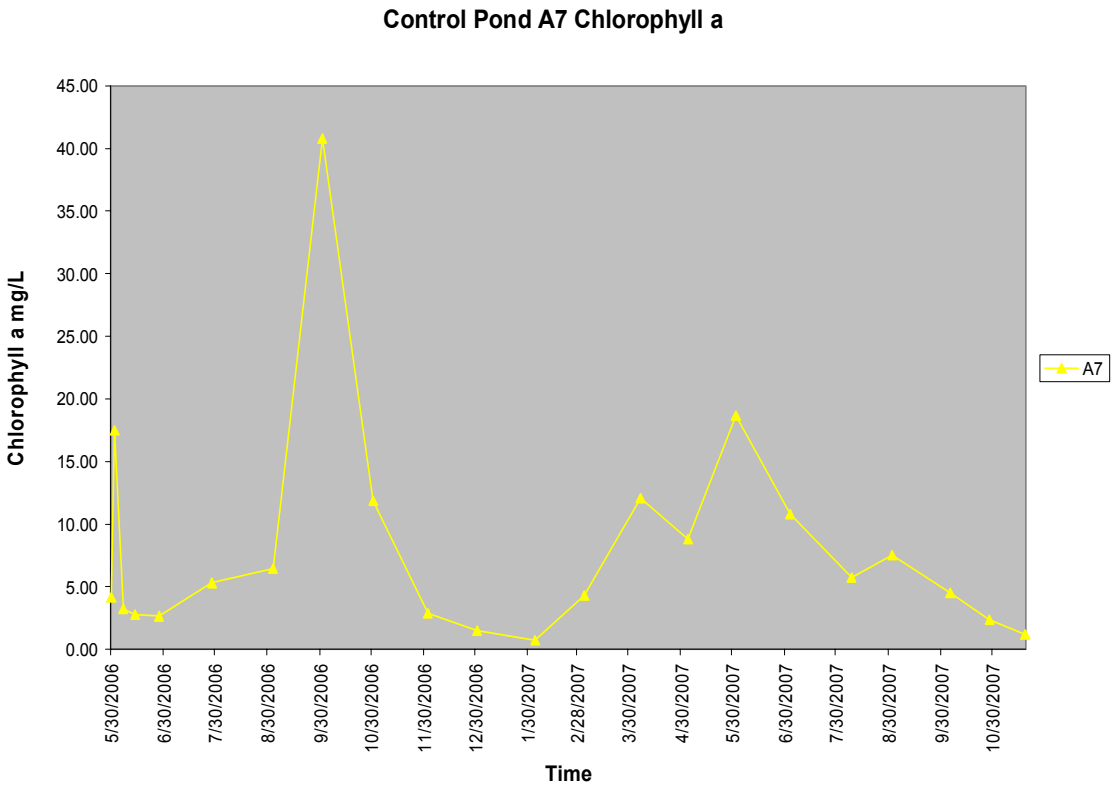

Figure A-43. Chlorophyll *a* for control pond A7

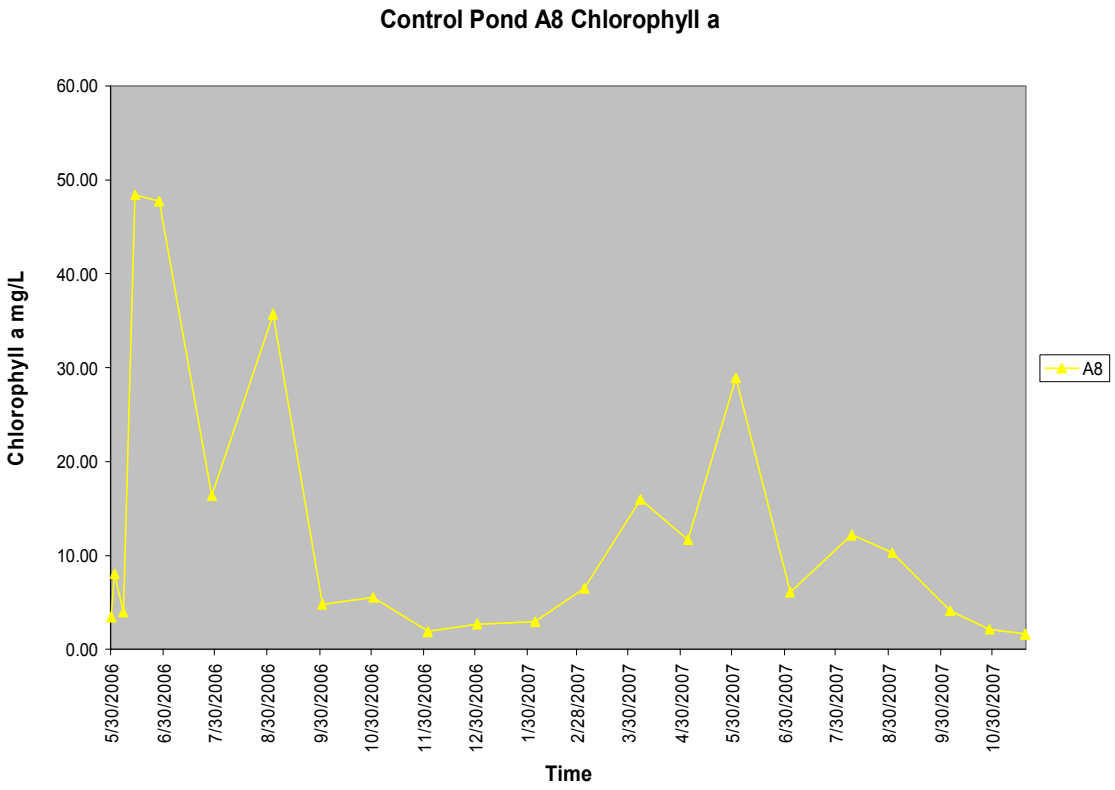

Figure A-44. Chlorophyll *a* for control pond A8

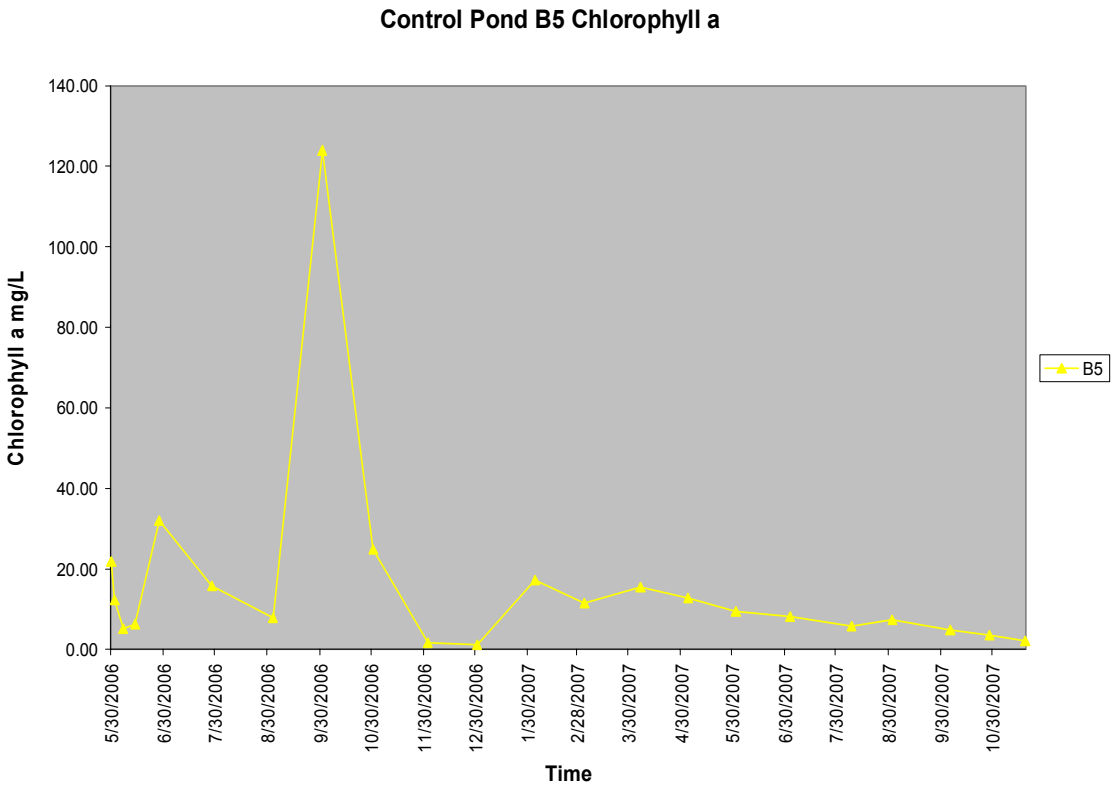

Figure A-45. Chlorophyll *a* for control pond B5

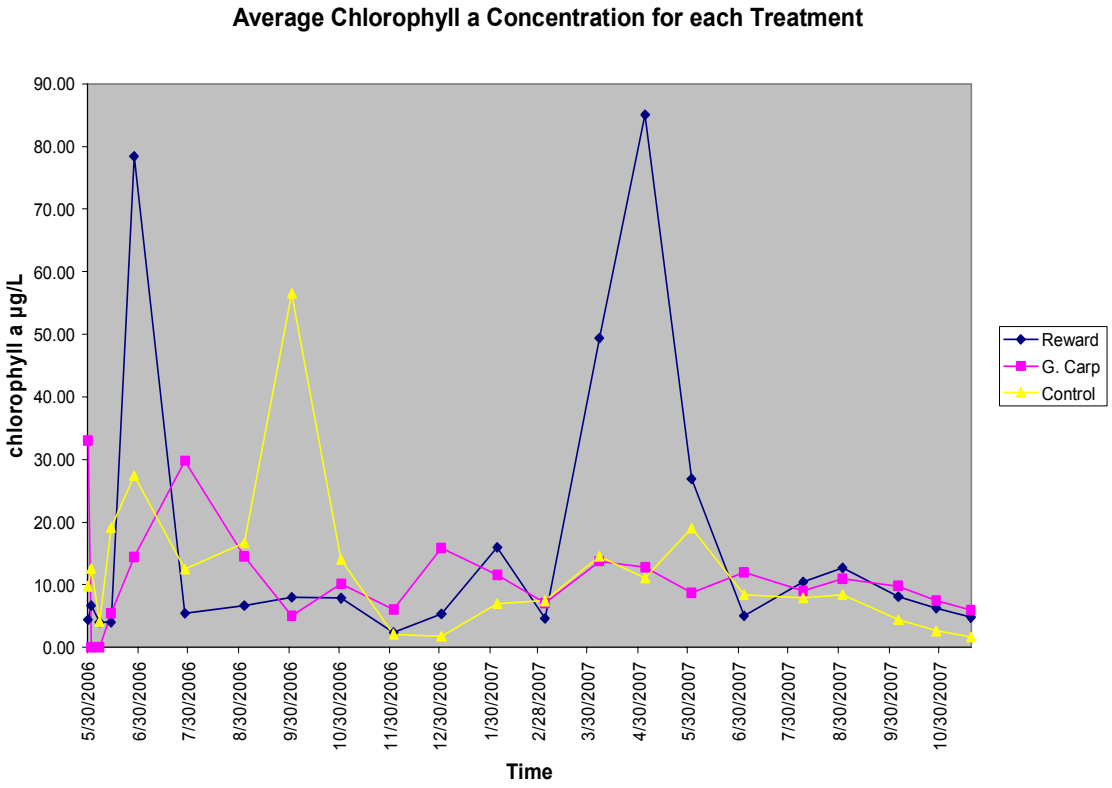

Figure A-46. Average chlorophyll *a* concentration for all treatments

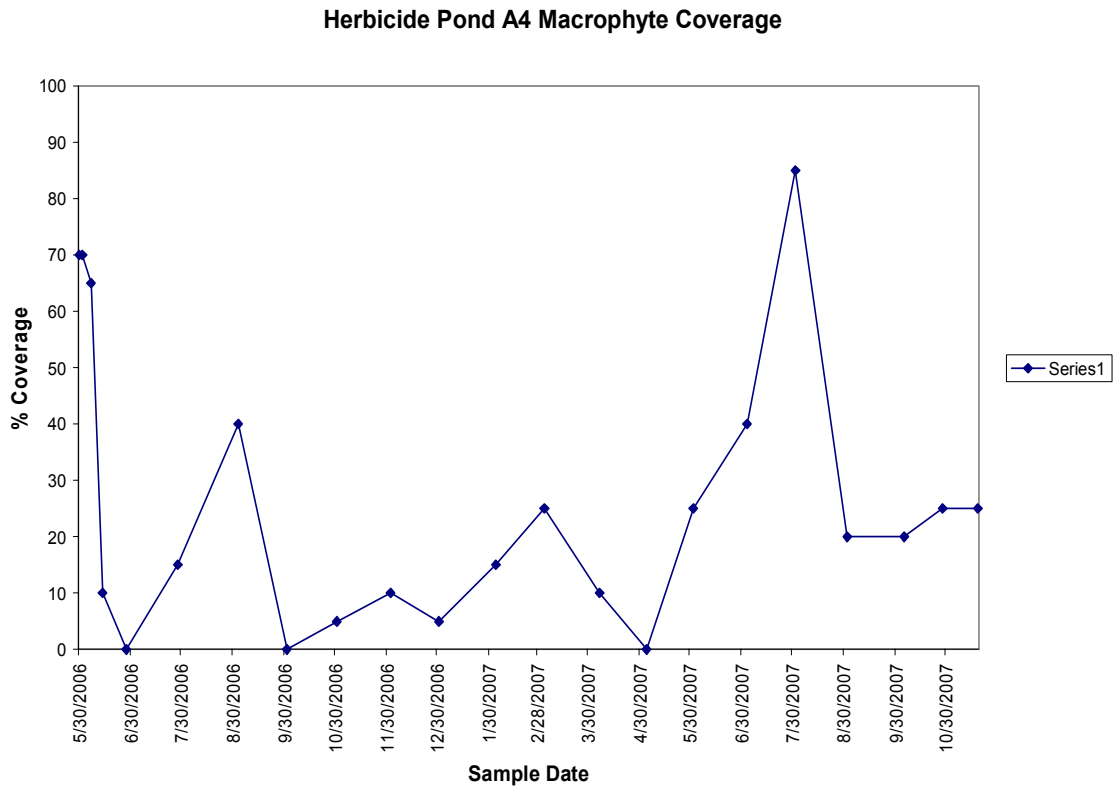

Figure A-47. Macrophyte coverage for herbicide pond A4

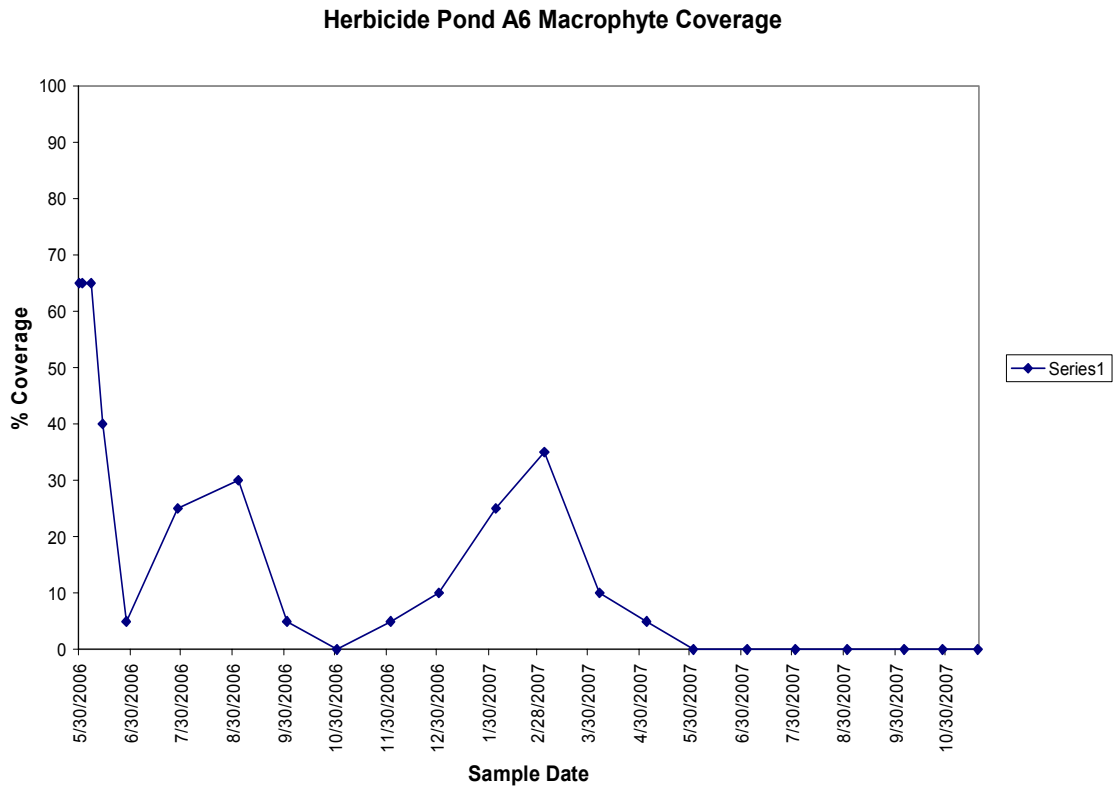

Figure A-48. Macrophyte coverage for herbicide pond A6

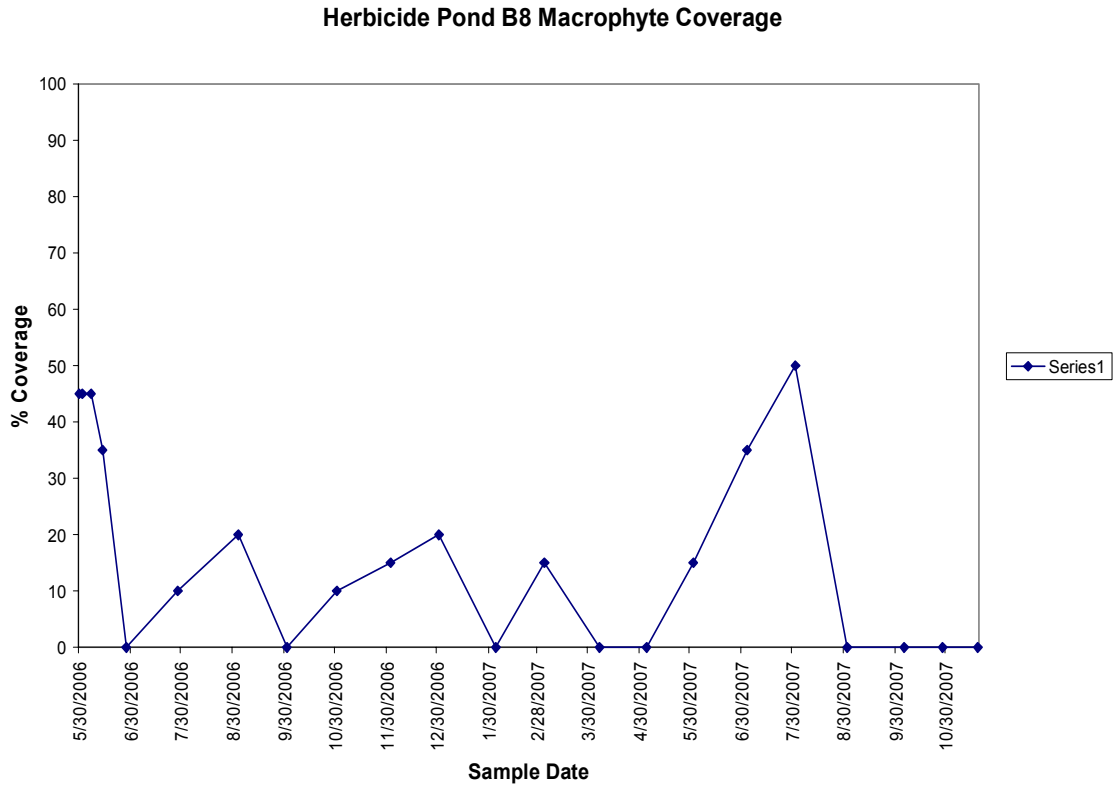

Figure A-49. Macrophyte coverage for herbicide pond B8

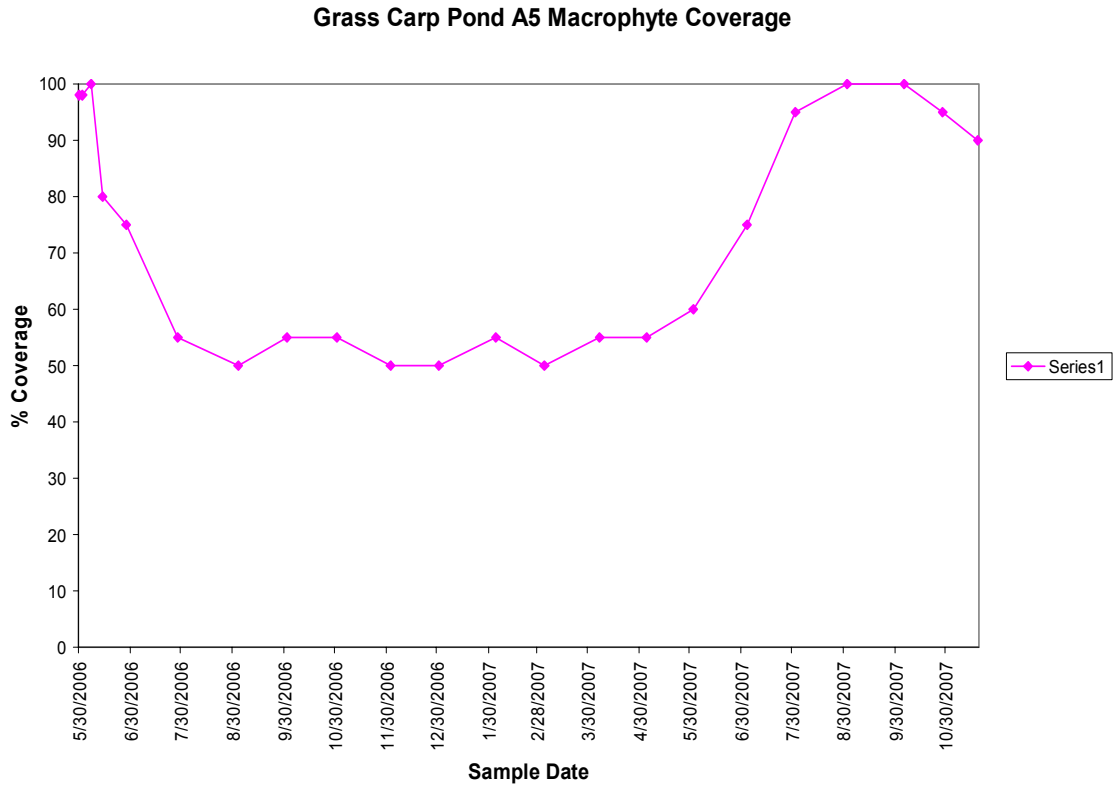

Figure A-50. Macrophyte coverage for grass carp pond A5

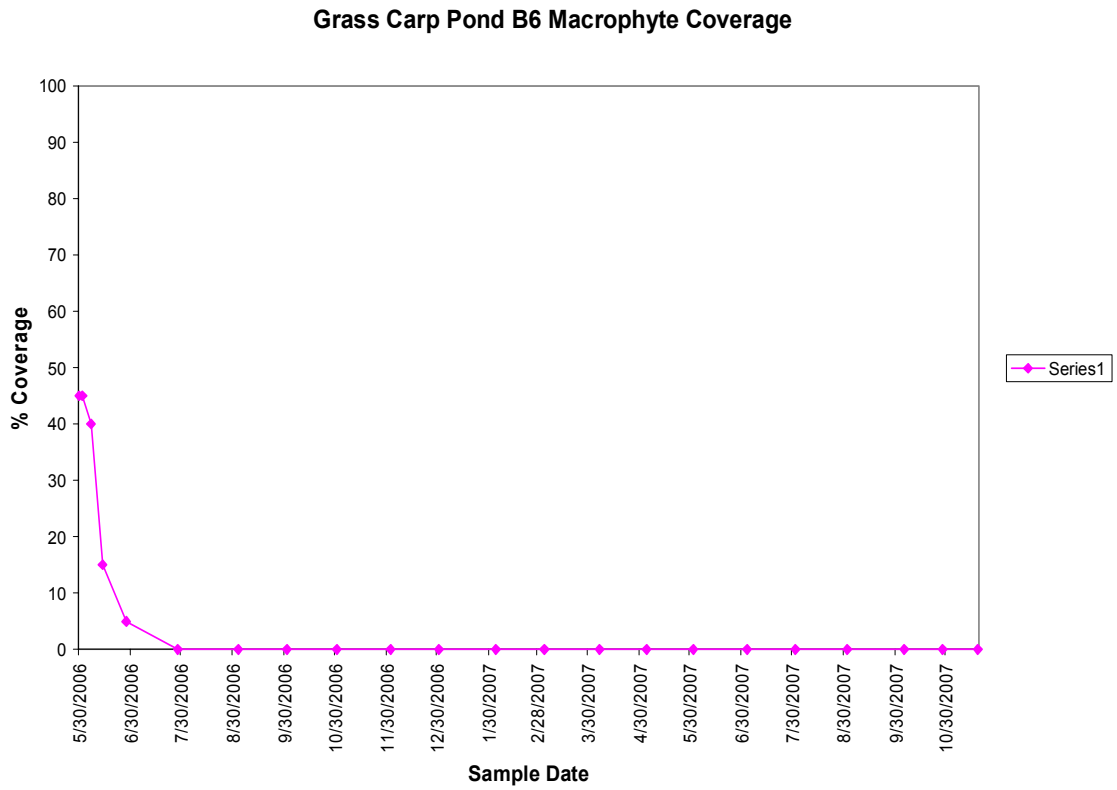

Figure A-51. Macrophyte coverage for grass carp pond B6

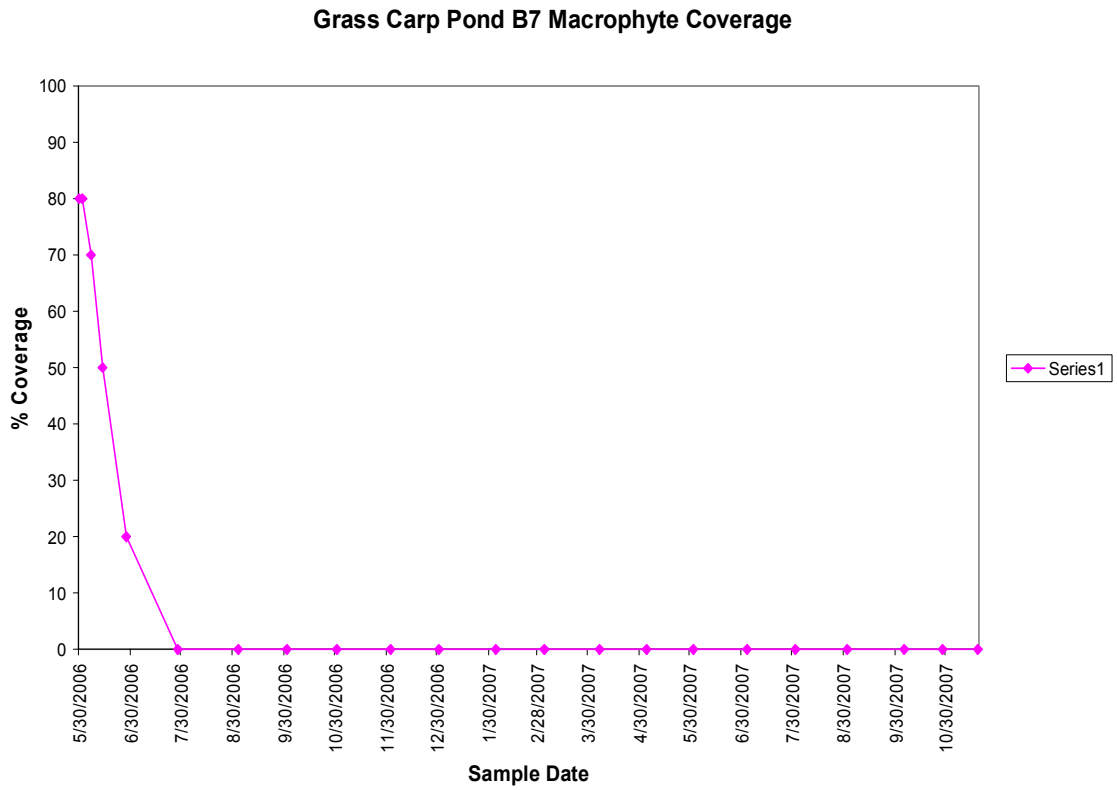

Figure A-52. Macrophyte coverage for grass carp pond B7

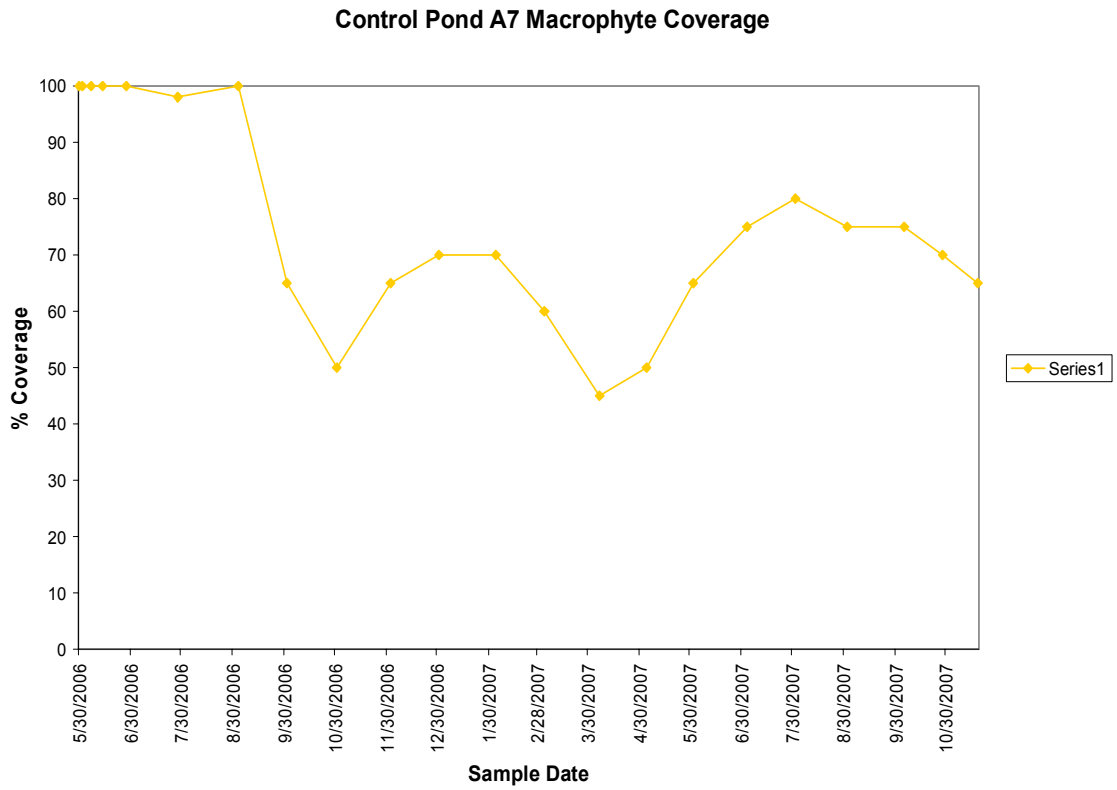

Figure A-53. Macropyhte coverage for control pond A7

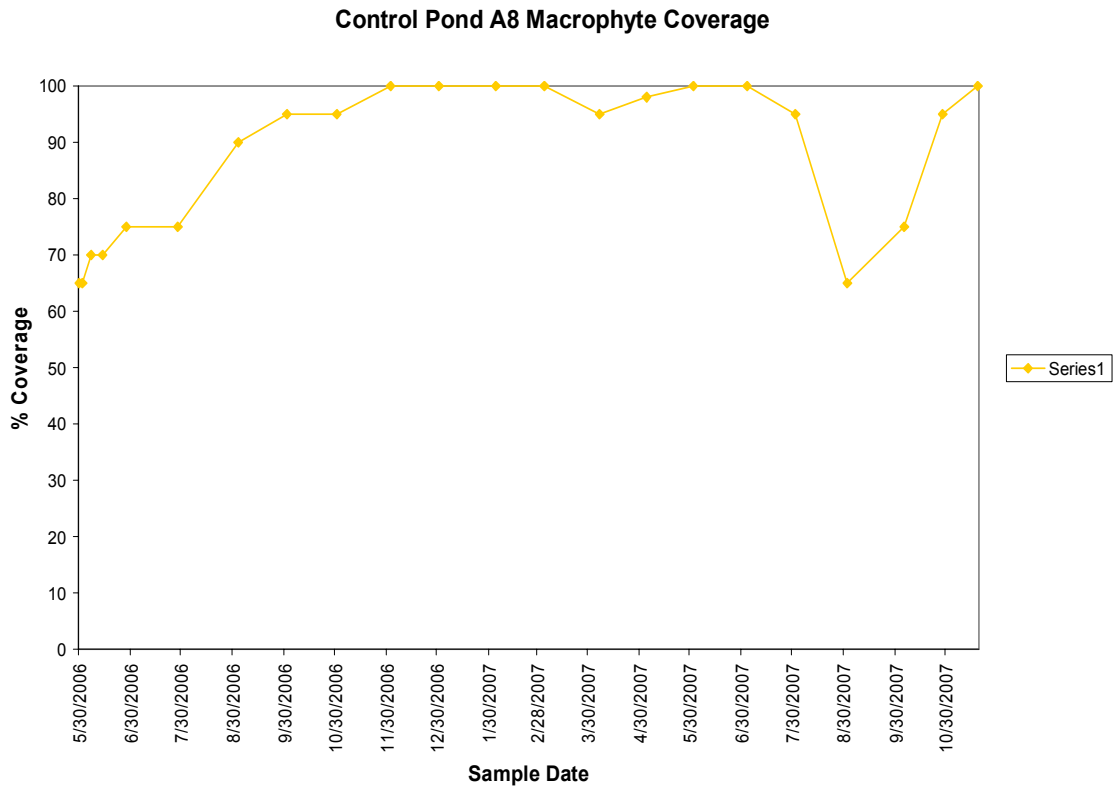

Figure A-54. Macrophyte coverage for control pond A8

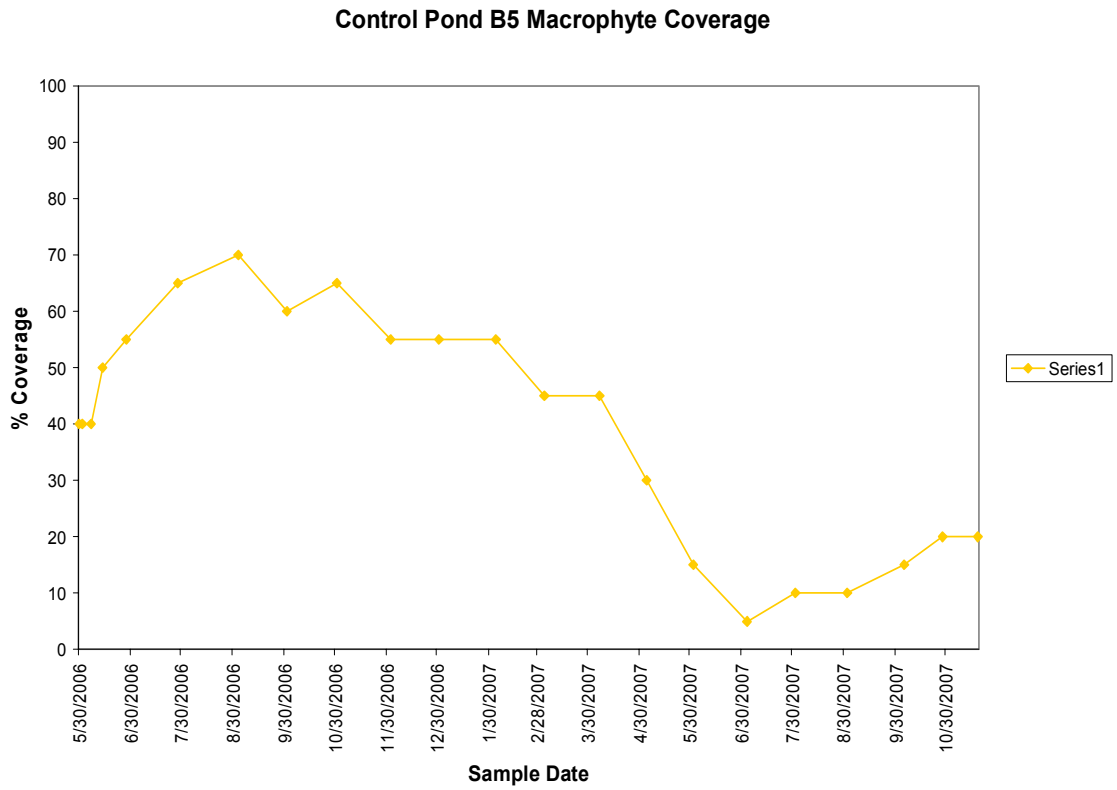

Figure A-55. Macrophyte coverage for control pond B5

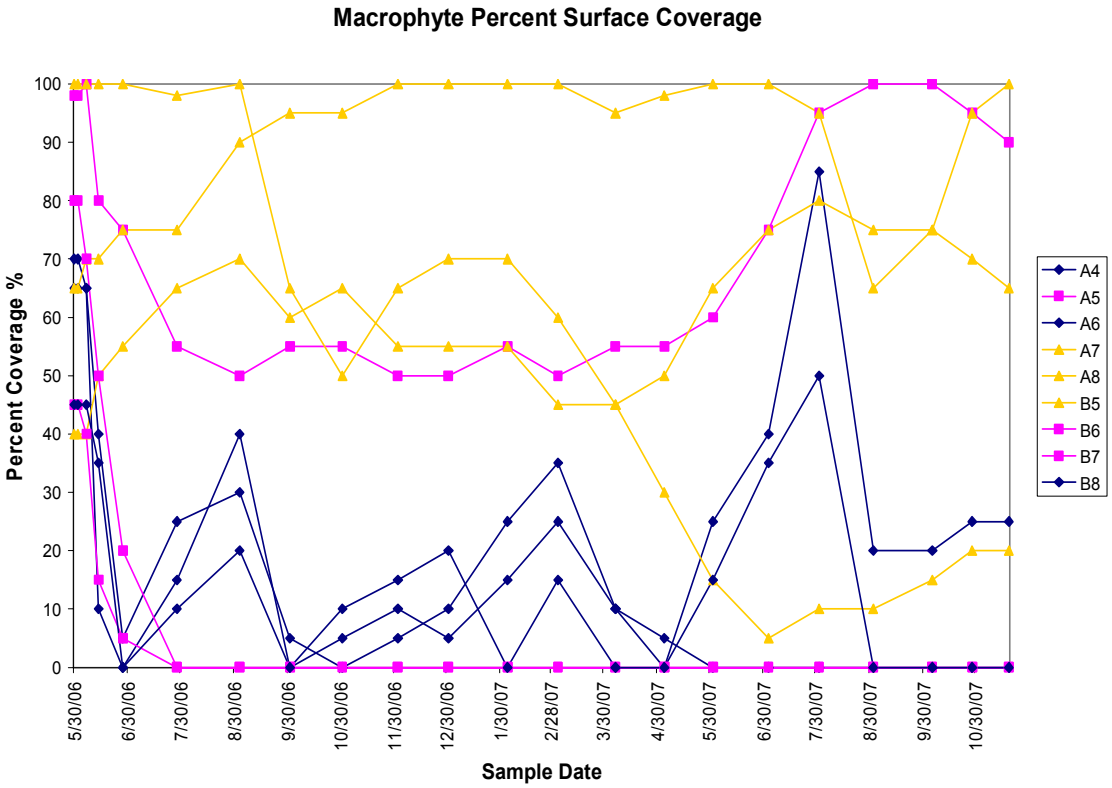

Figure A-56. Macrophyte coverage for all ponds

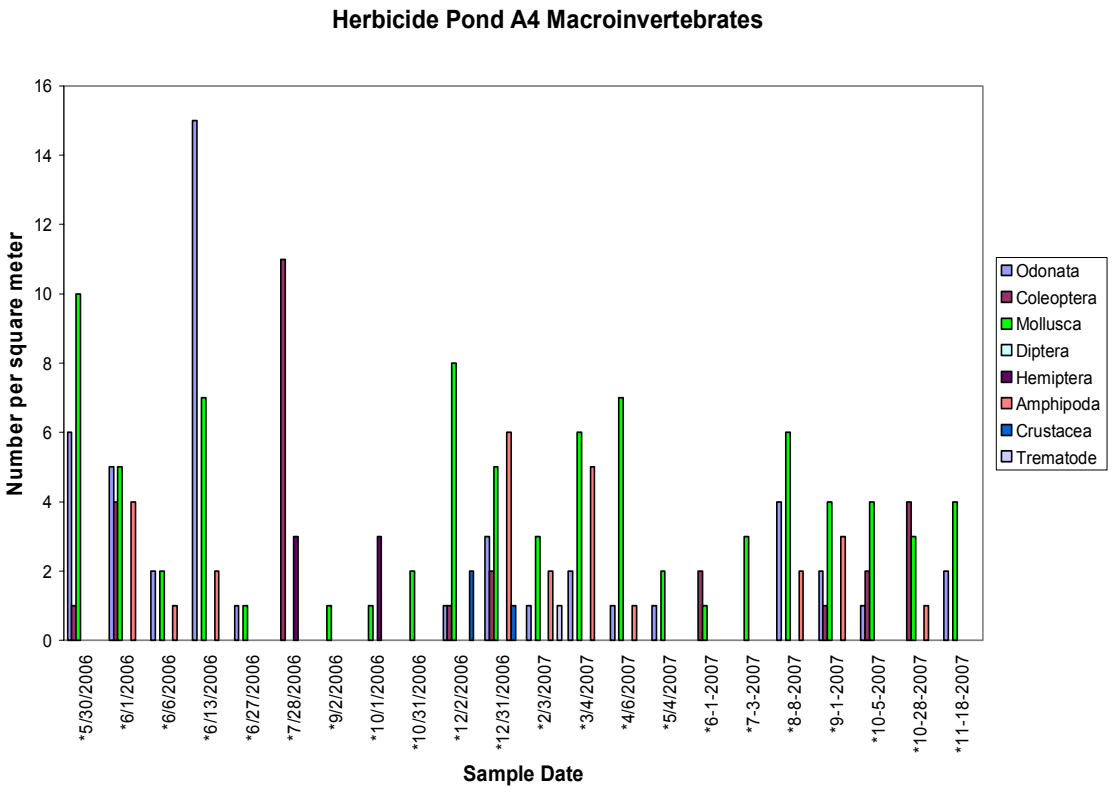

Figure A-57. Macroinvertebrate abundance for herbicide pond A4

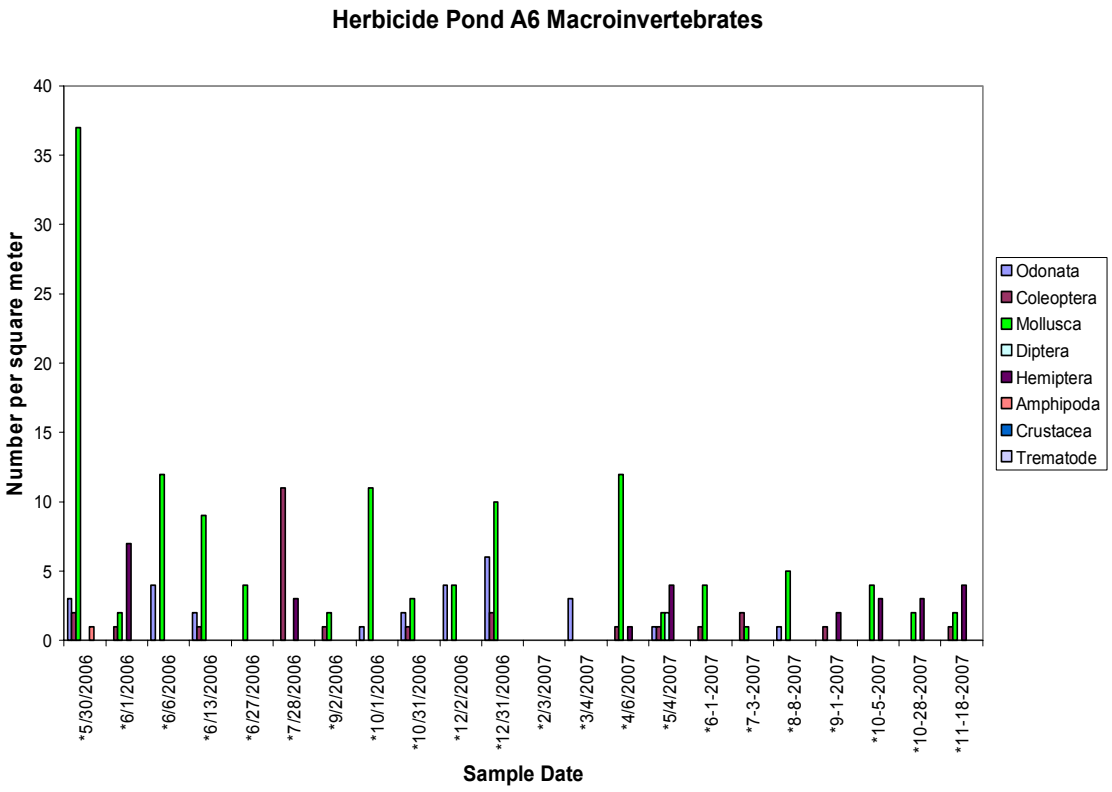

Figure A-58. Macroinvertebrate abundance for herbicide pond A6

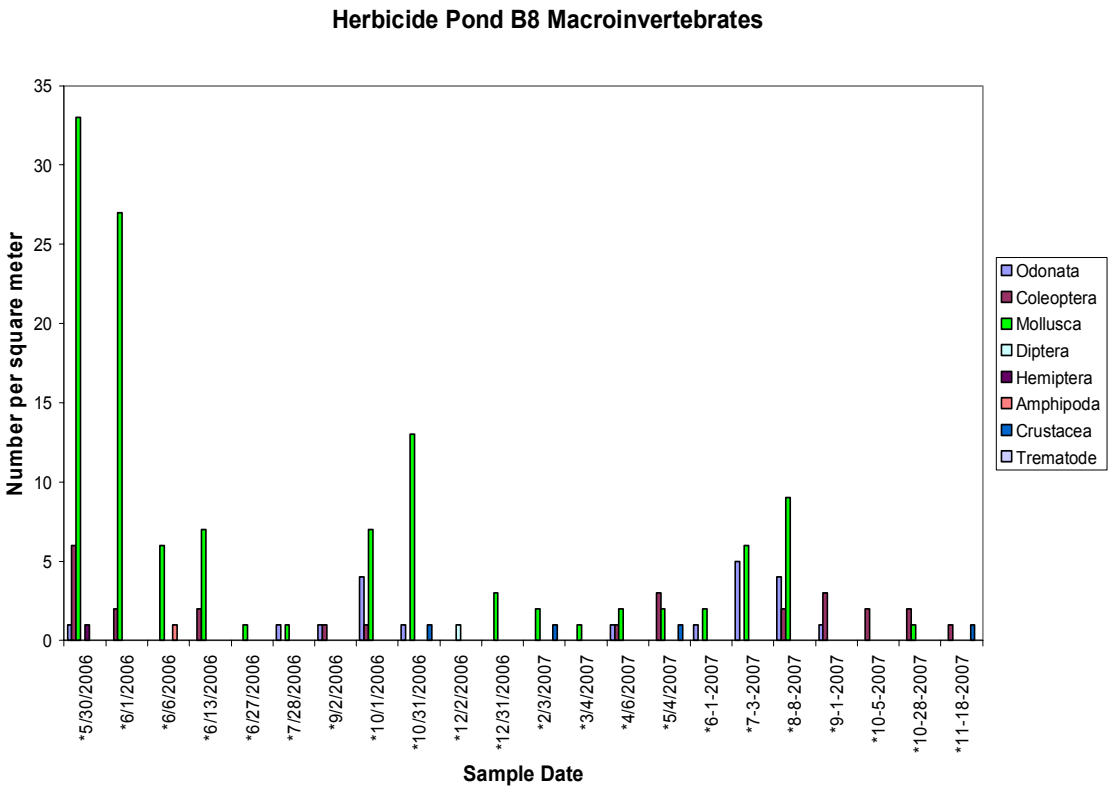

Figure A-59. Macroinvertebrate abundance for herbicide pond B8

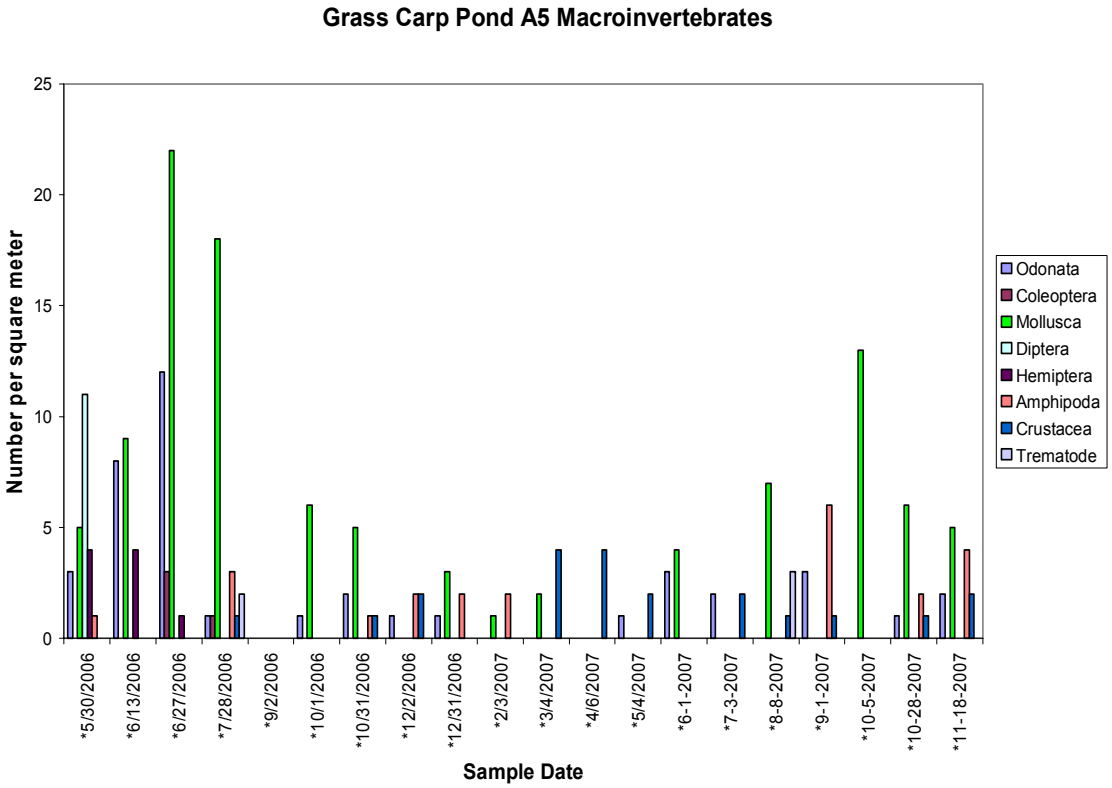

Figure A-60. Macroinvertebrate abundance for grass carp pond A5

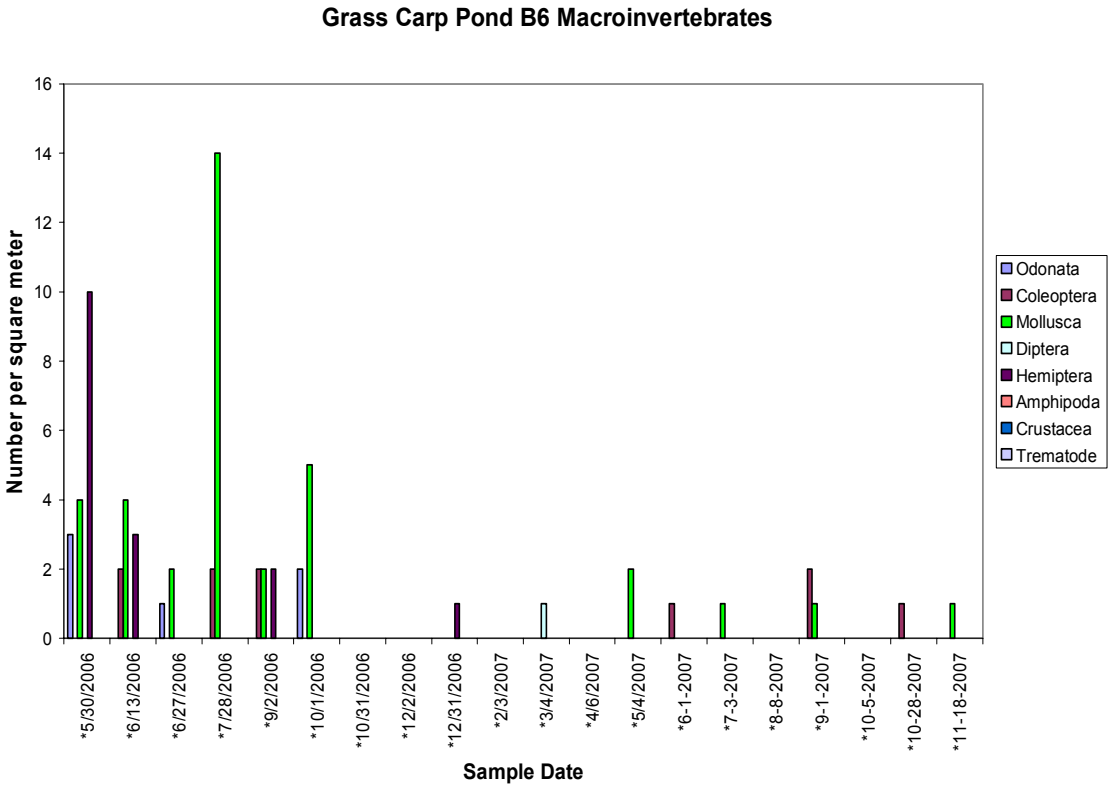

Figure A-61. Macroinvertebrate abundance for grass carp pond B6

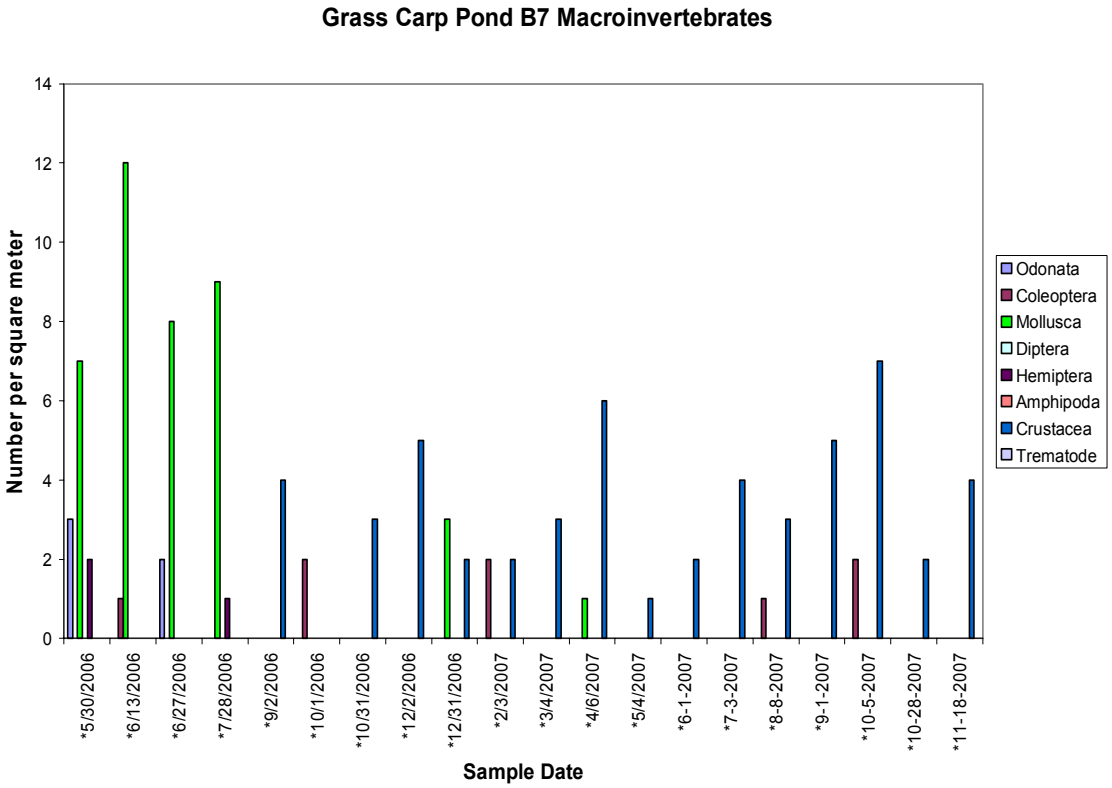

Figure A-62. Macroinvertebrate abundance for grass carp pond B7

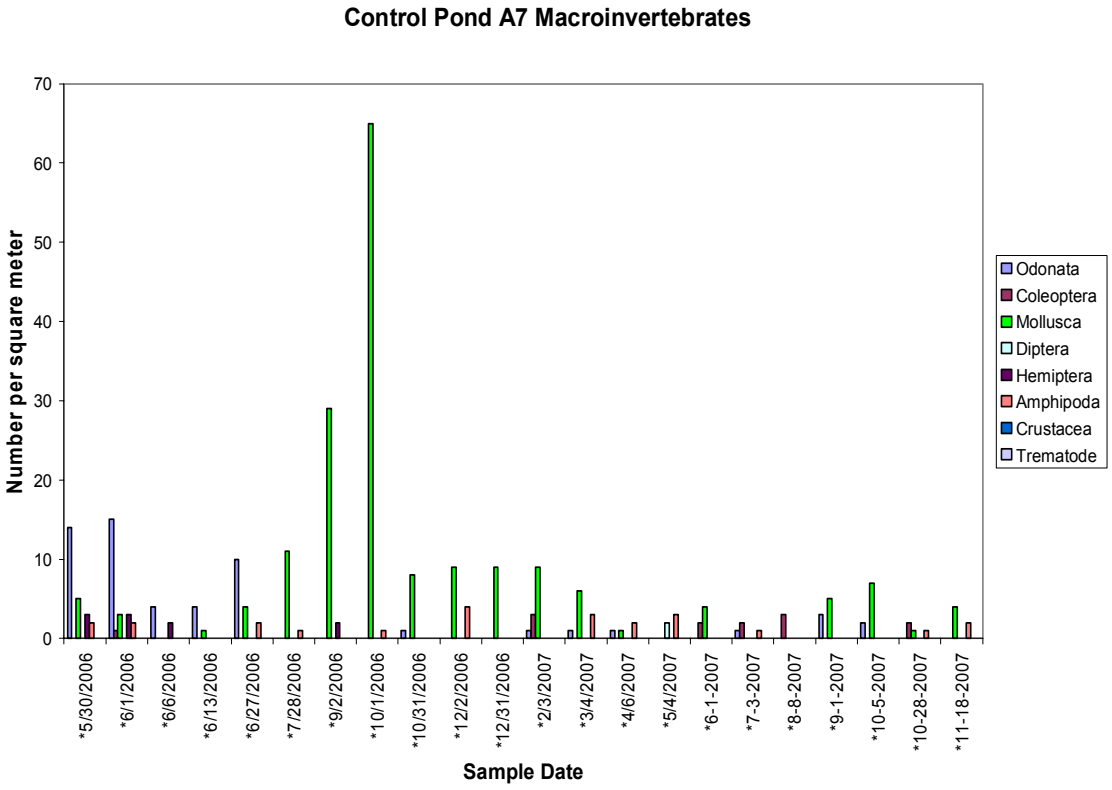

Figure A-63. Macroinvertebrate abundance for control pond A7

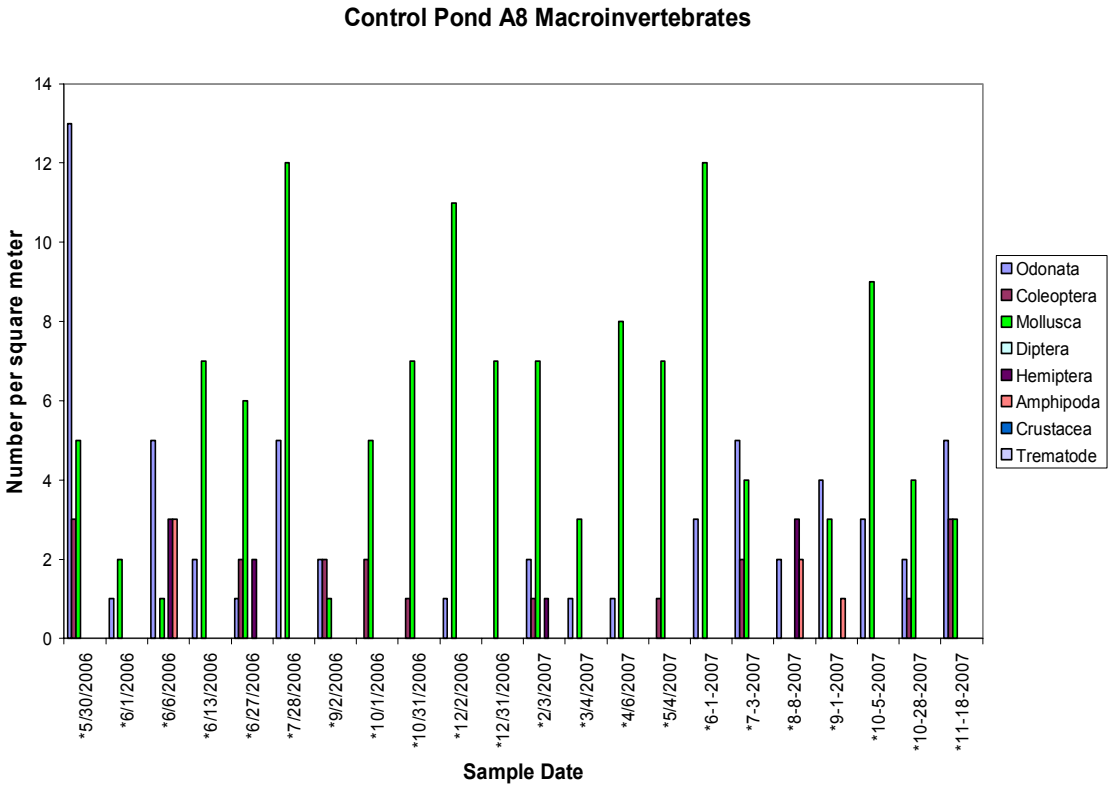

Figure A-64. Macroinvertebrate abundance for control pond A8

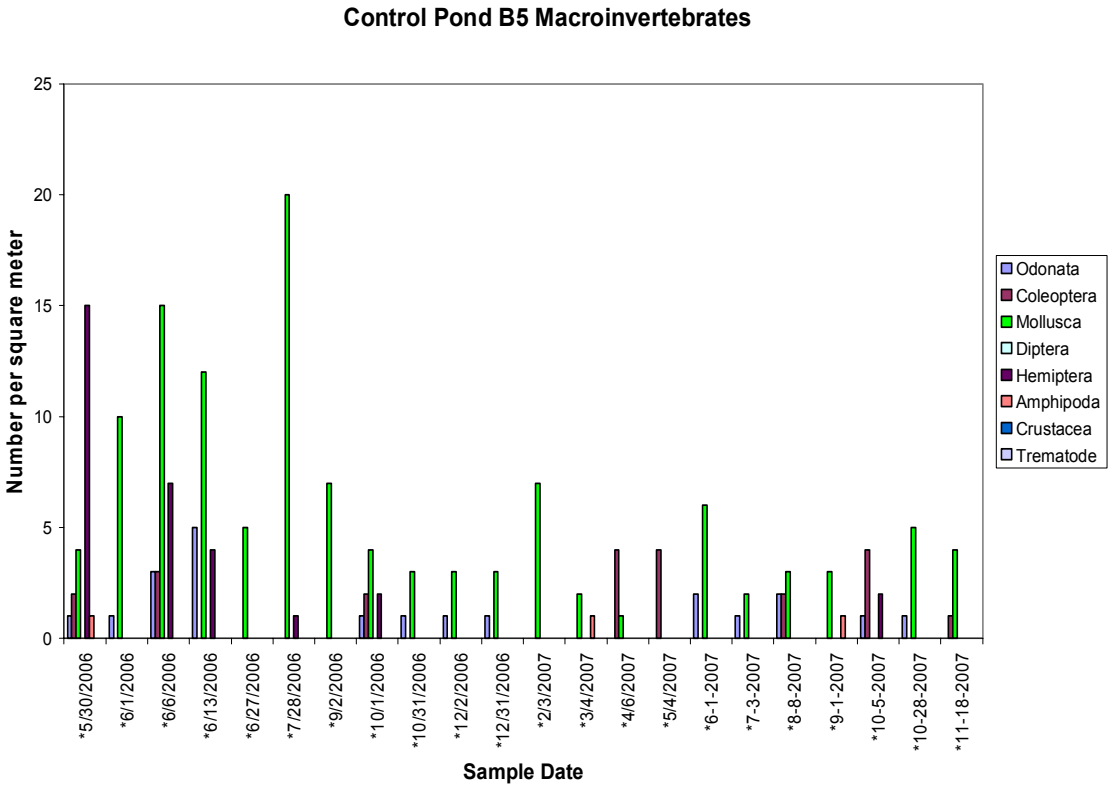

Figure A-65. Macroinvertebrate abundance for control pond B5

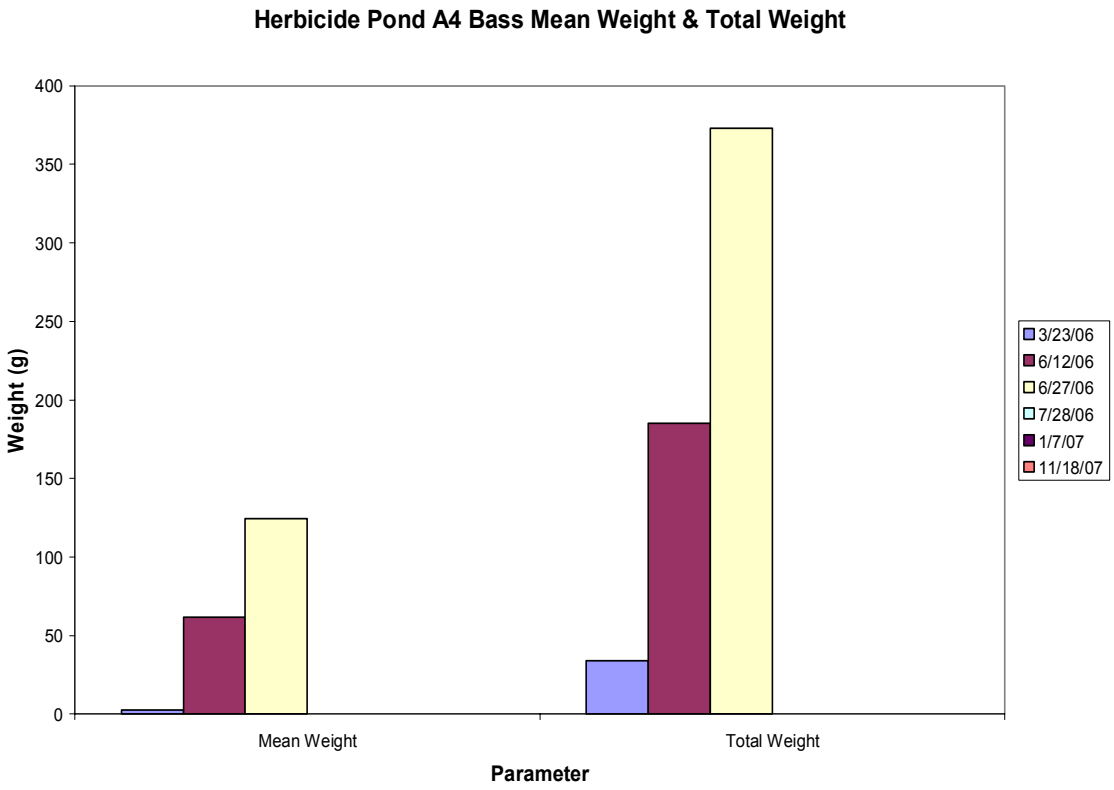

Figure A-66. Bass mean weight and total weight for herbicide pond A4

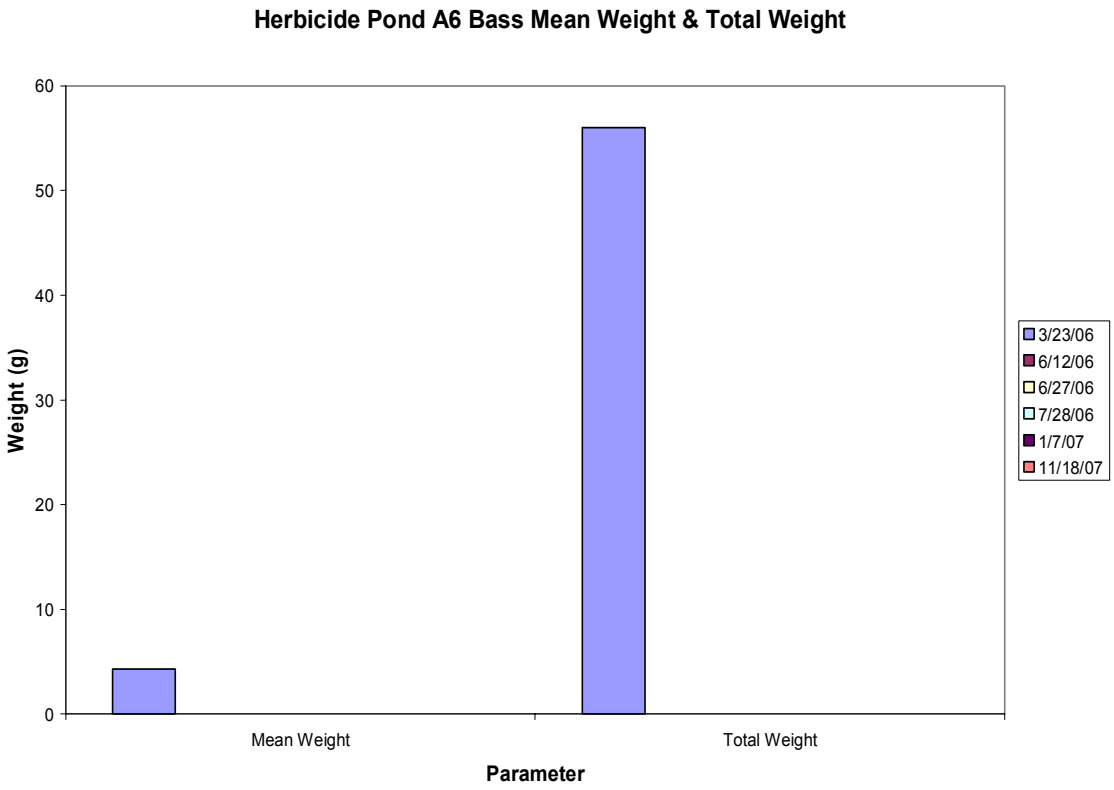

Figure A-67. Bass mean weight and total weight for herbicide pond A6

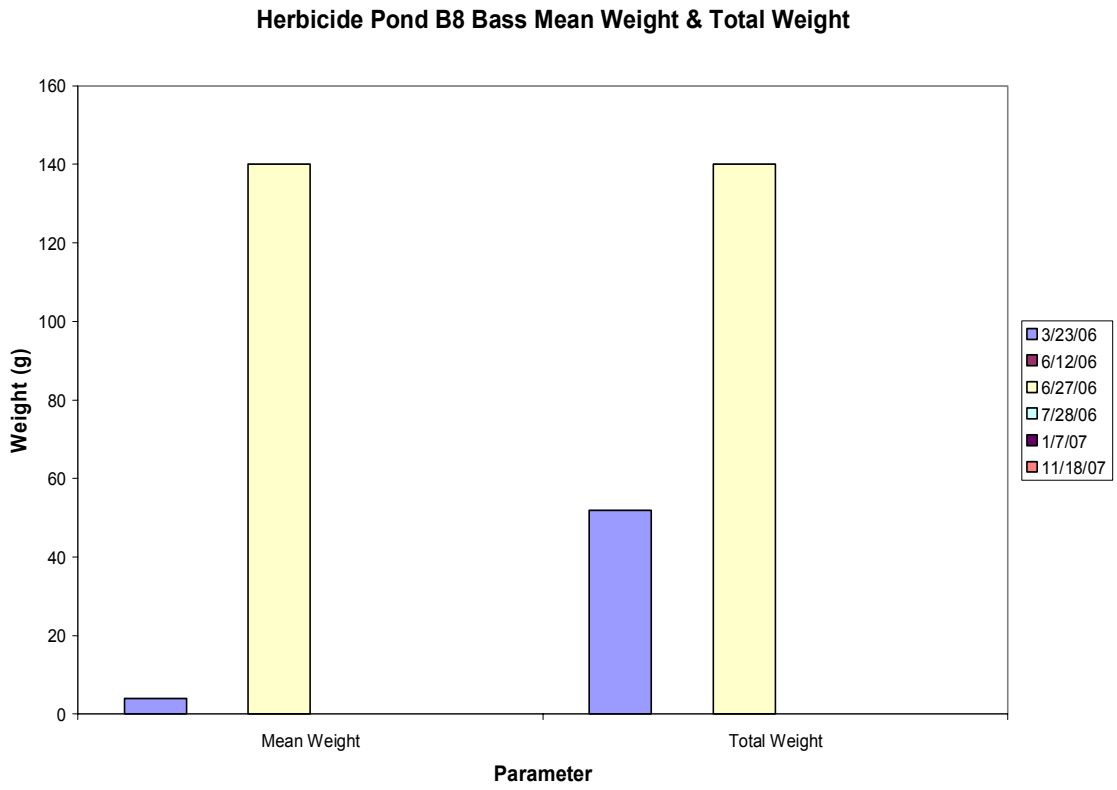

Figure A-68. Bass mean weight and total weight for herbicide pond B8

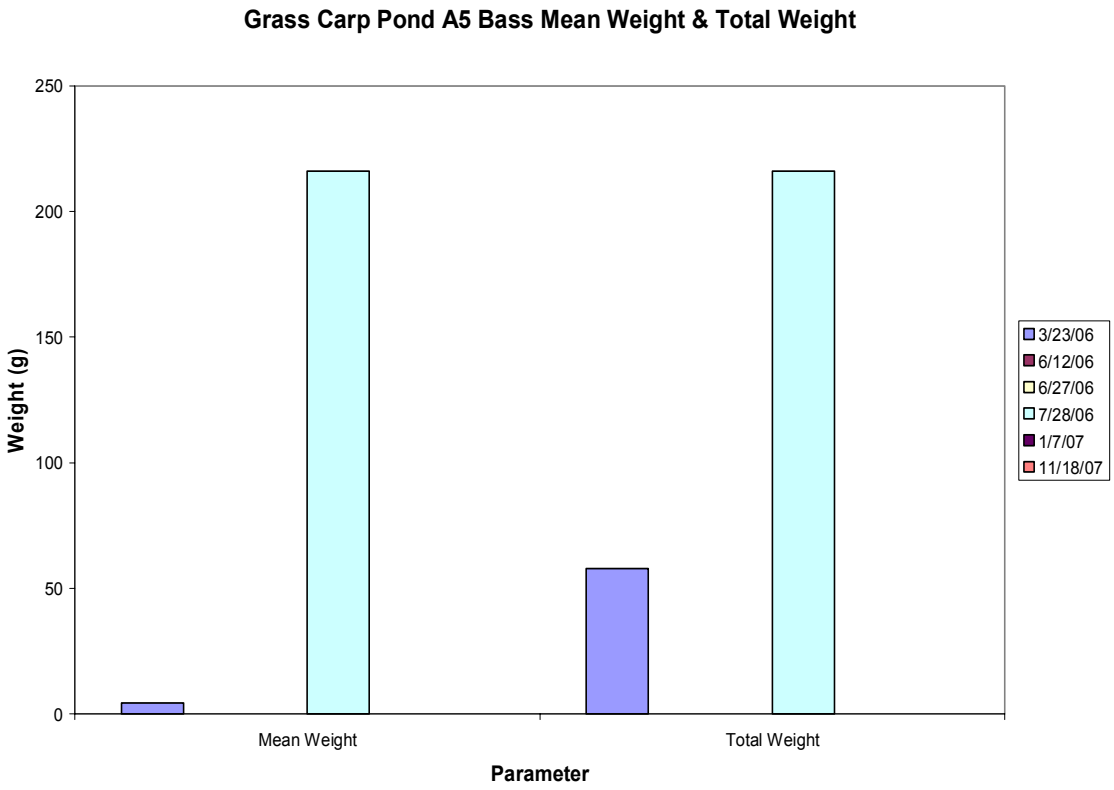

Figure A-69. Bass mean weight and total weight for grass carp pond A5

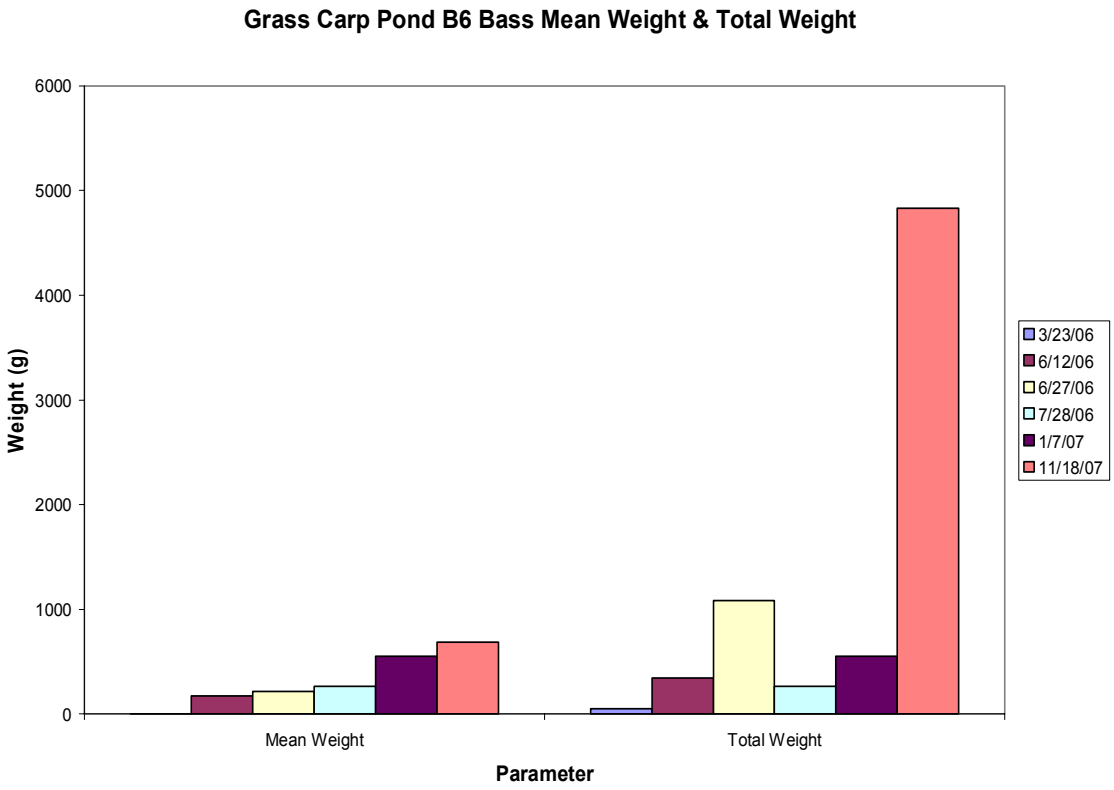

Figure A-70. Bass mean weight and total weight for grass carp pond B6

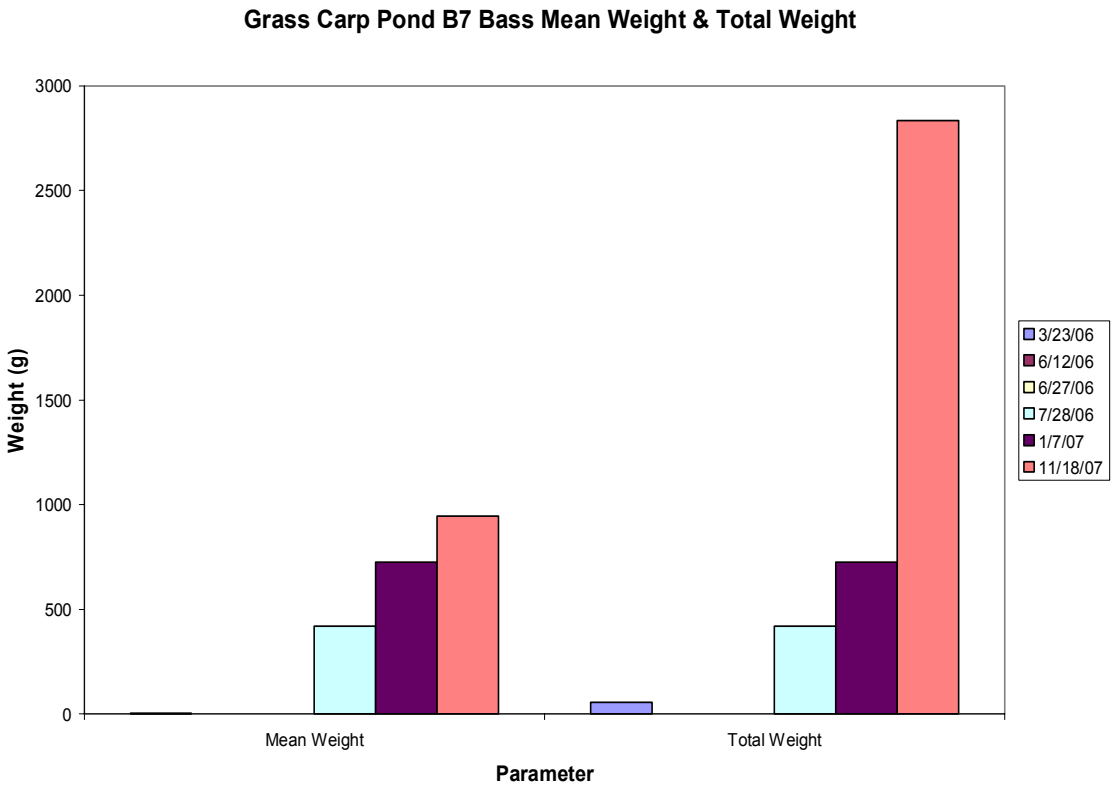

Figure A-71. Bass mean weight and total weight for grass carp pond B7

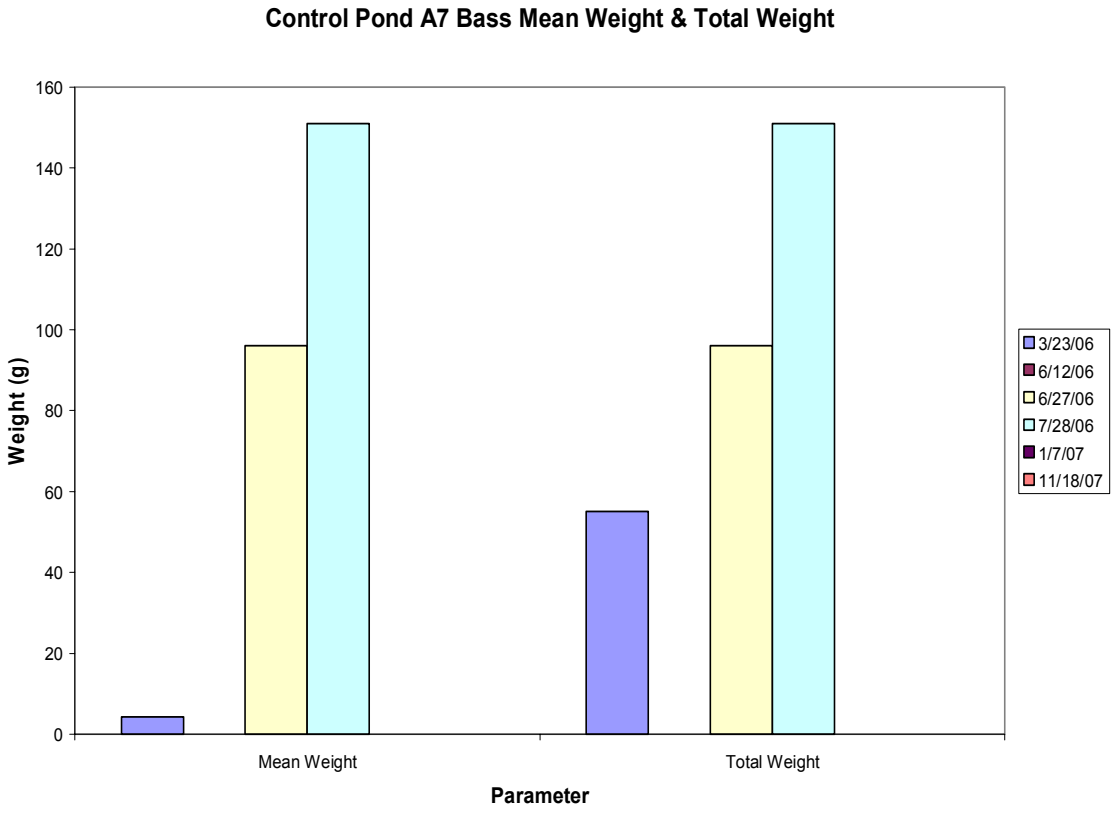

Figure A-72. Bass mean weight and total weight for control pond A7

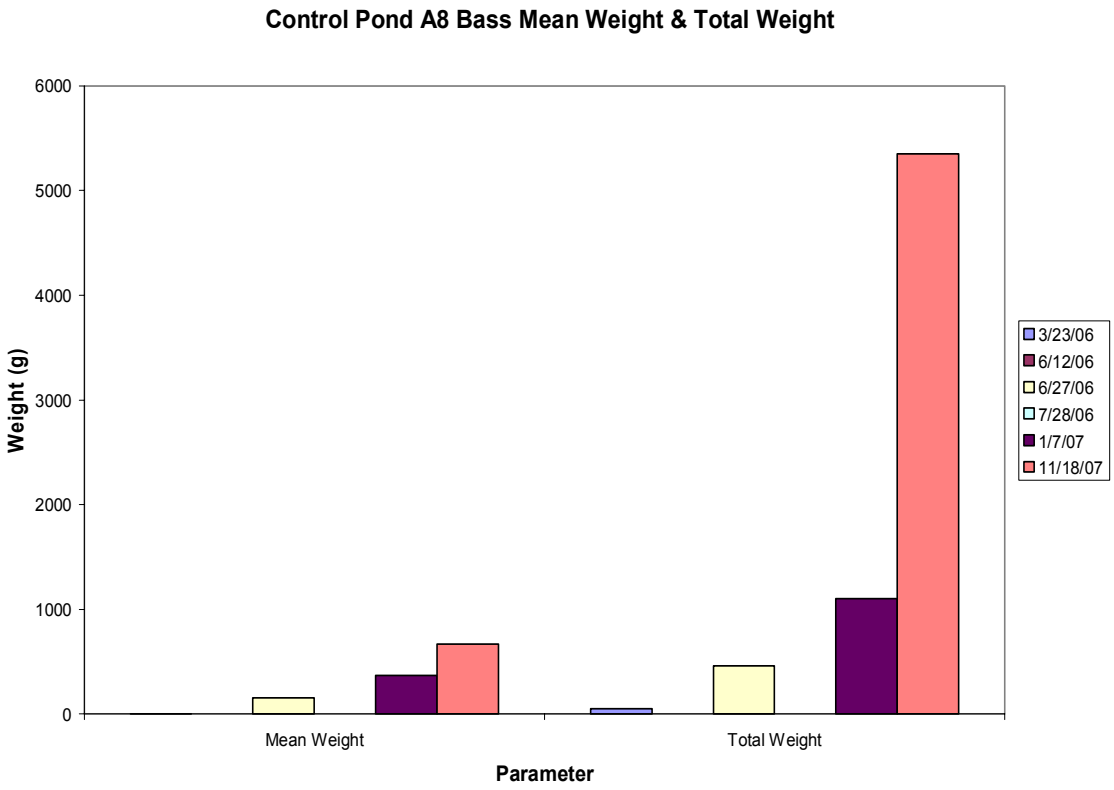

Figure A-73. Bass mean weight and total weight for control pond A8

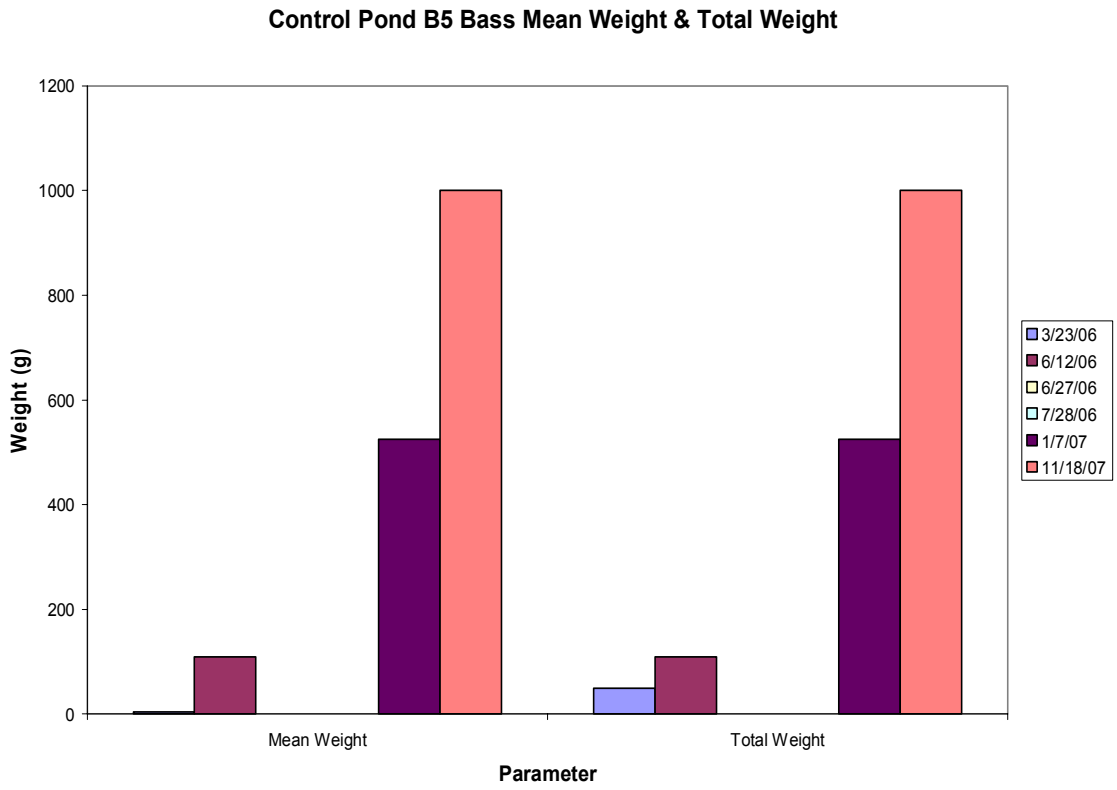

Figure A-74. Bass mean weight and total weight for control pond B5

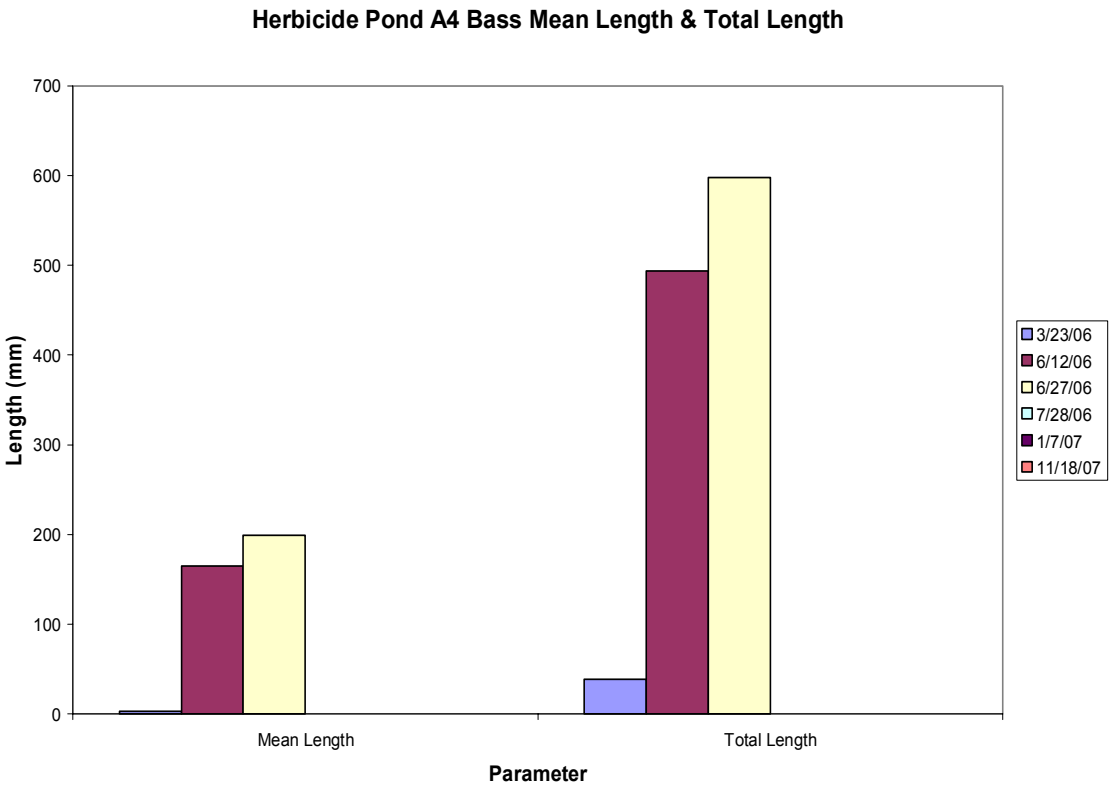

Figure A-75. Bass mean length and total length for herbicide pond A4

Figure A-76. Bass mean length and total length for herbicide pond A6

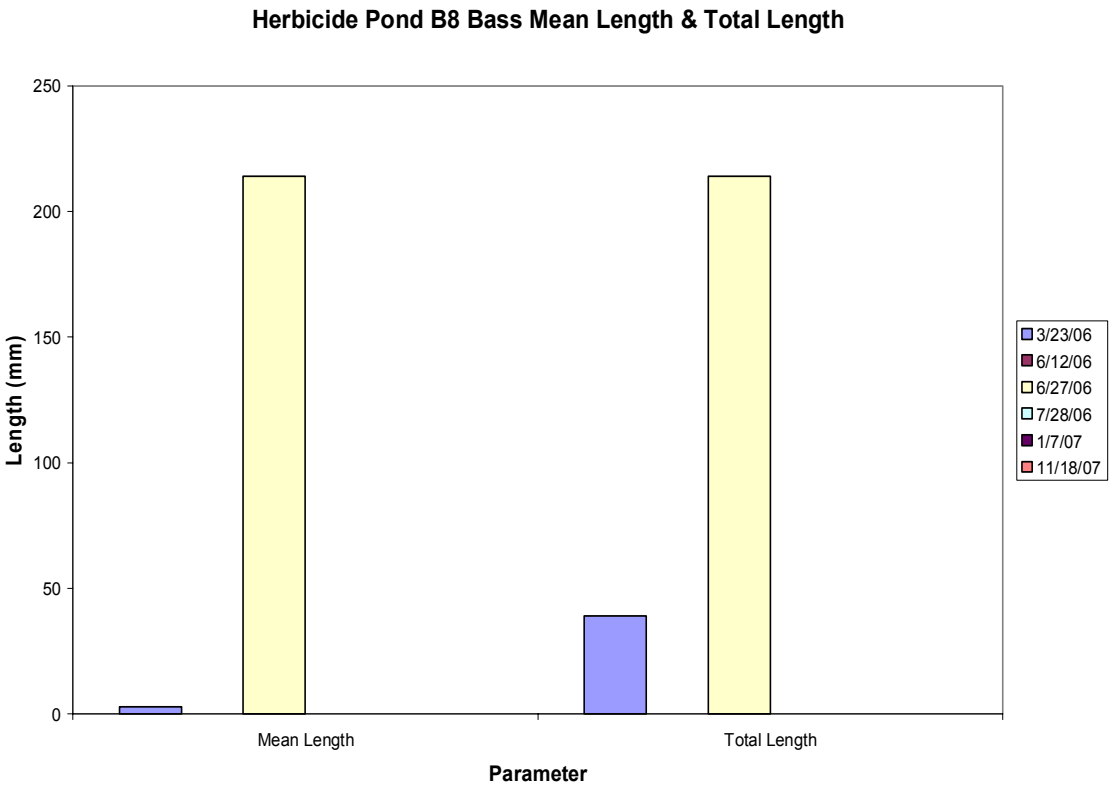

Figure A-77. Bass mean length and total length for herbicide pond B8

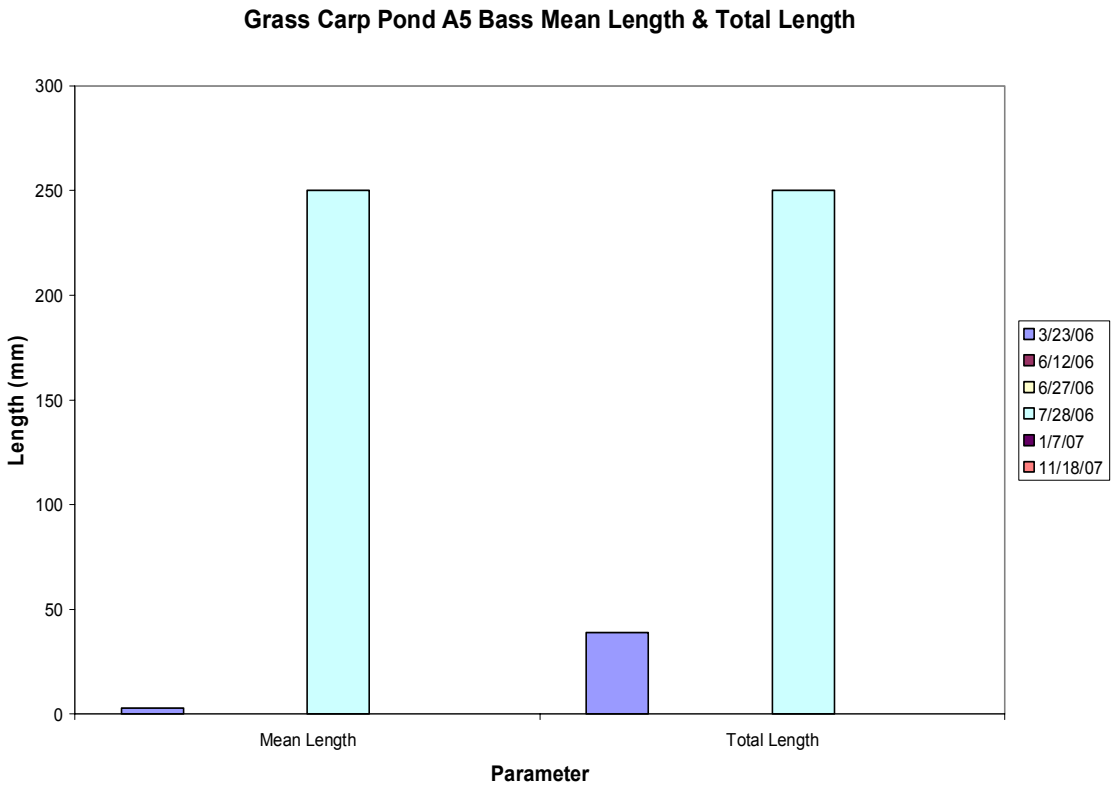

Figure A-78. Bass mean length and total length for grass carp pond A5

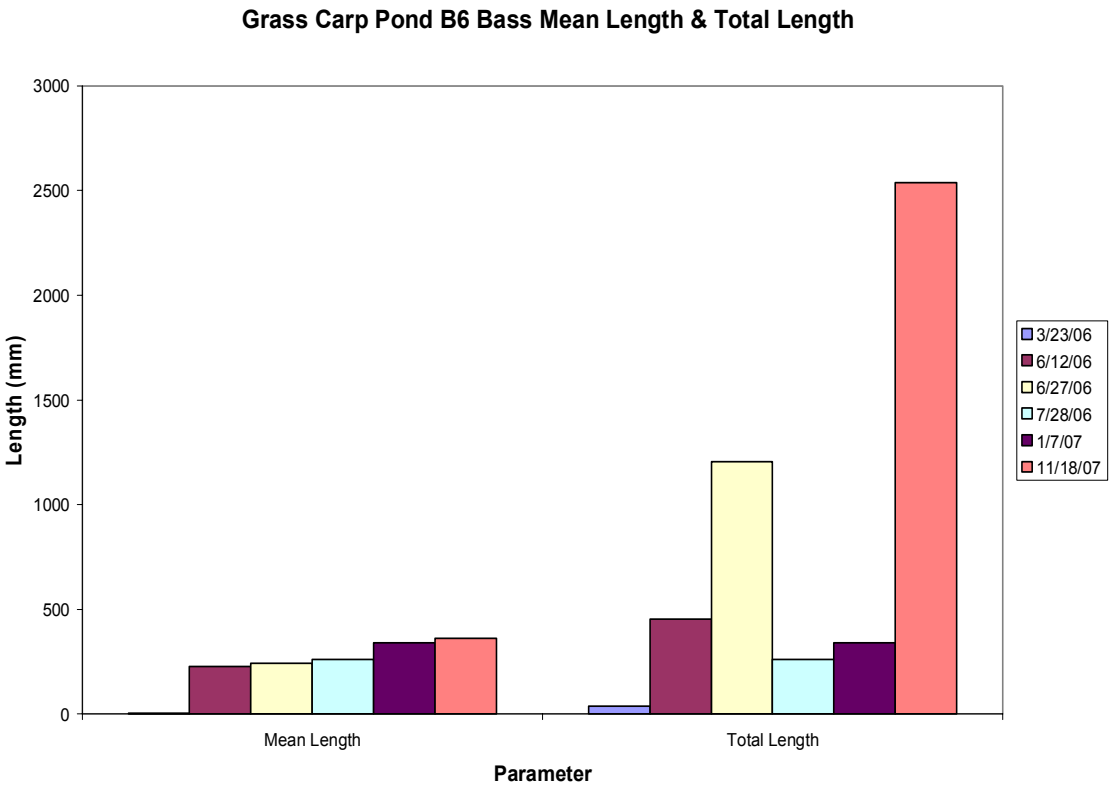

Figure A-79. Bass mean length and total length for grass carp pond B6

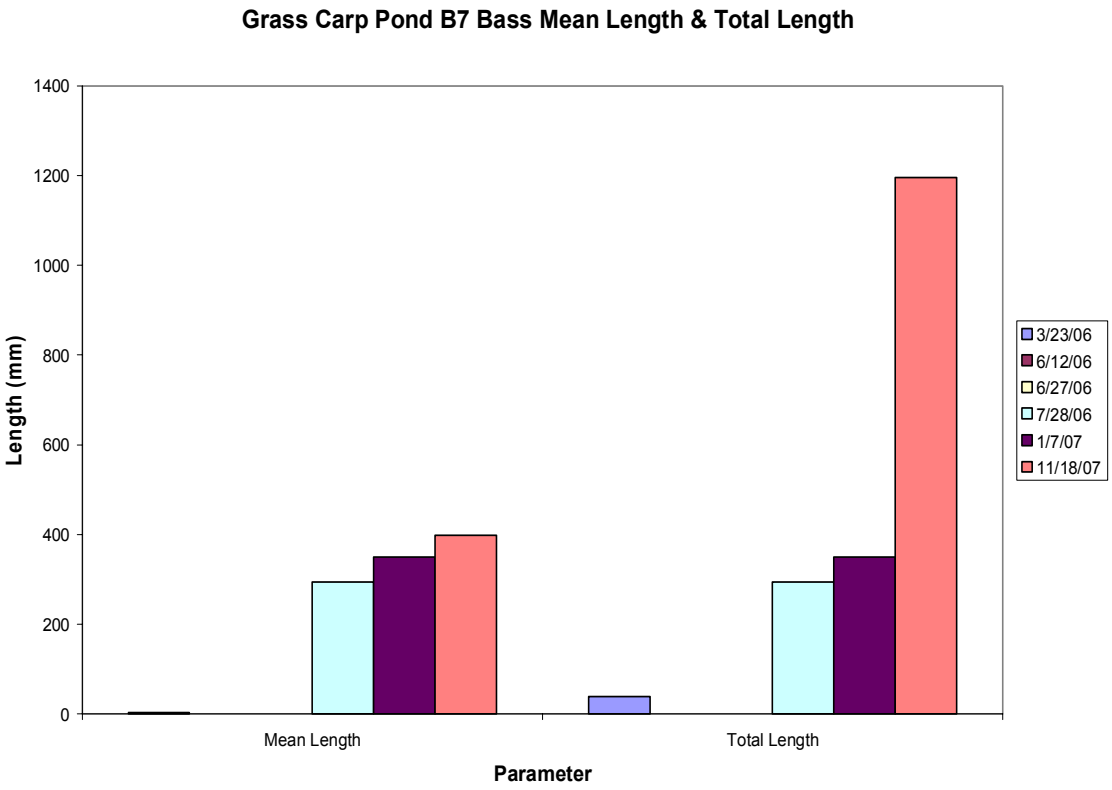

Figure A-80. Bass mean length and total length for grass carp pond B7

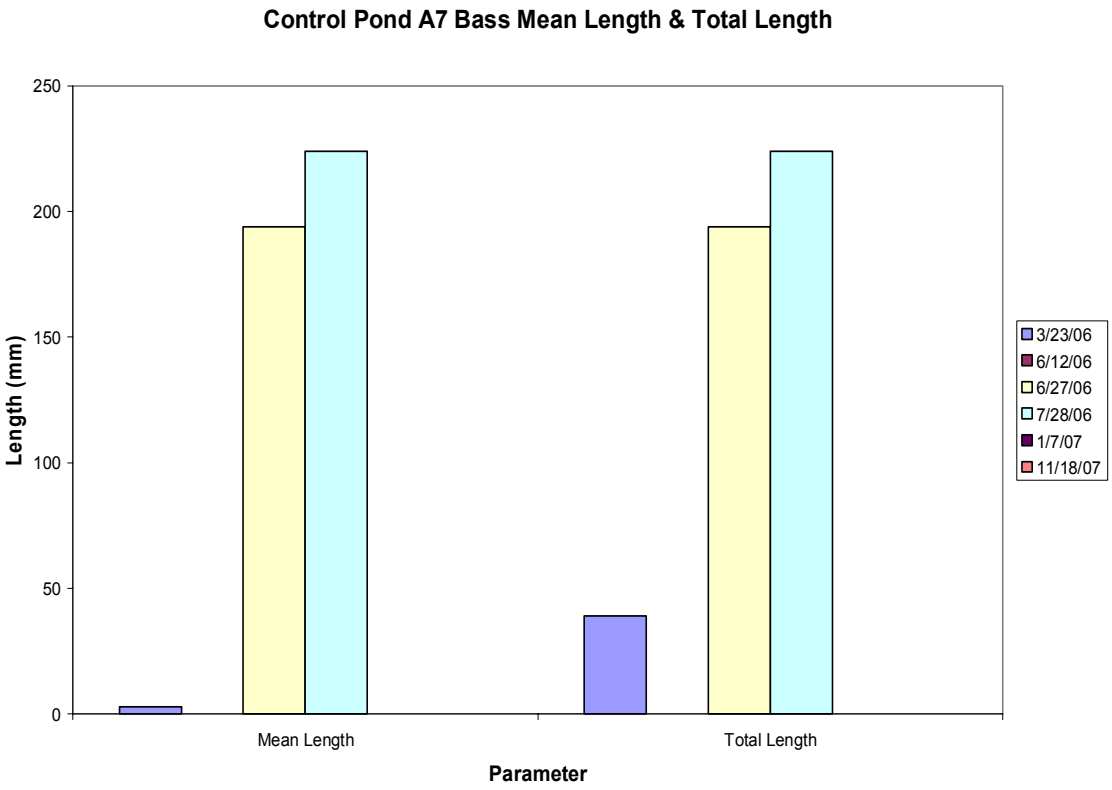

Figure A-81. Bass mean length and total length for control pond A7

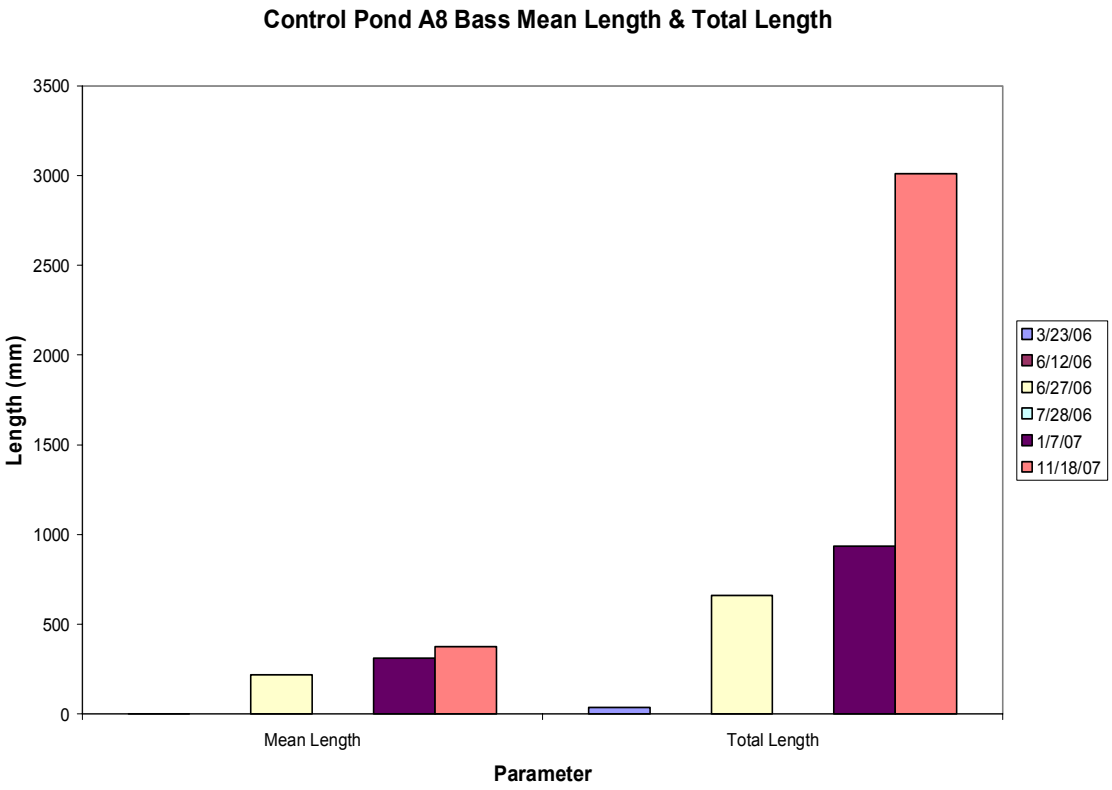

Figure A-82. Bass mean length and total length for control pond A8

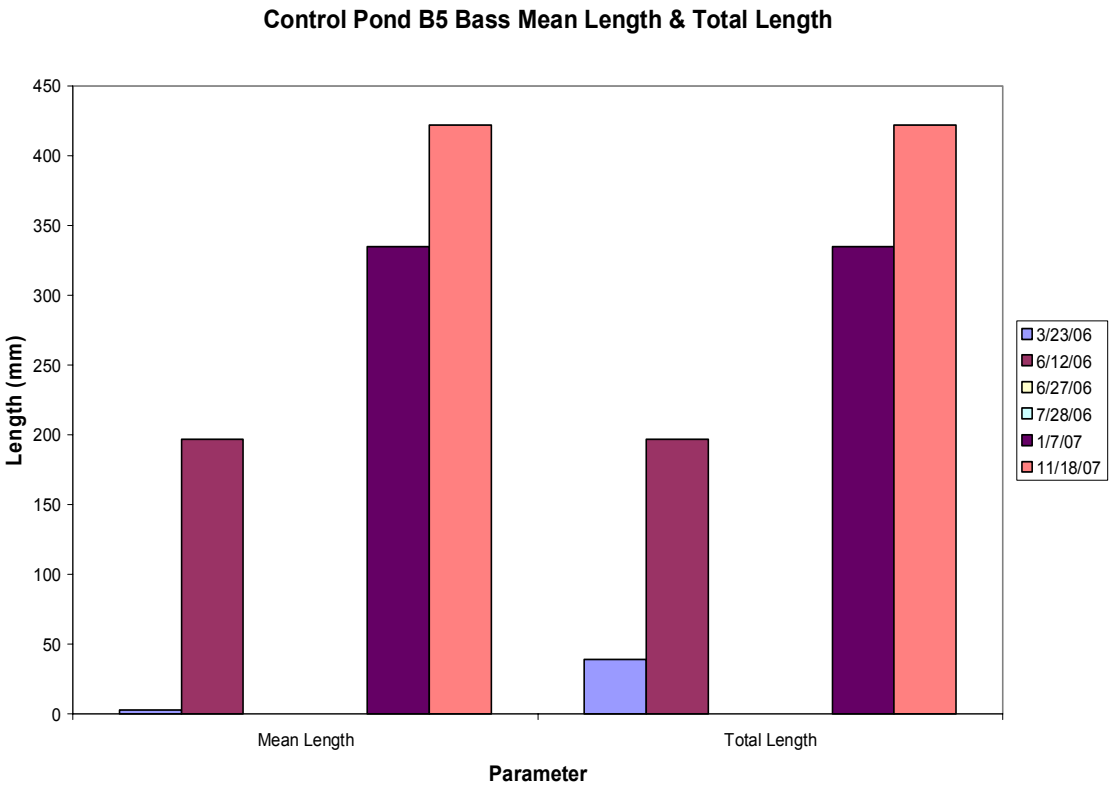

Figure A-83. Bass mean length and total length for control pond B5

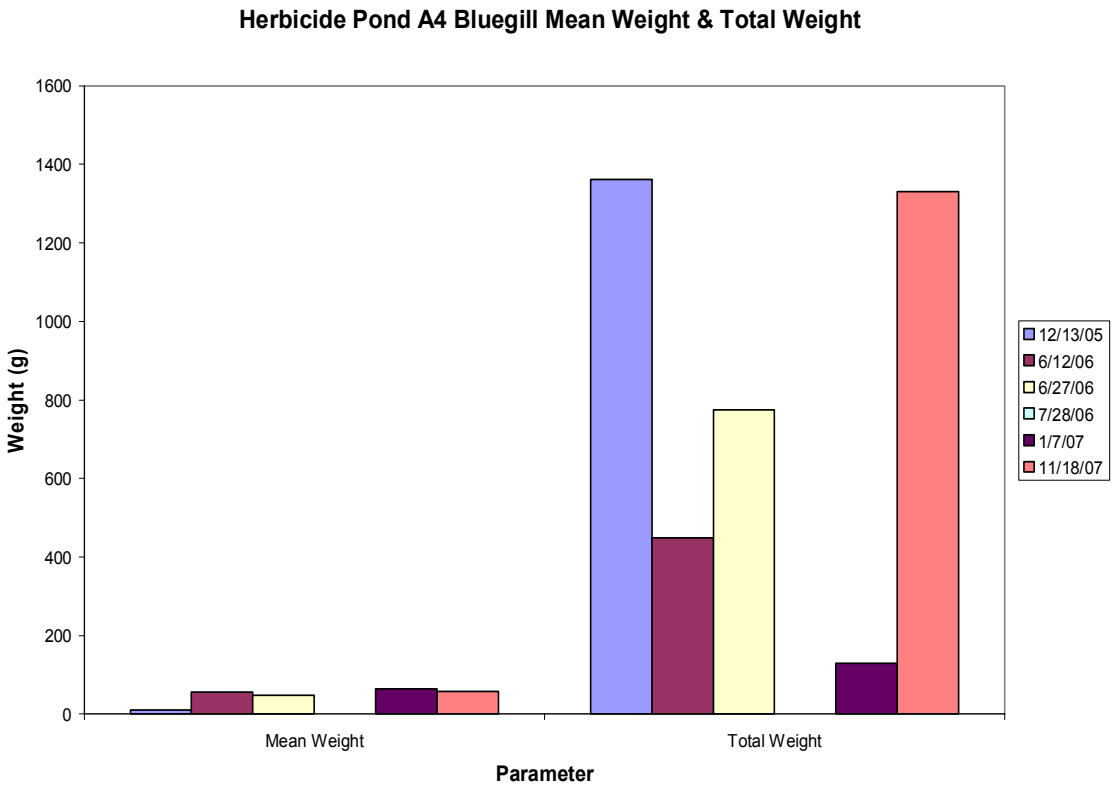

Figure A-84. Bluegill mean weight and total weight for herbicide pond A4

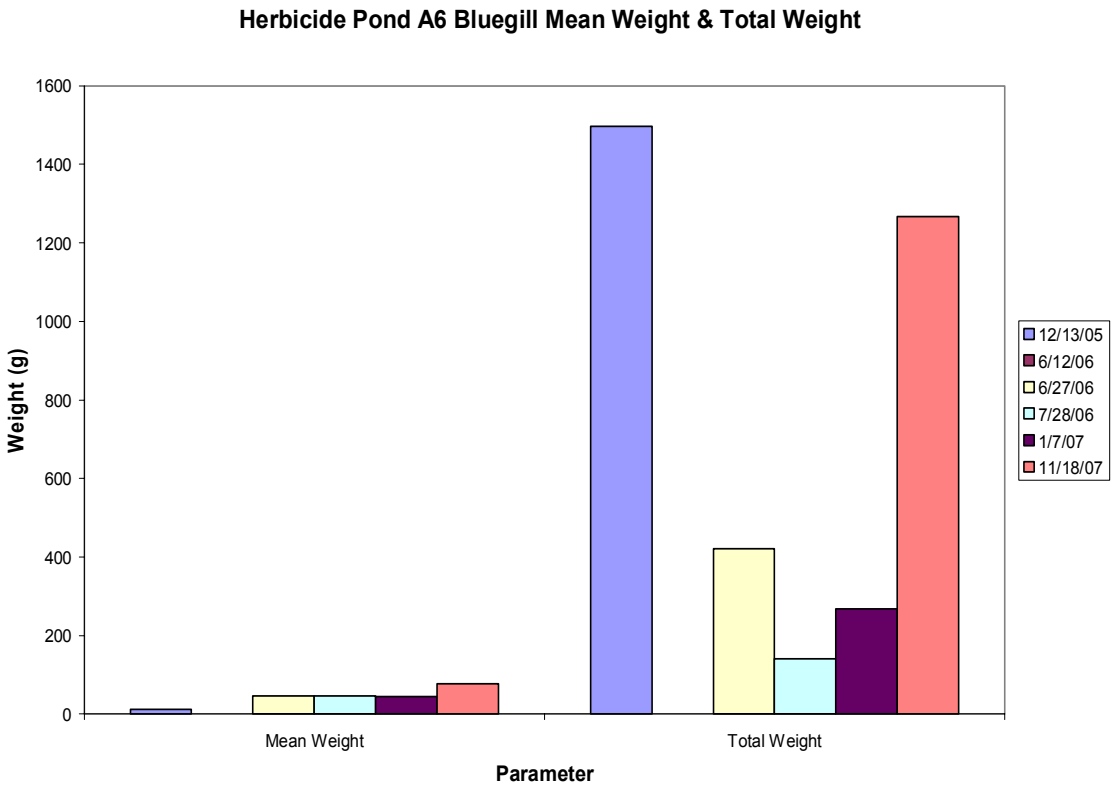

Figure A-85. Bluegill mean weight and total weight for herbicide pond A6

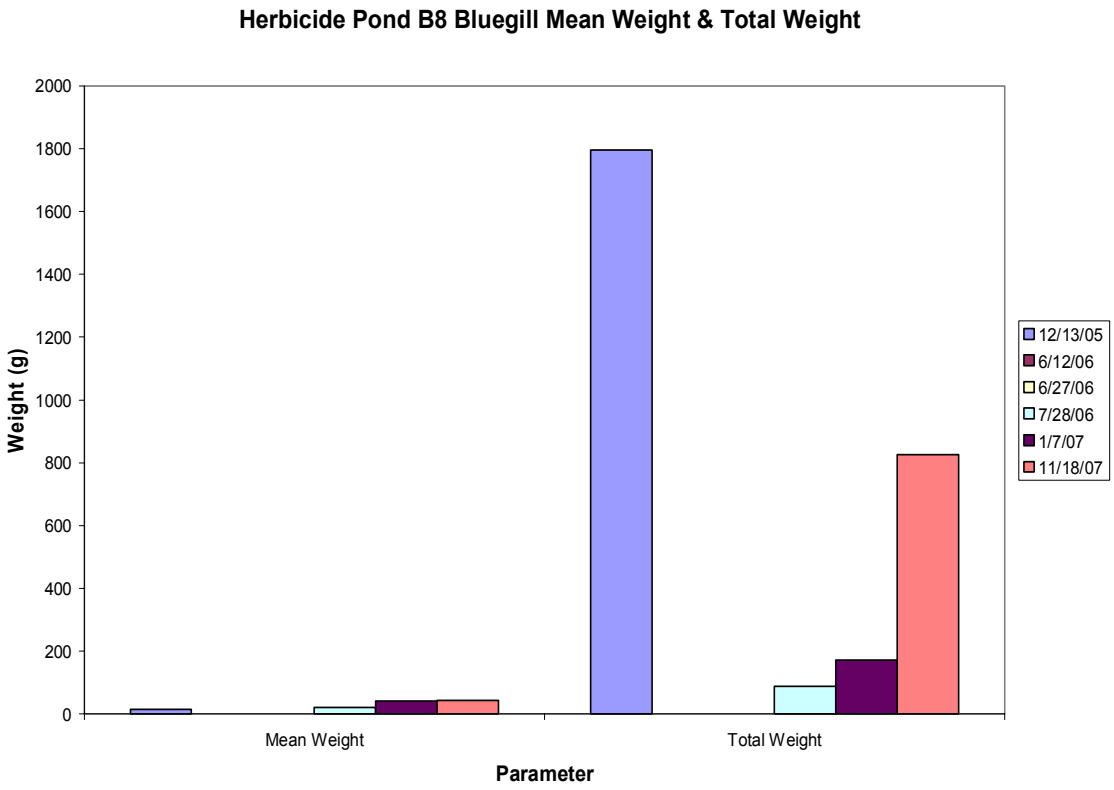

Figure A-86. Bluegill mean weight and total weight for herbicide pond B8

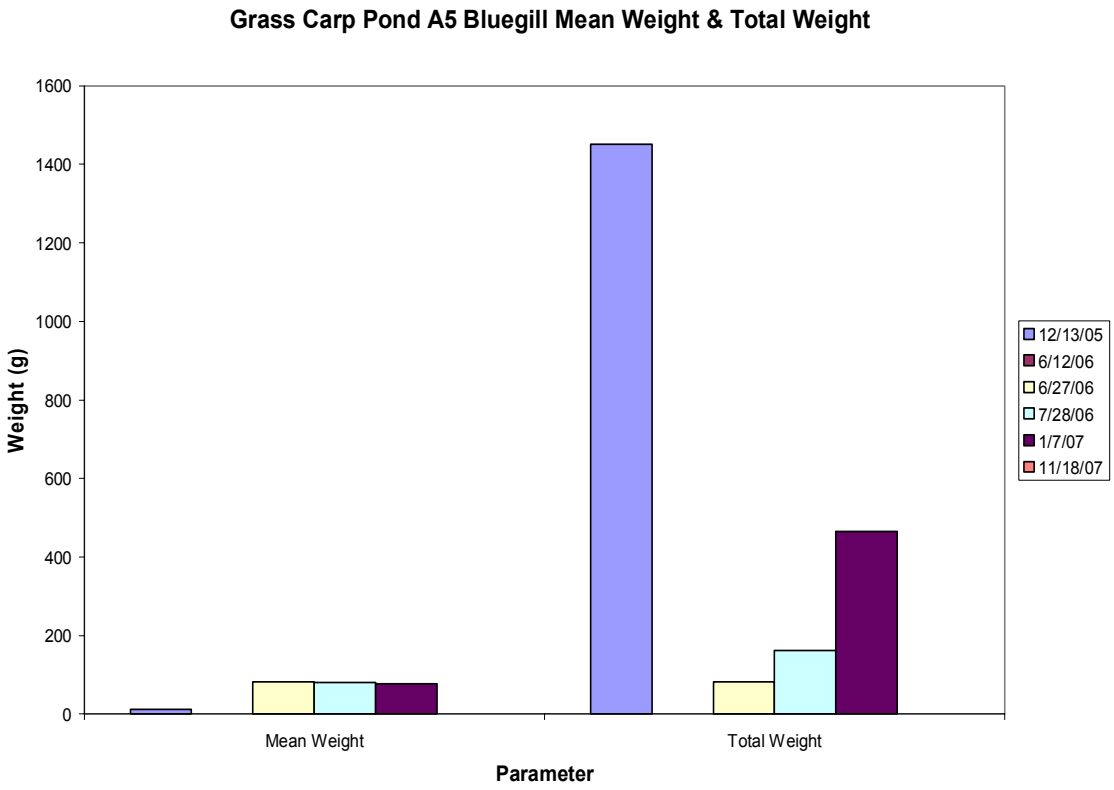

Figure A-87. Bluegill mean weight and total weight for grass carp pond A5

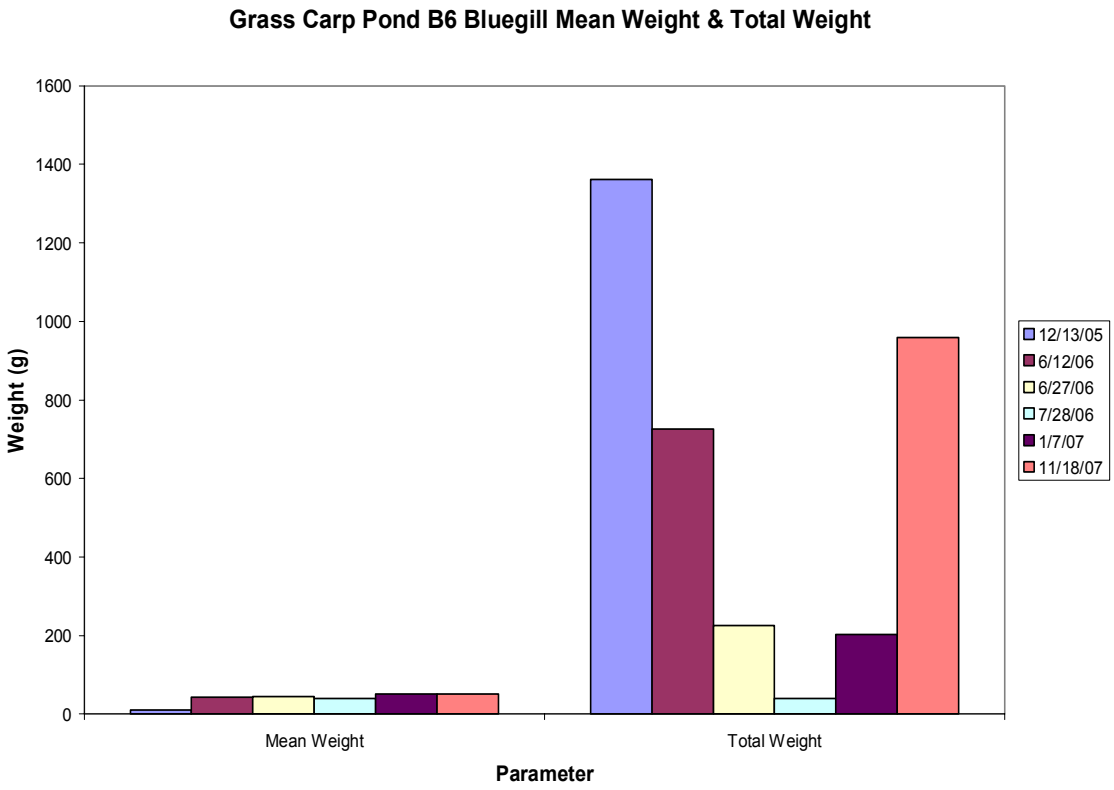

Figure A-88. Bluegill mean weight and total weight for grass carp pond B6

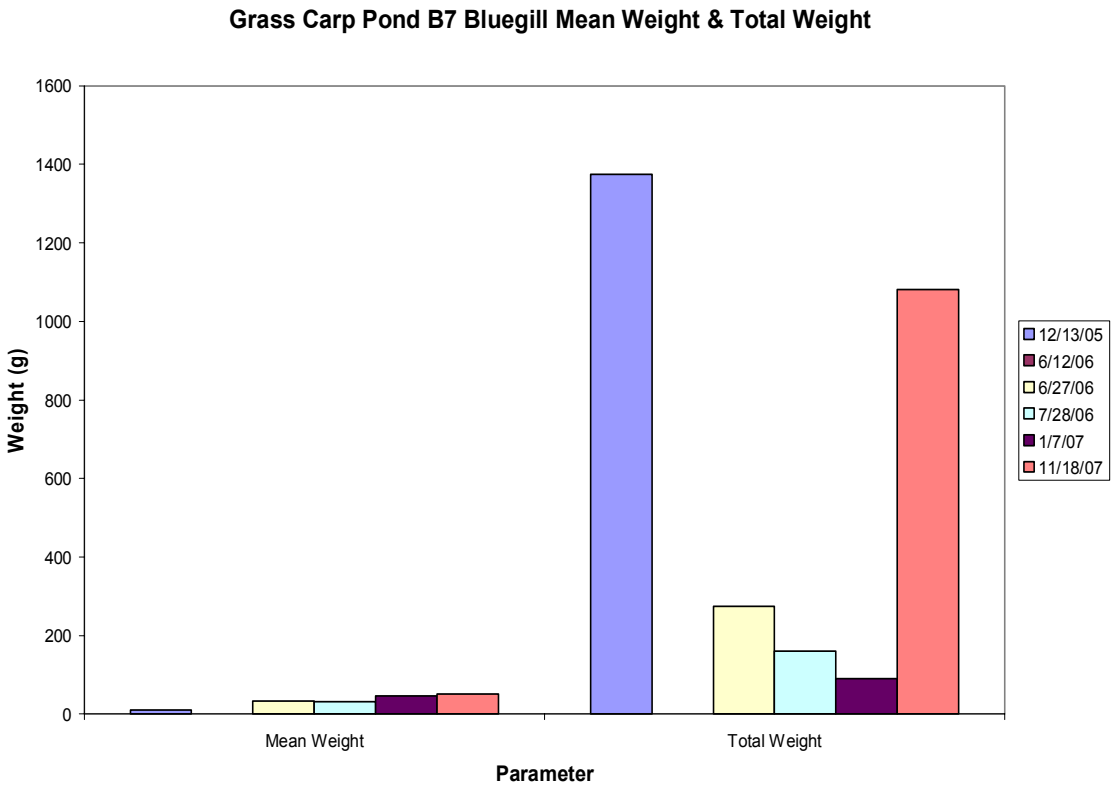

Figure A-89. Bluegill mean weight and total weight for grass carp pond B7

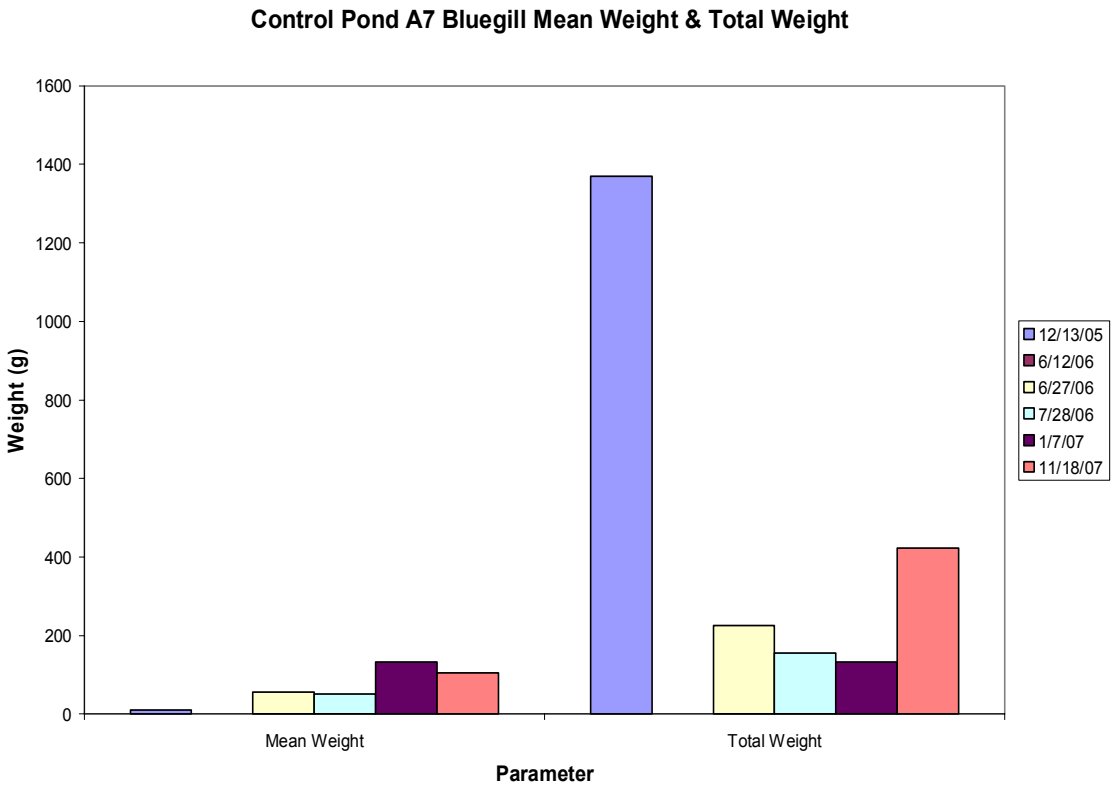

Figure A-90. Bluegill mean weight and total weight for control pond A7

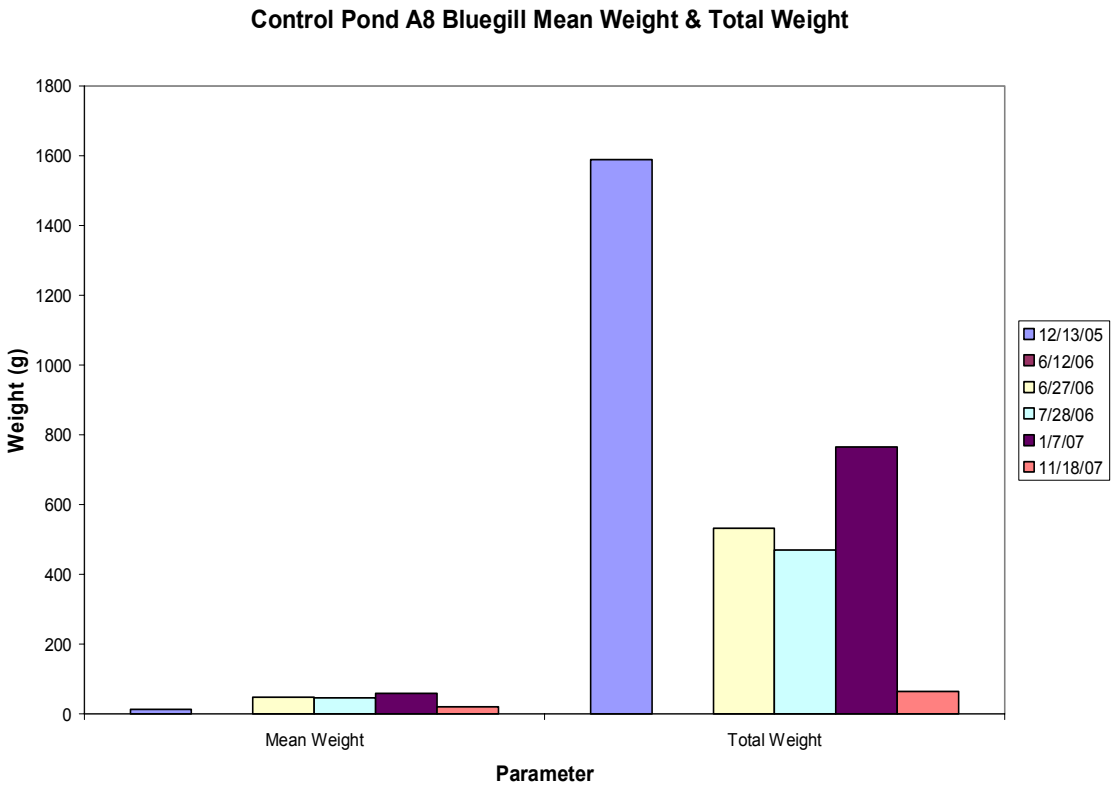

Figure A-91. Bluegill mean weight and total weight for control pond A8

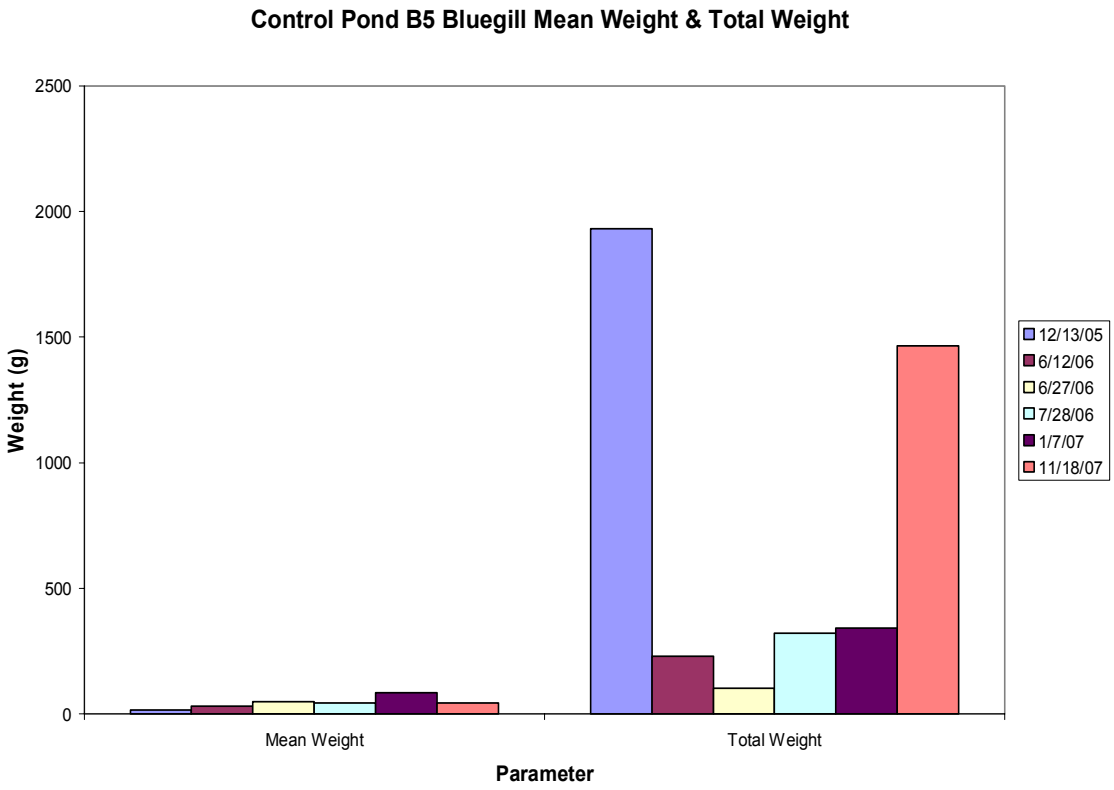

Figure A-92. Bluegill mean weight and total weight for control pond B5

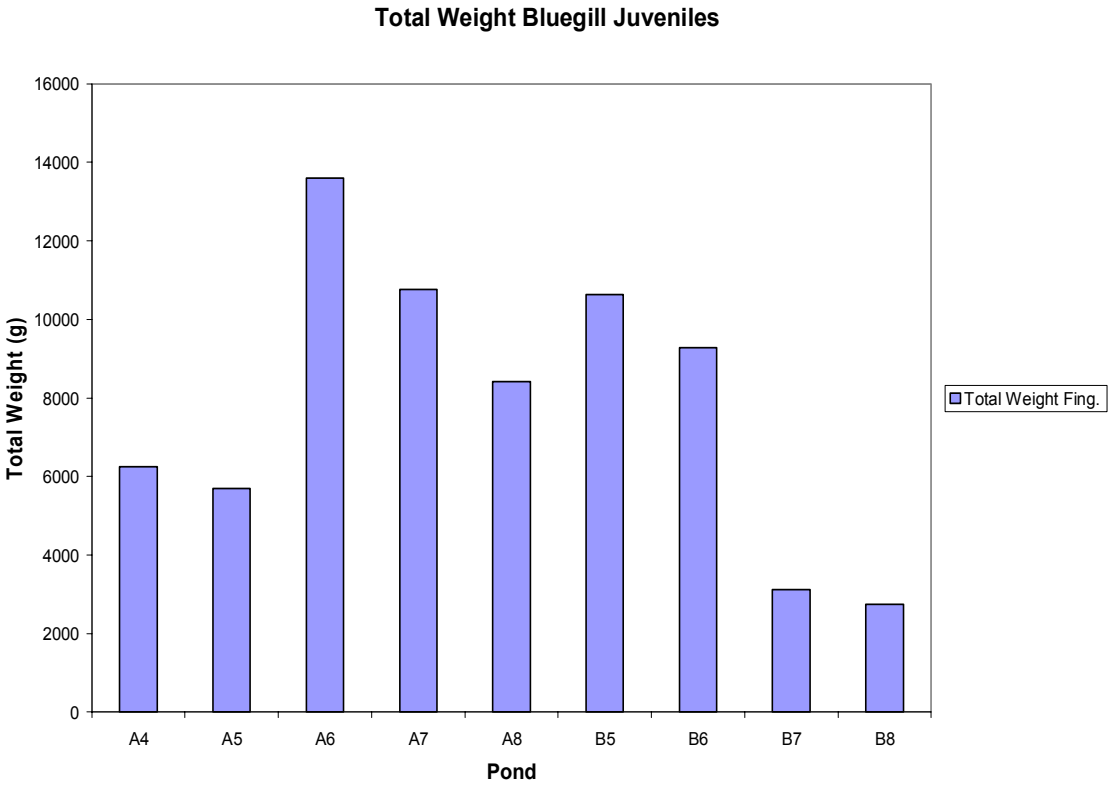

Figure A-93. Total weight of juvenile bluegill in all ponds

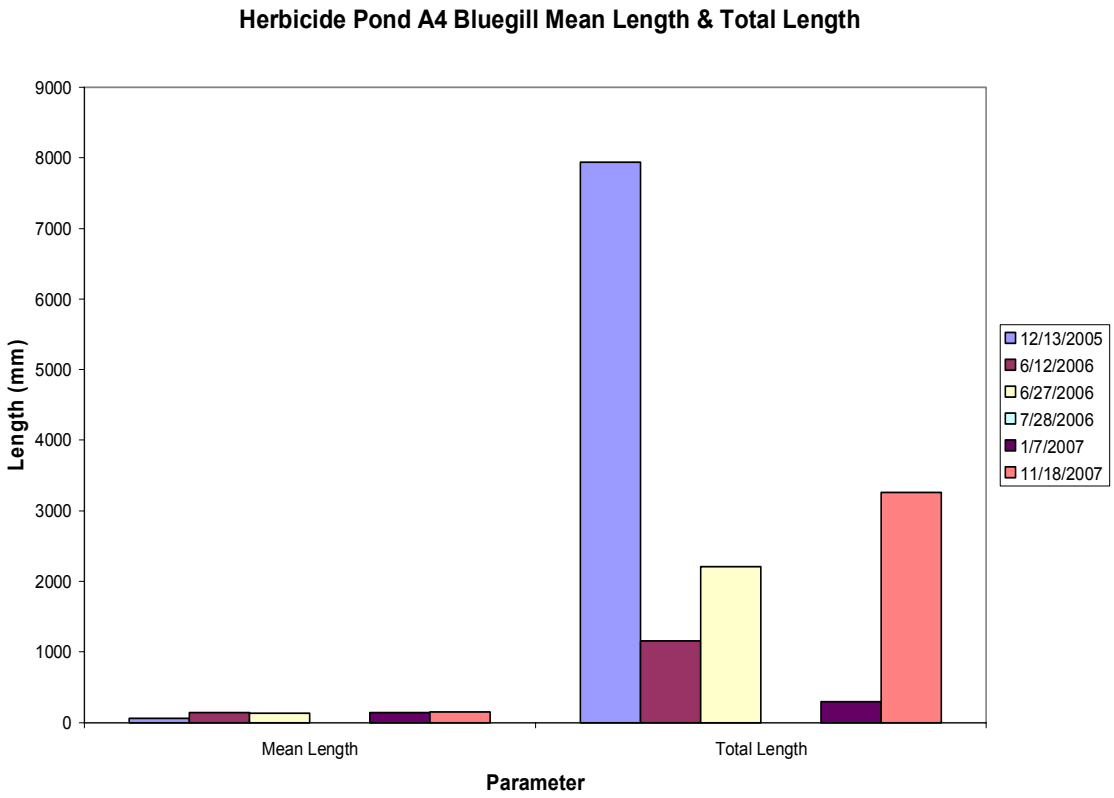

Figure A-94. Bluegill mean length and total length for herbicide pond A4

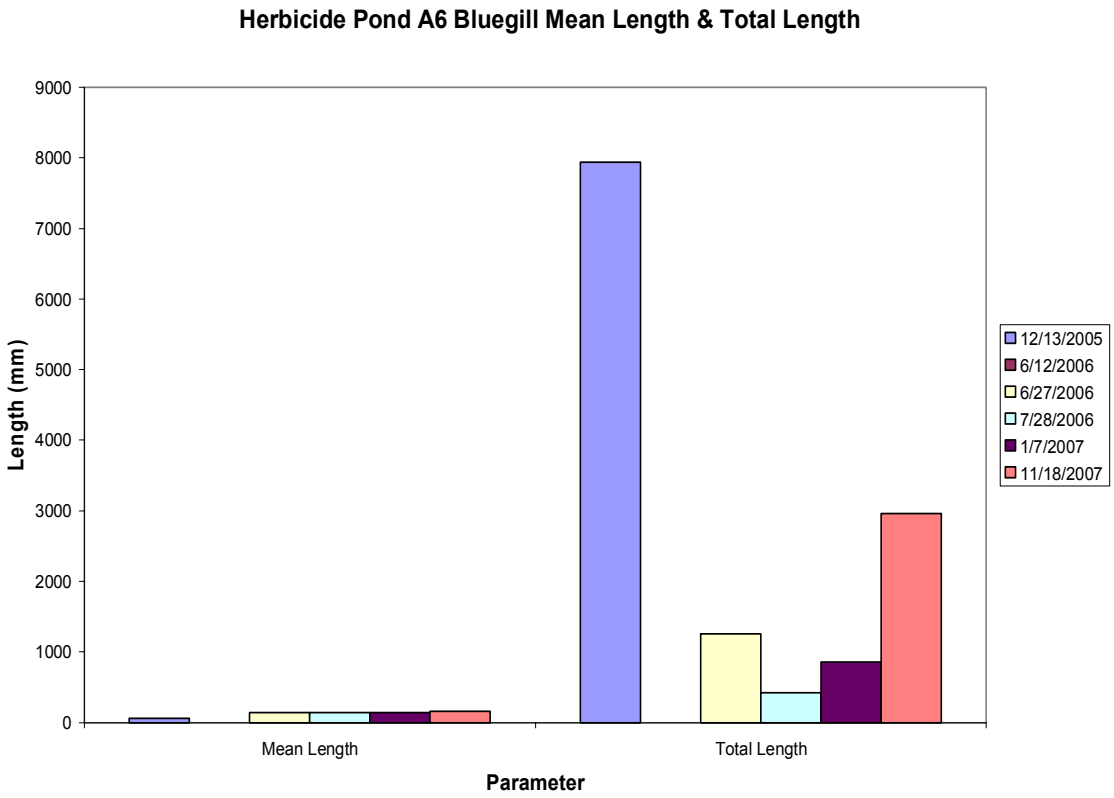

Figure A-95. Bluegill mean length and total length for herbicide pond A6

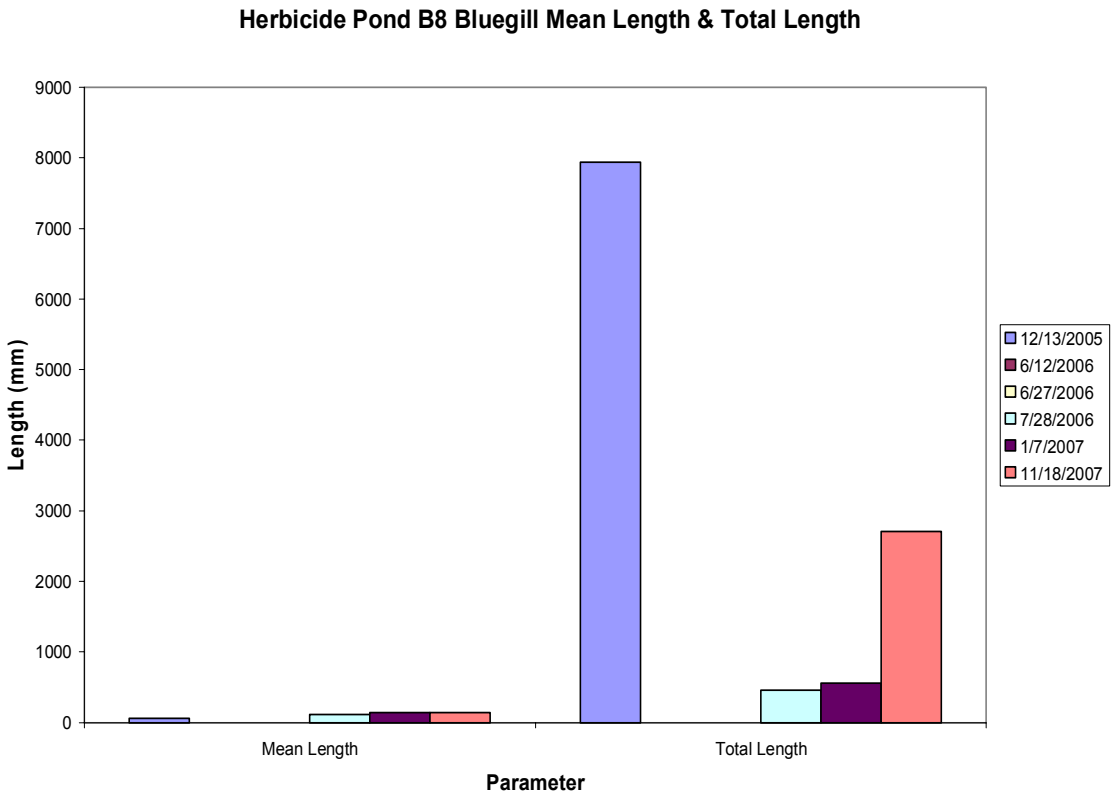

Figure A-96. Bluegill mean length and total length for herbicide pond B8

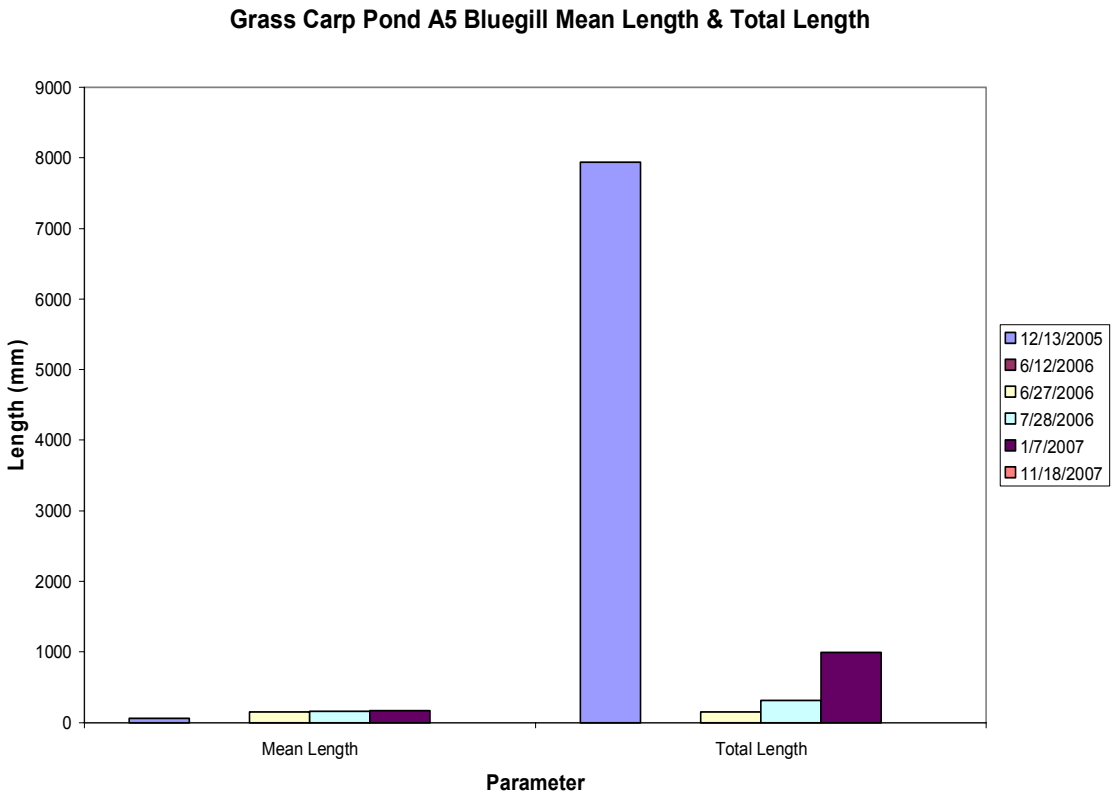

Figure A-97. Bluegill mean length and total length for grass carp pond A5

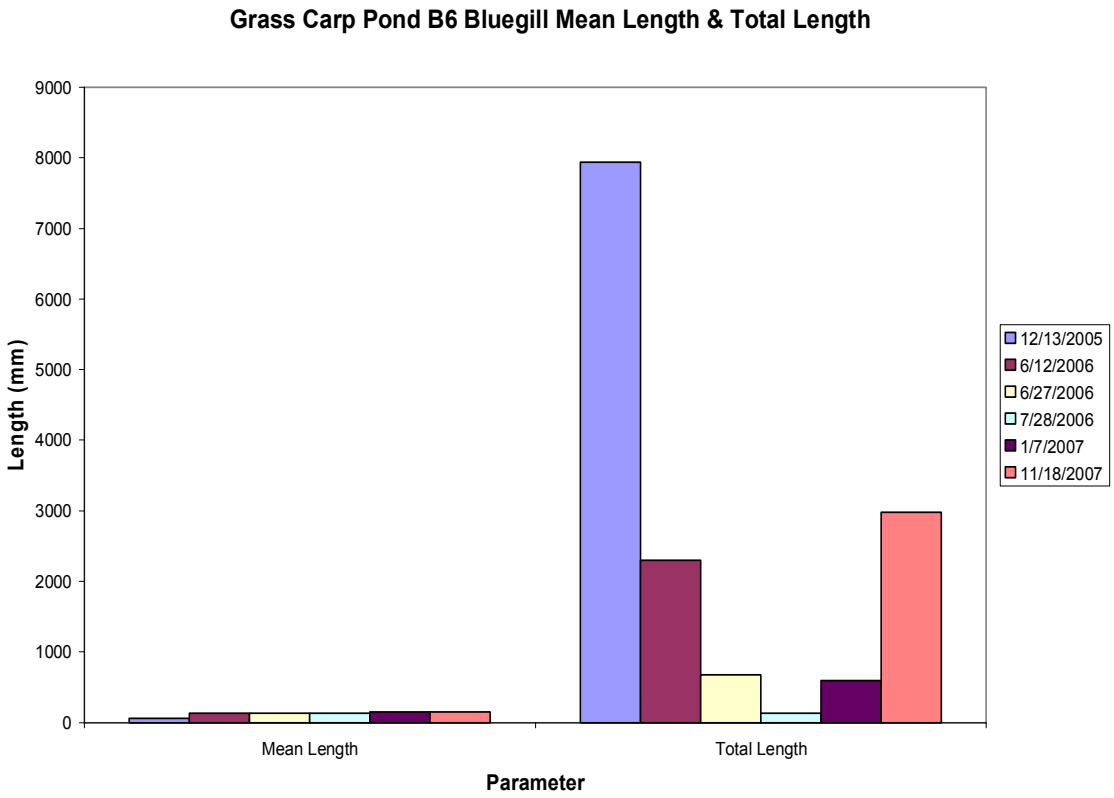

Figure A-98. Bluegill mean length and total length for grass carp pond B6

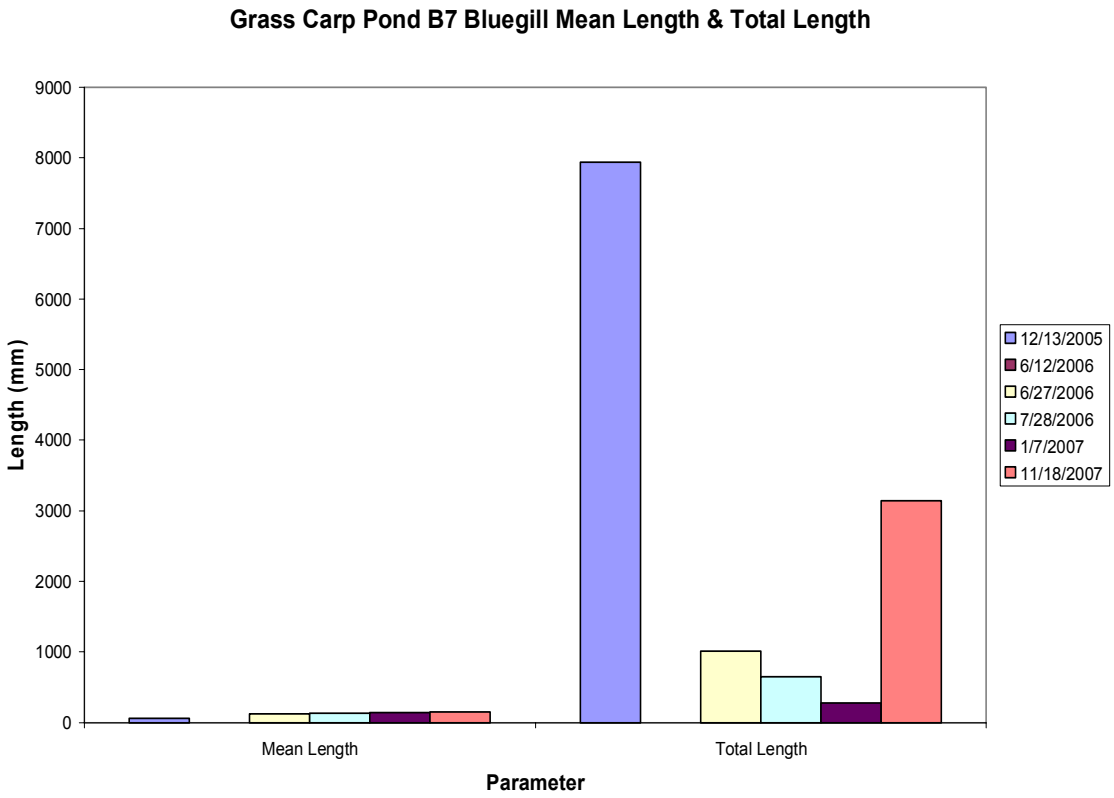

Figure A-99. Bluegill mean length and total length for grass carp pond B7

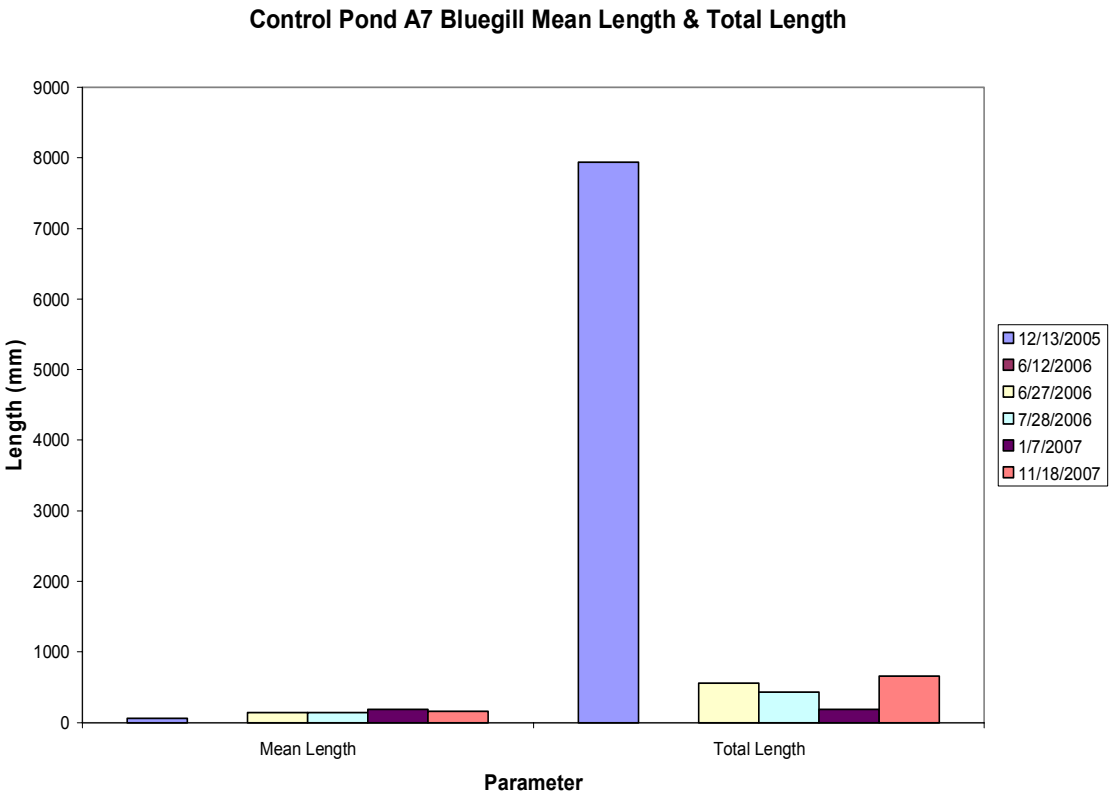

Figure A-100. Bluegill mean length and total length for control pond A7

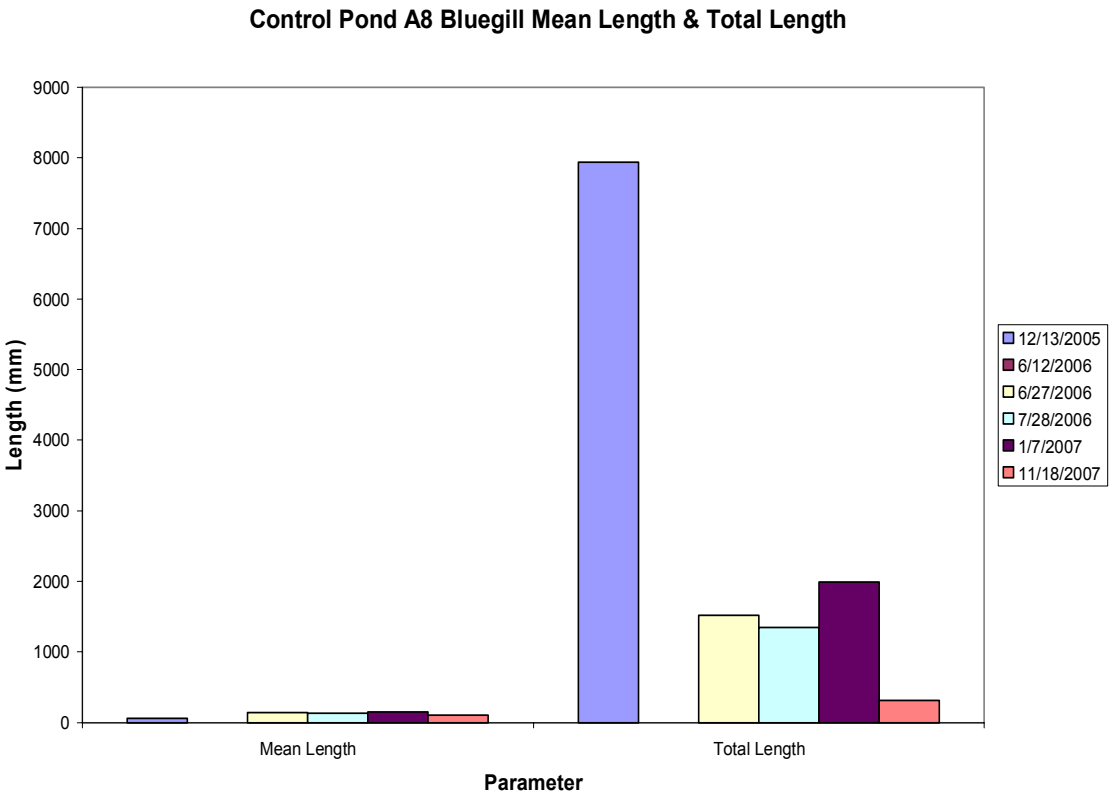

Figure A-101. Bluegill mean length and total length for control pond A8

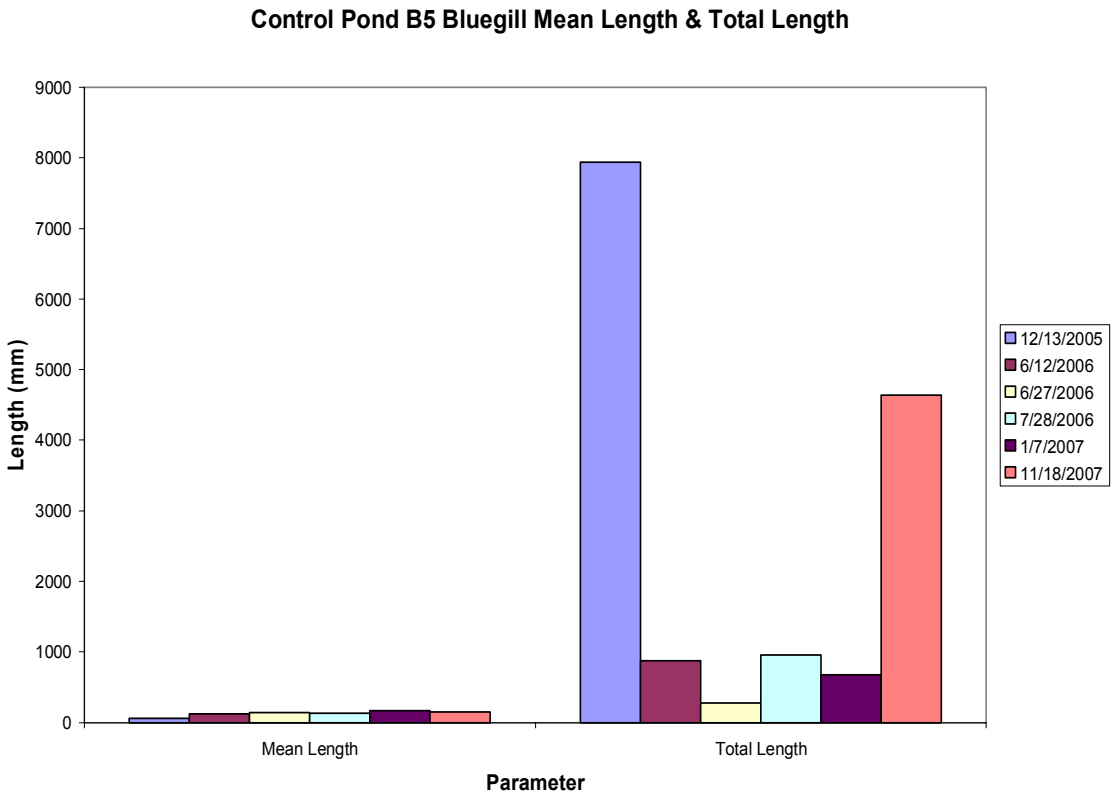

Figure A-102. Bluegill mean length and total length for control pond B5

APPENDIX B

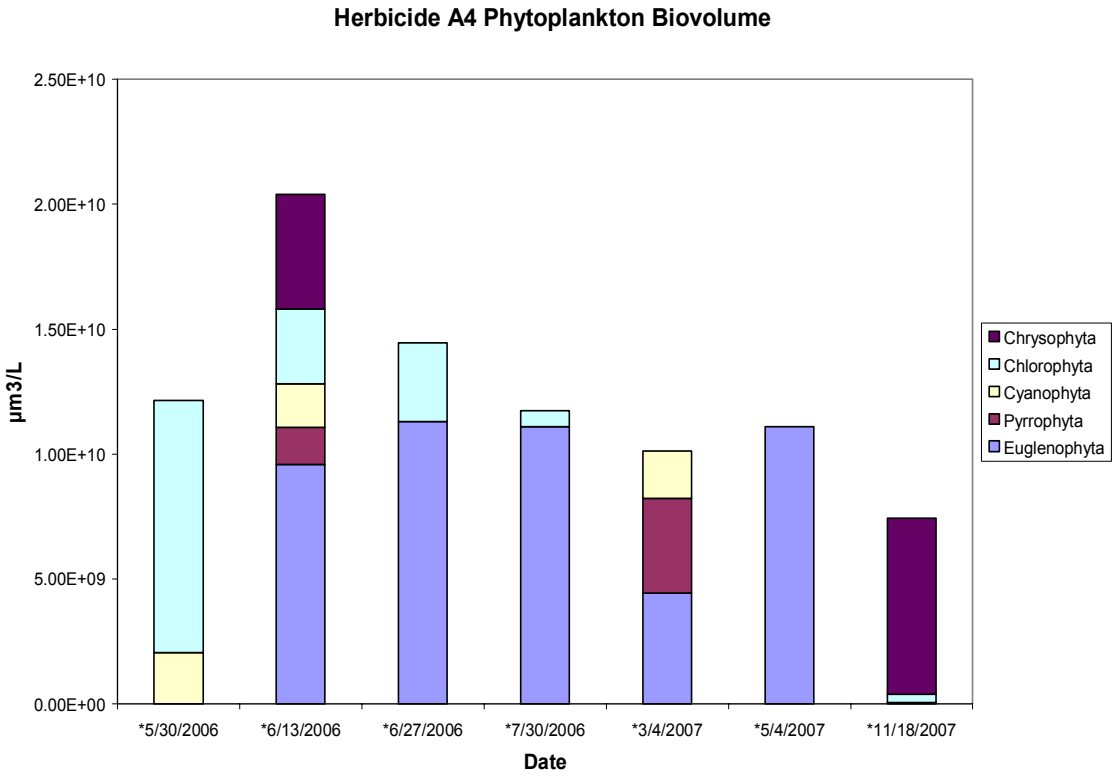

Figure B-1. Phytoplankton biovolume for herbicide pond A4

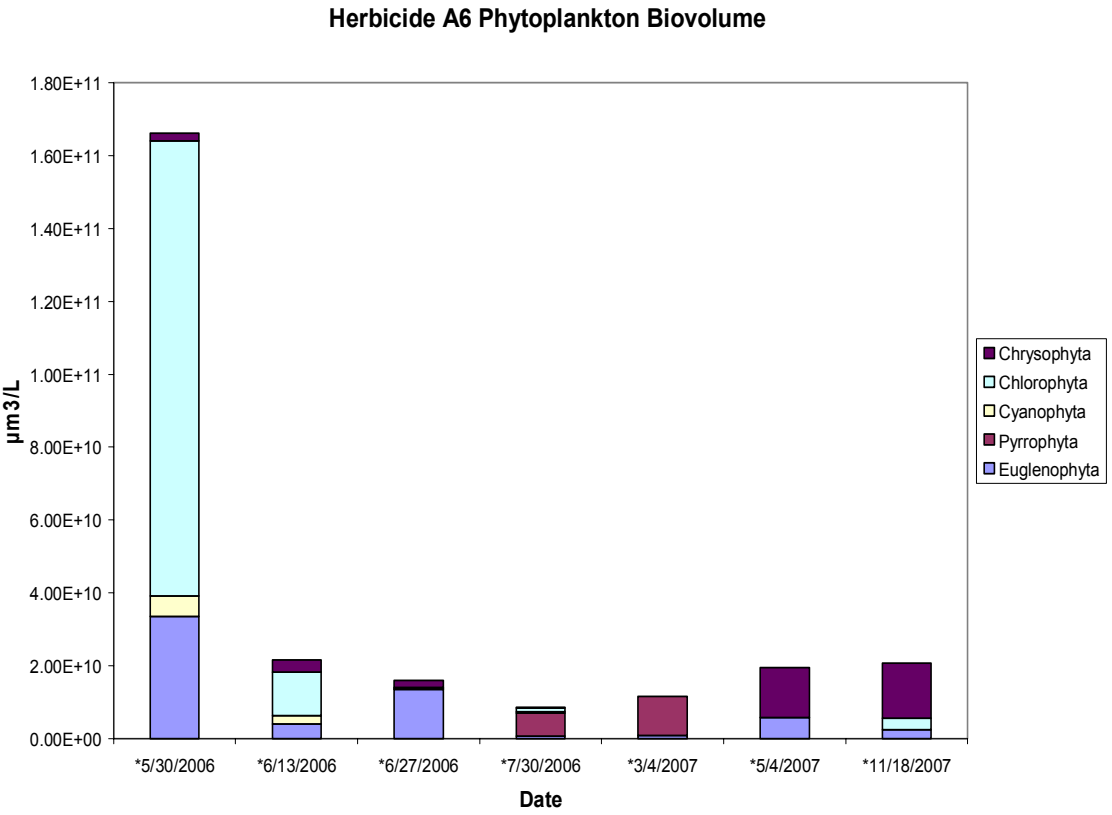

Figure B-2. Phytoplankton biovolume for herbicide pond A6

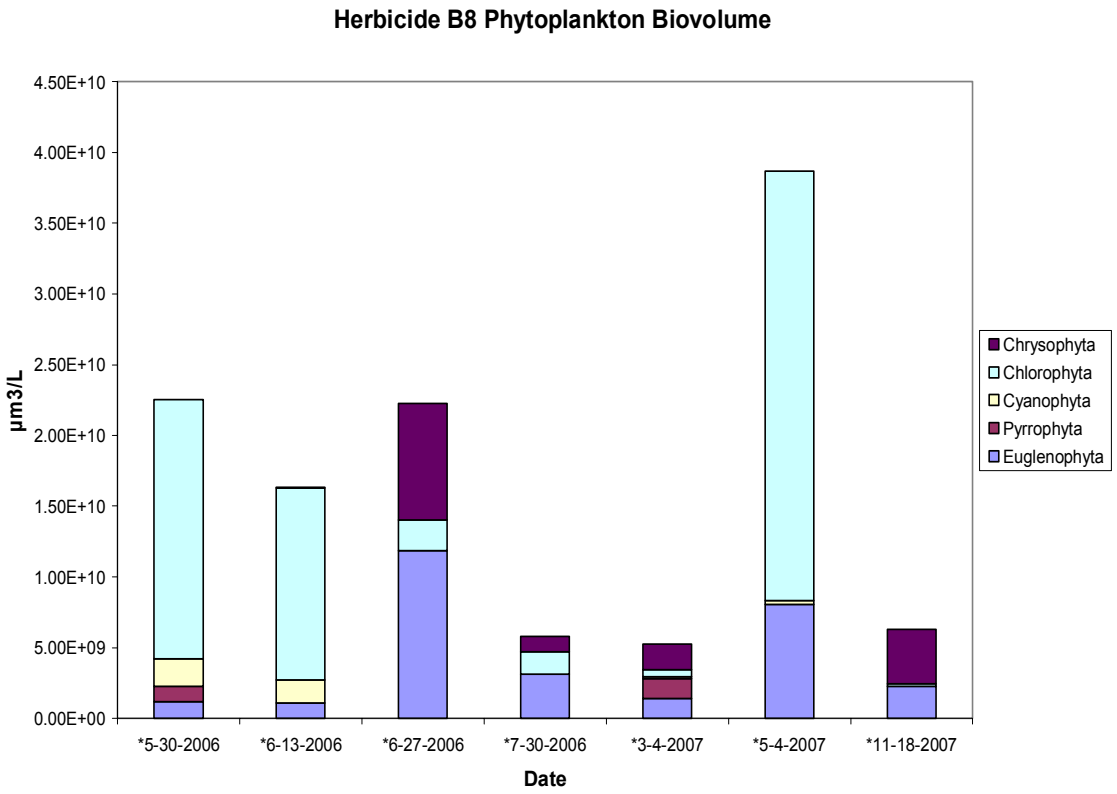

Figure B-3. Phytoplankton biovolume for herbicide pond B8

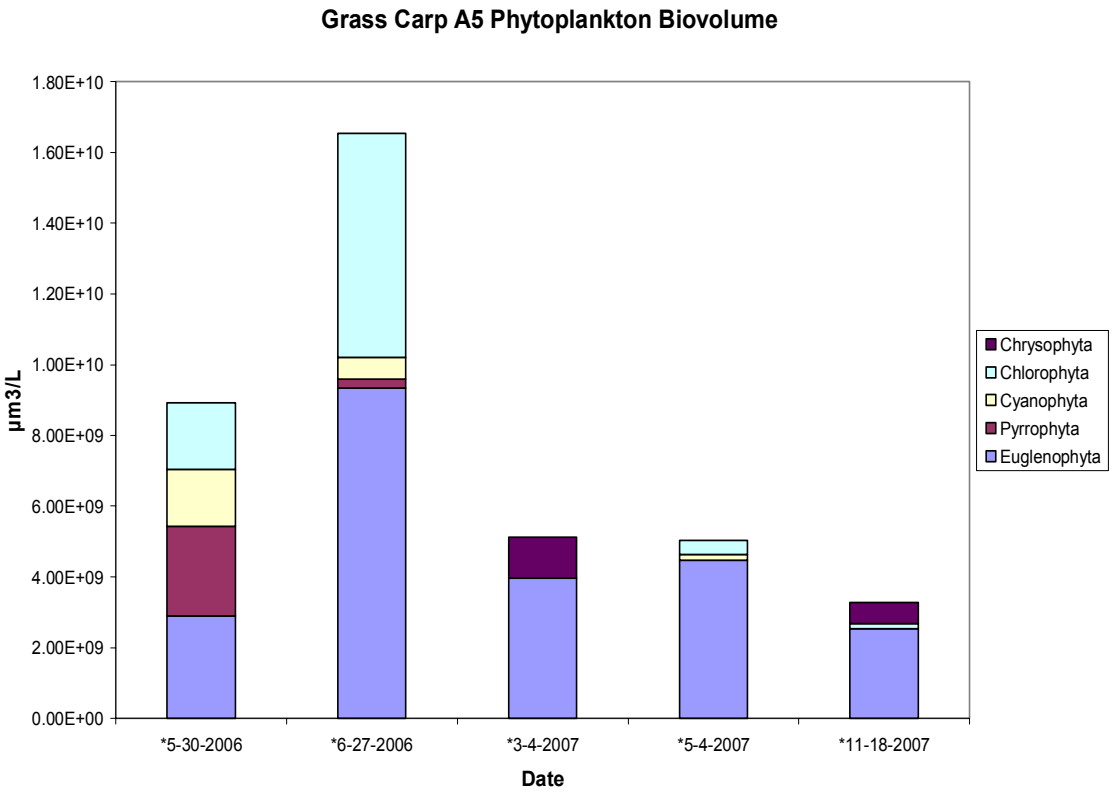

Figure B-4. Phytoplankton biovolume for grass carp pond A5

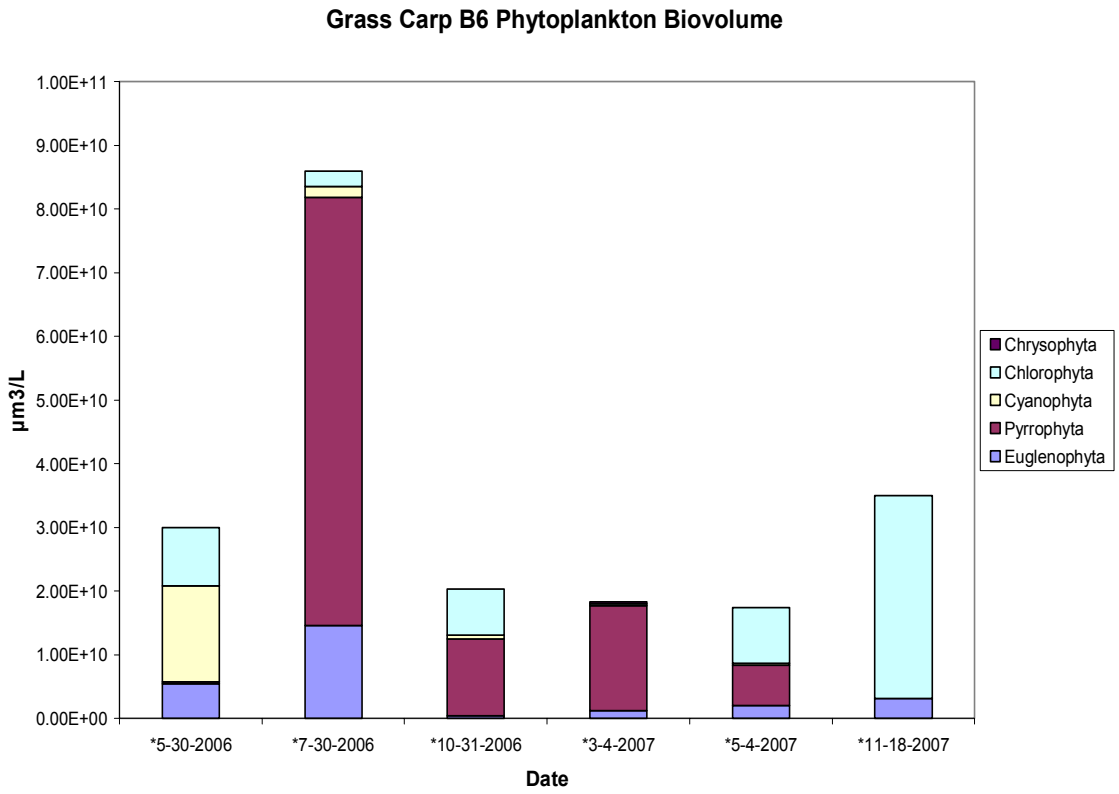

Figure B-5. Phytoplankton biovolume for grass carp pond B6

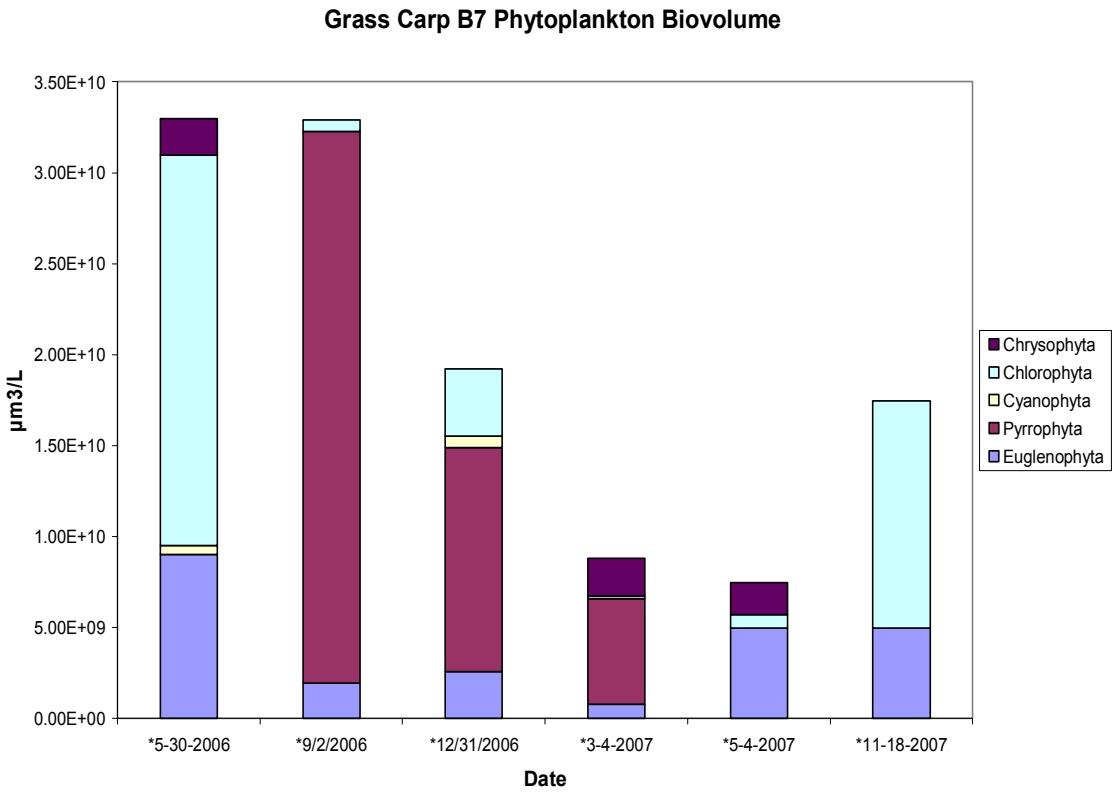

Figure B-6. Phytoplankton biovolume for grass carp pond B7

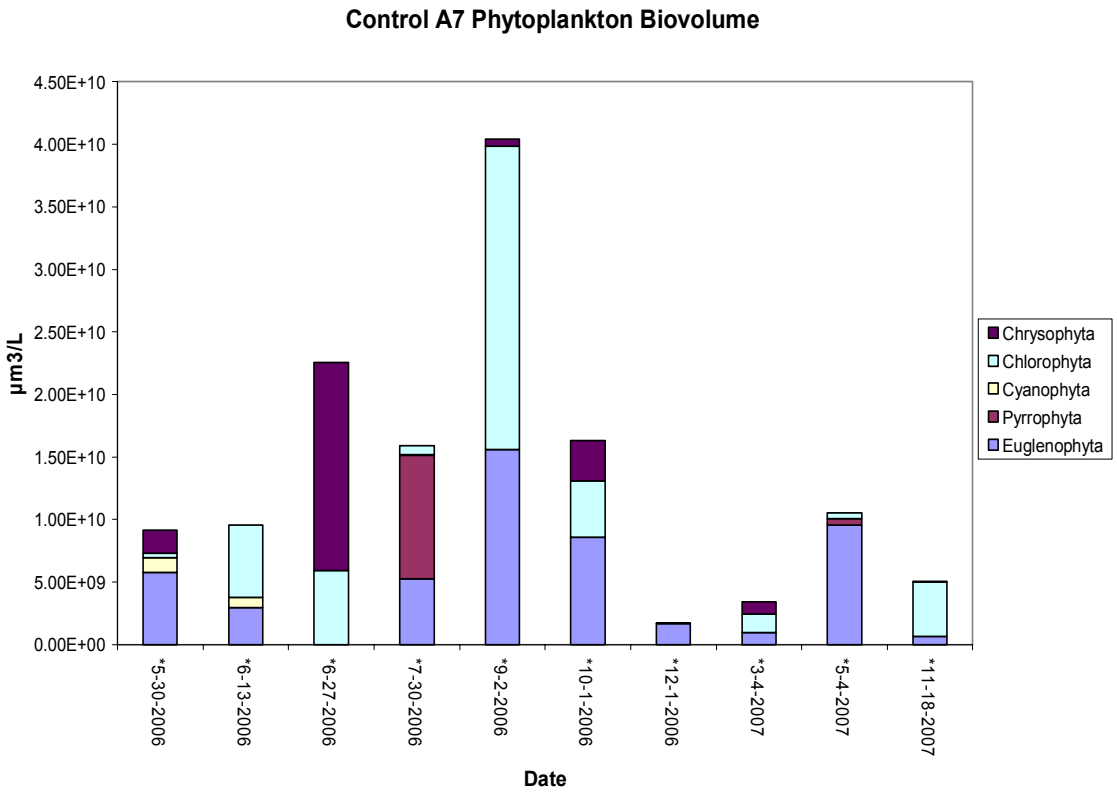

Figure B-7. Phytoplankton biovolume for control pond A7

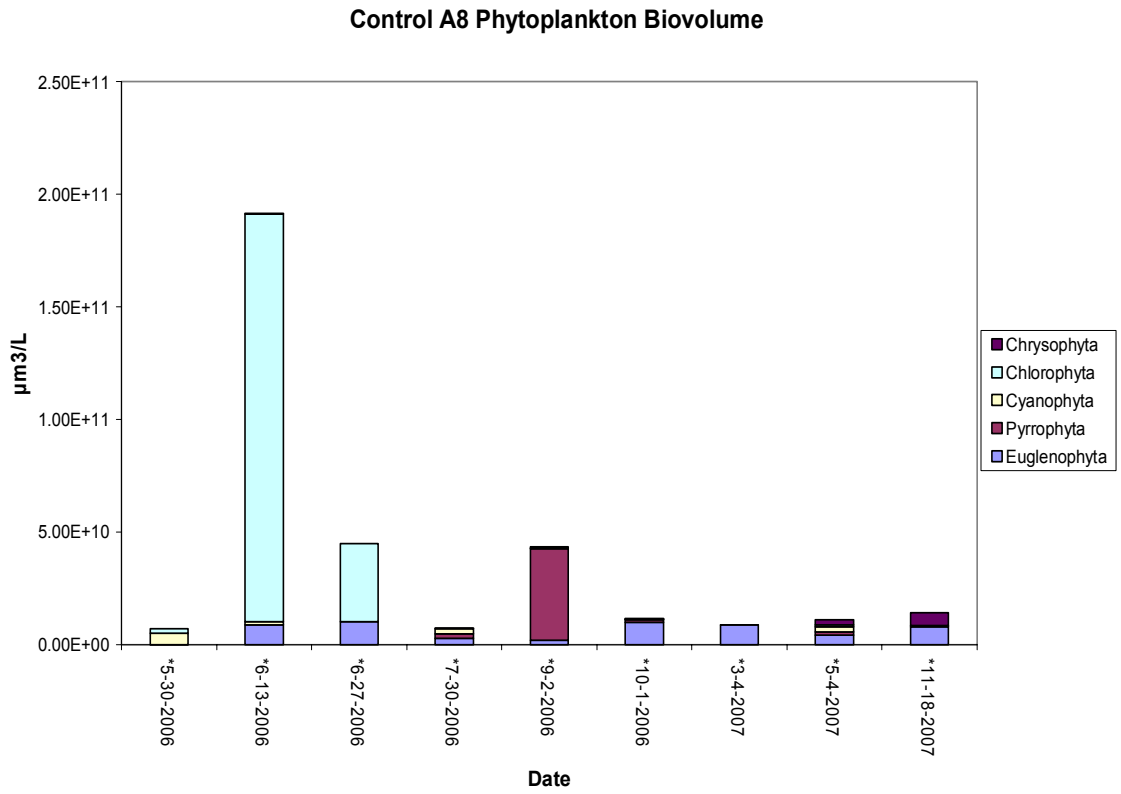

Figure B-8. Phytoplankton biovolume for control pond A8

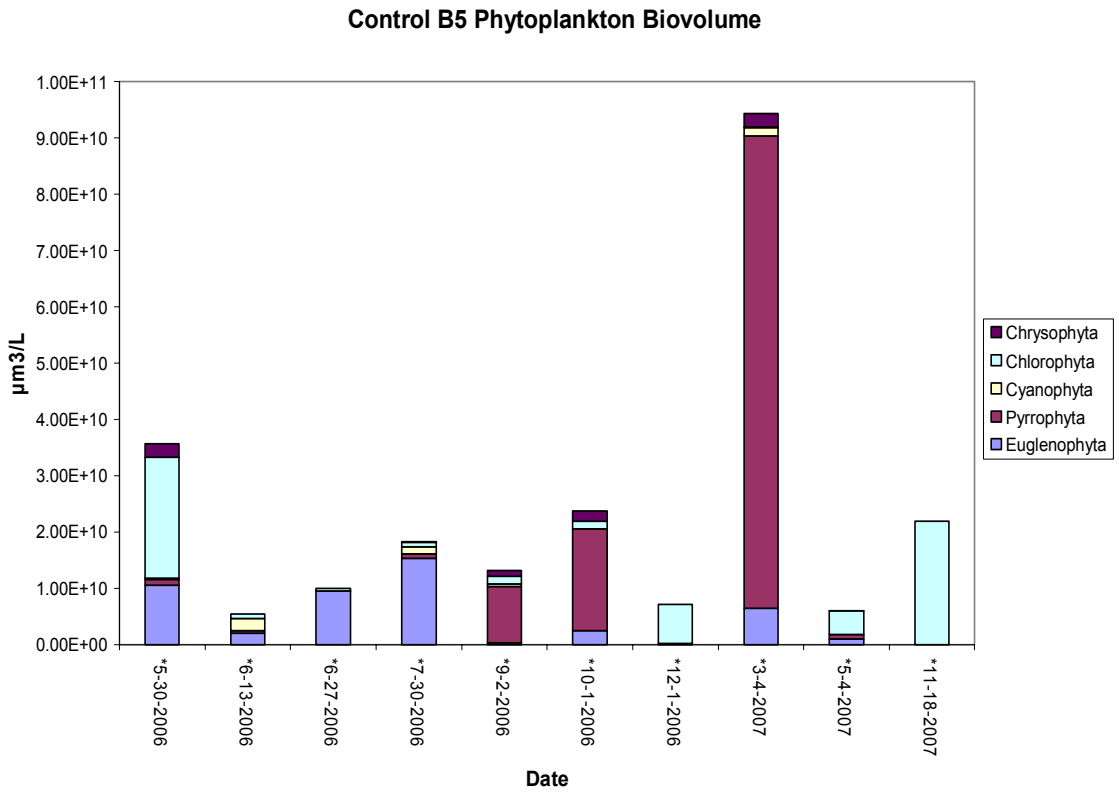

Figure B-9. Phytoplankton biovolume for control pond B5

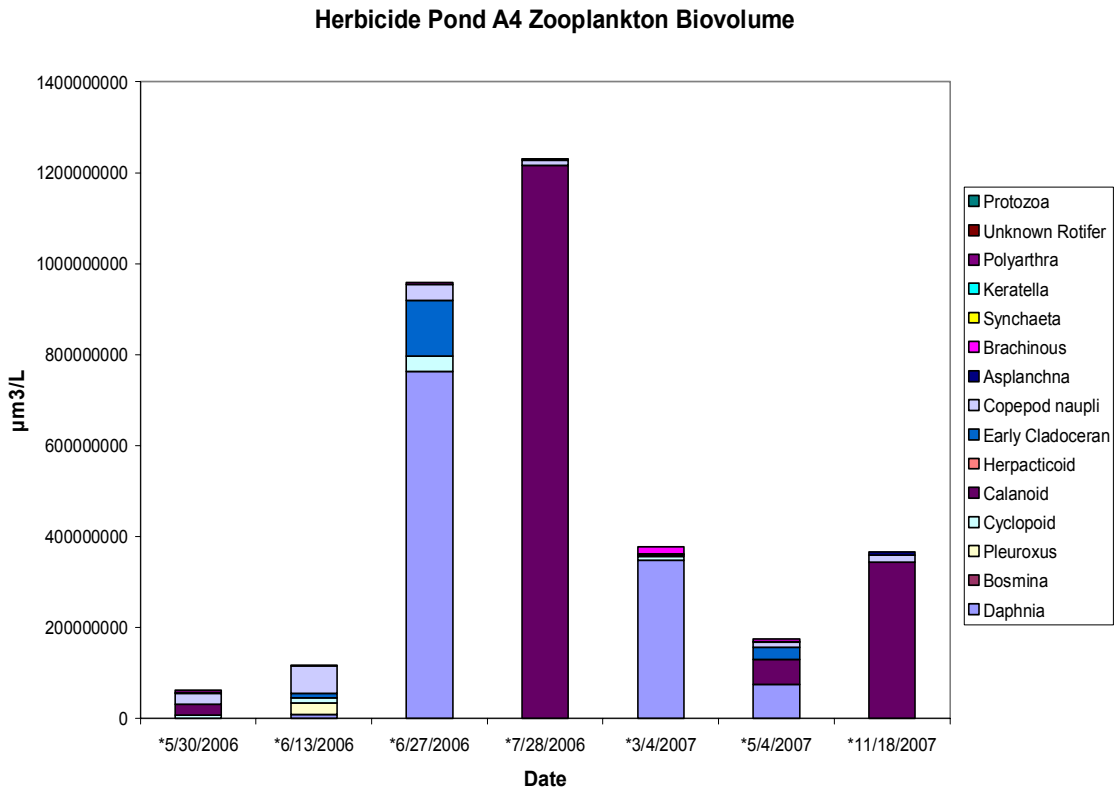

Figure B-10. Zooplankton biovolume for herbicide pond A4

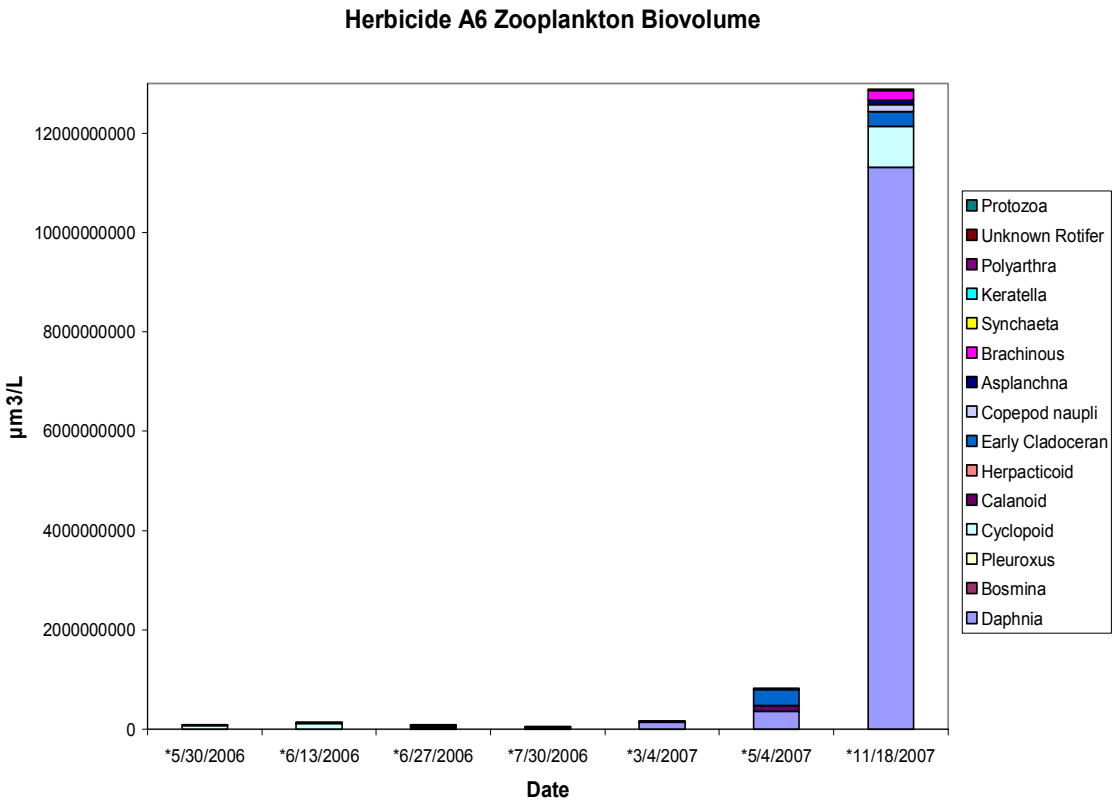

Figure B-11. Zooplankton biovolume for herbicide pond A6

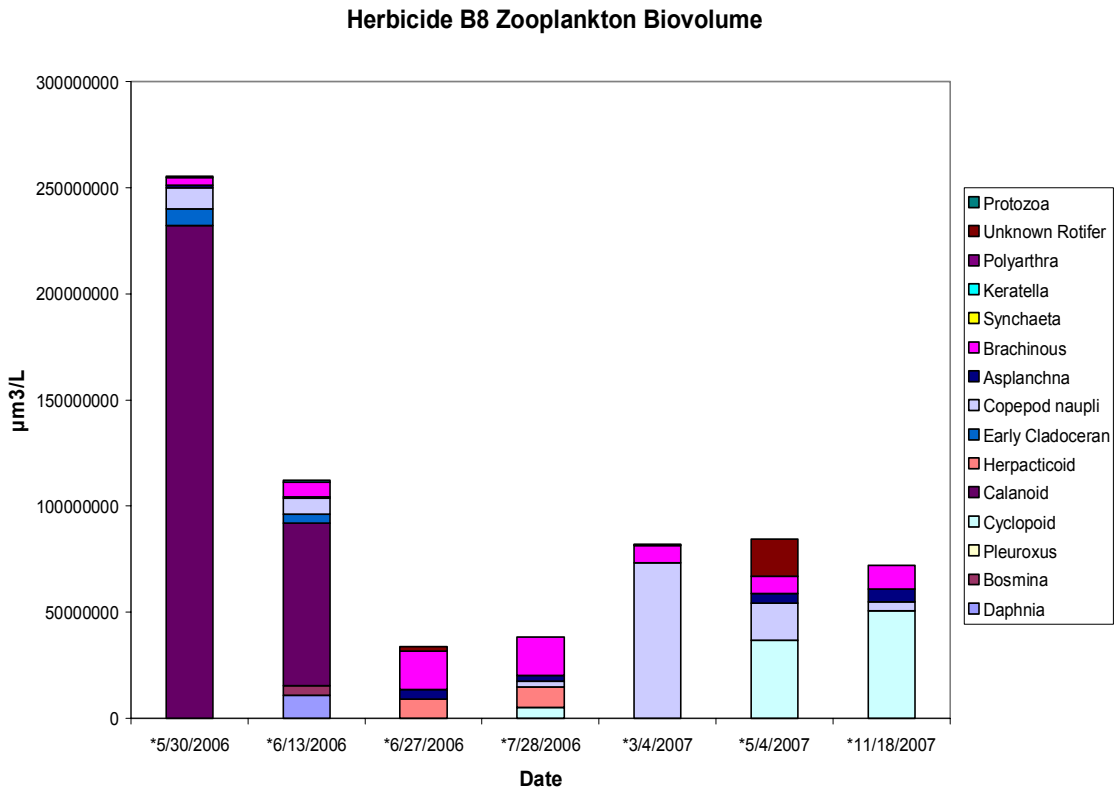

Figure B-12. Zooplankton biovolume for herbicide pond B8

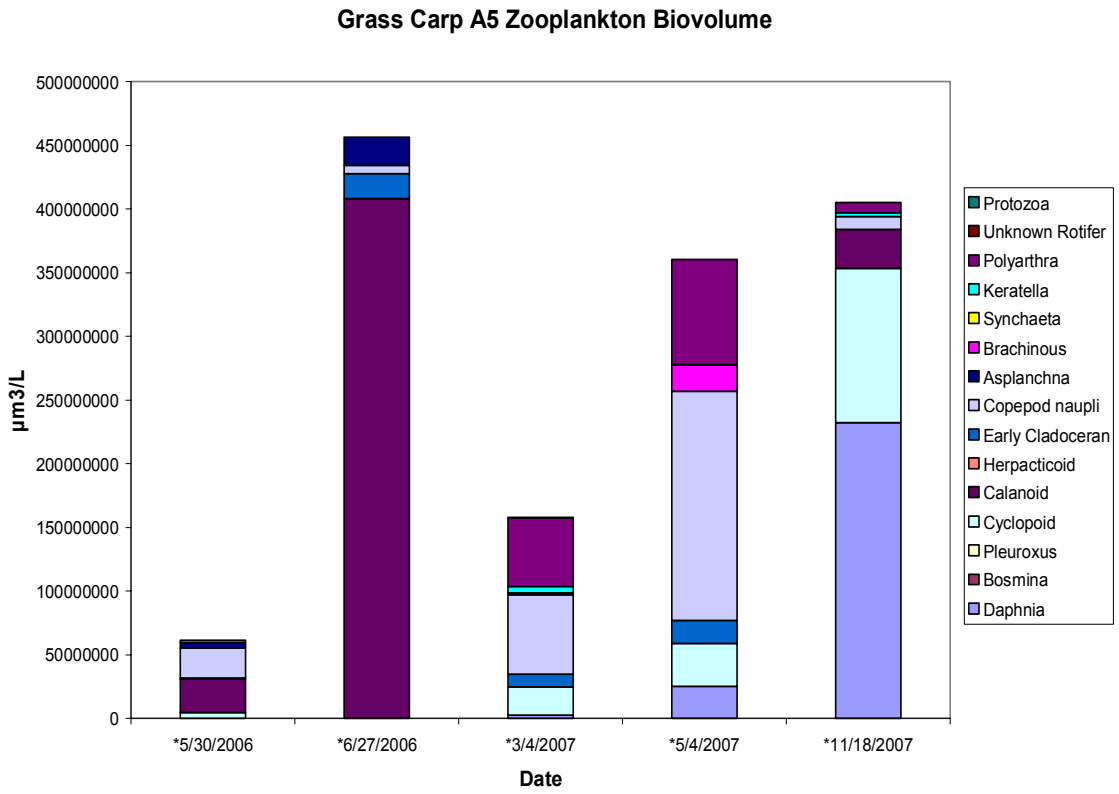

Figure B-13. Zooplankton biovolume for grass carp pond A5

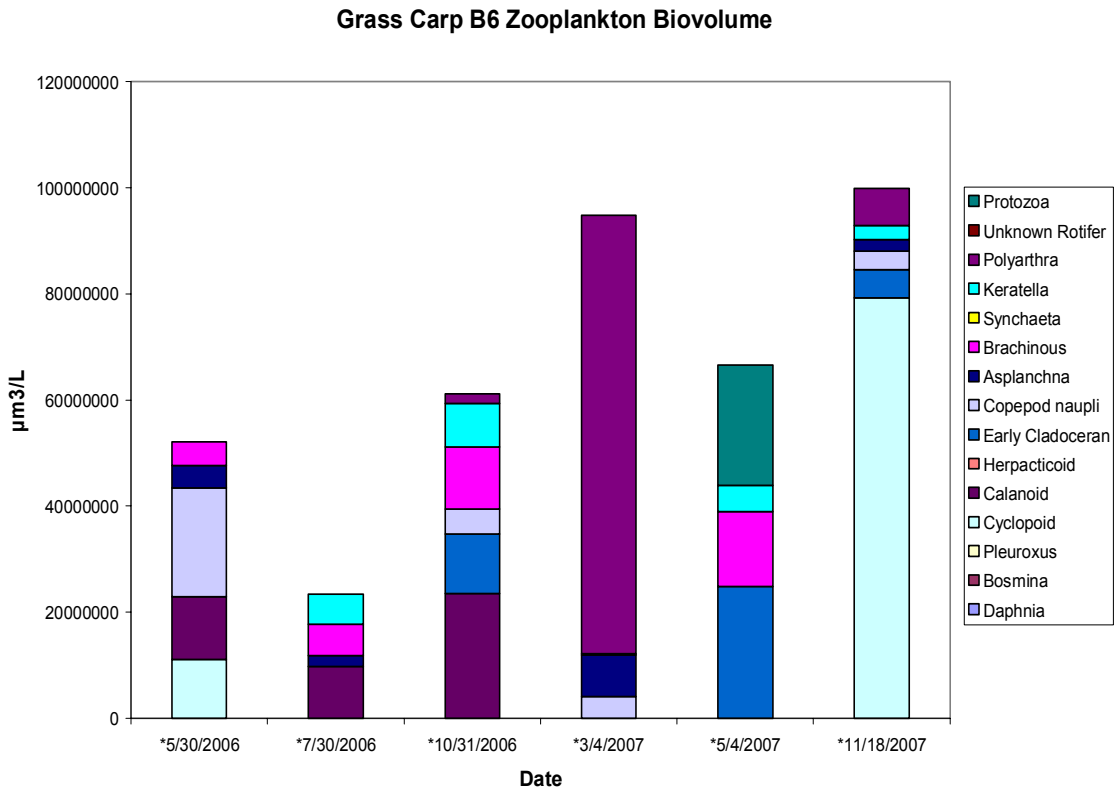

Figure B-14. Zooplankton biovolume for grass carp pond B6

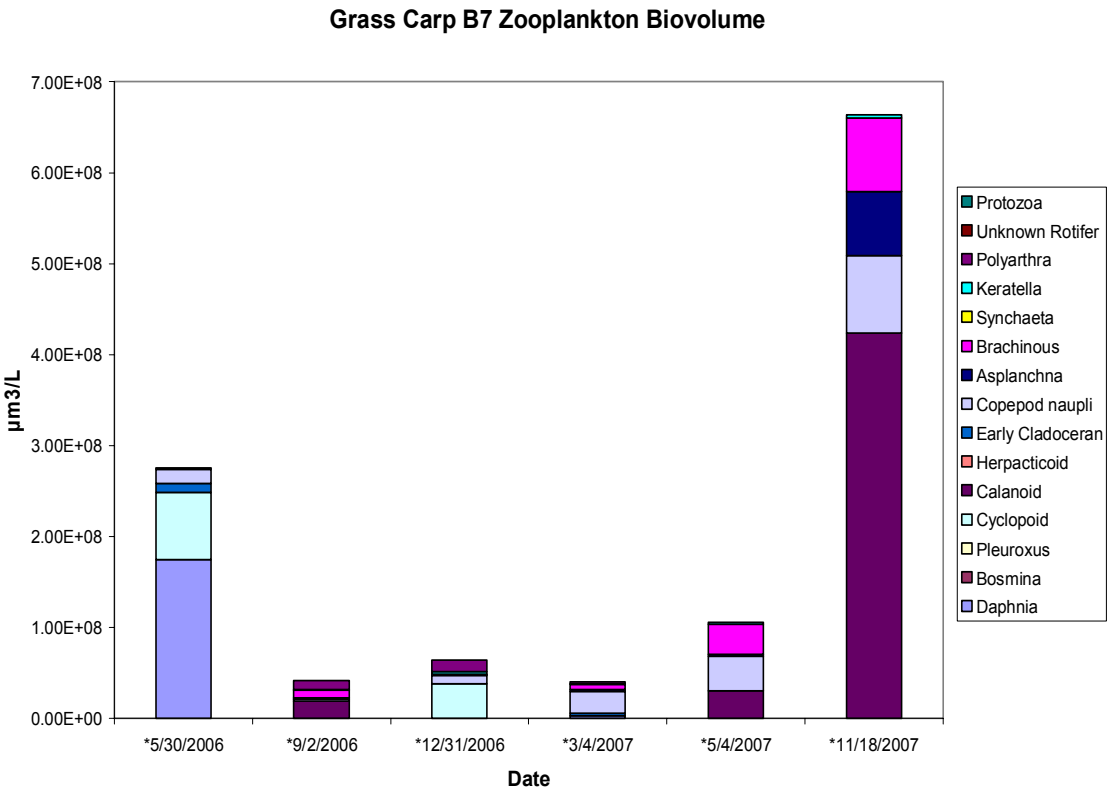

Figure B-15. Zooplankton biovolume for grass carp pond B7

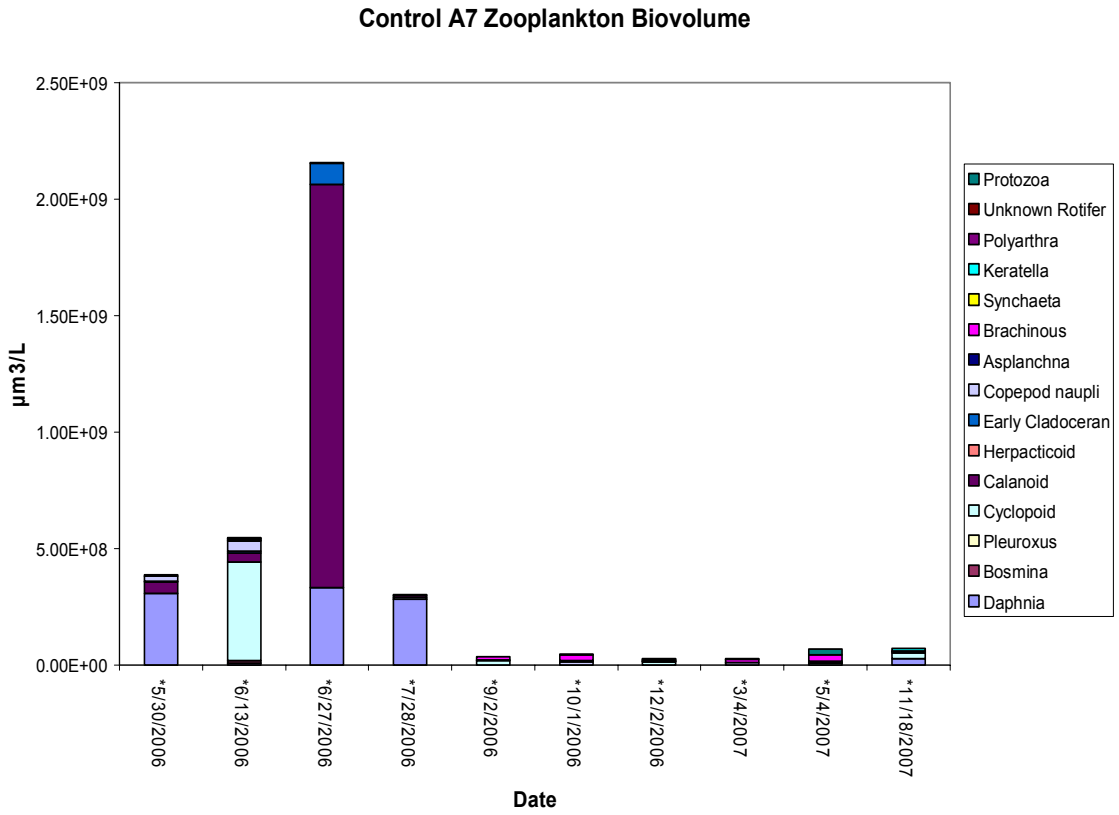

Figure B-16. Zooplankton biovolume for control pond A7

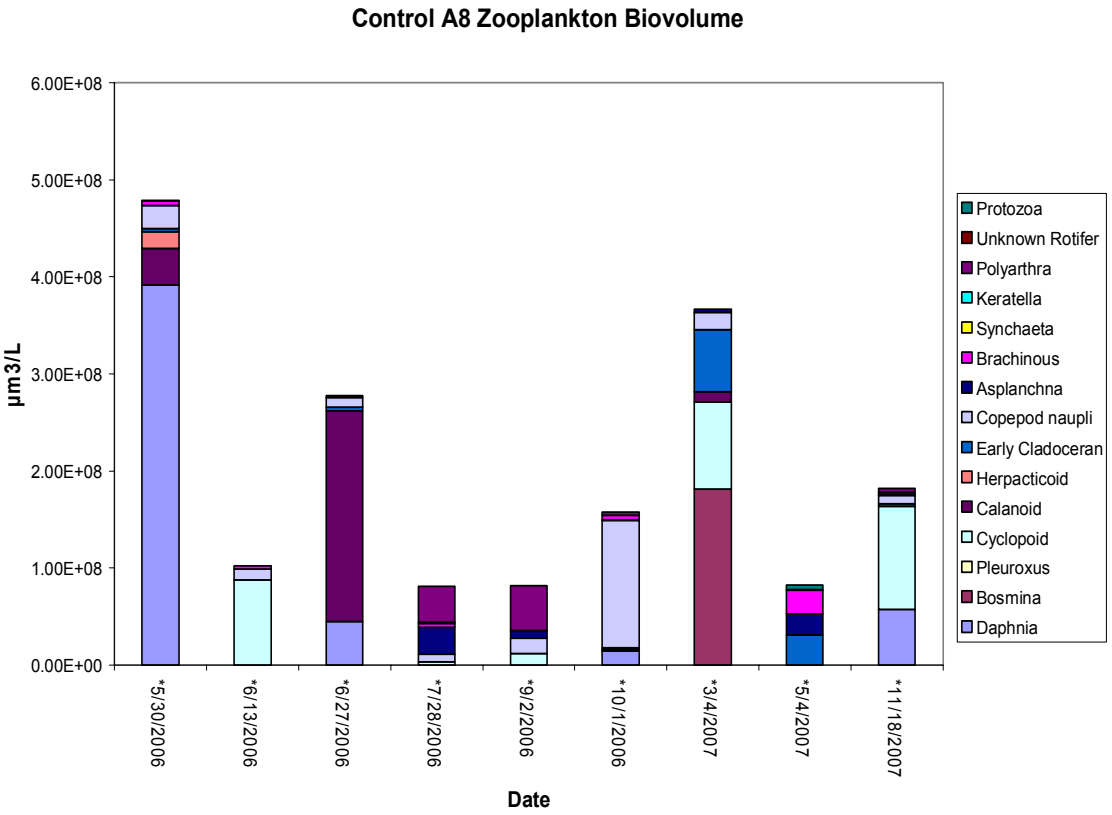

Figure B-17. Zooplankton biovolume for control pond A8

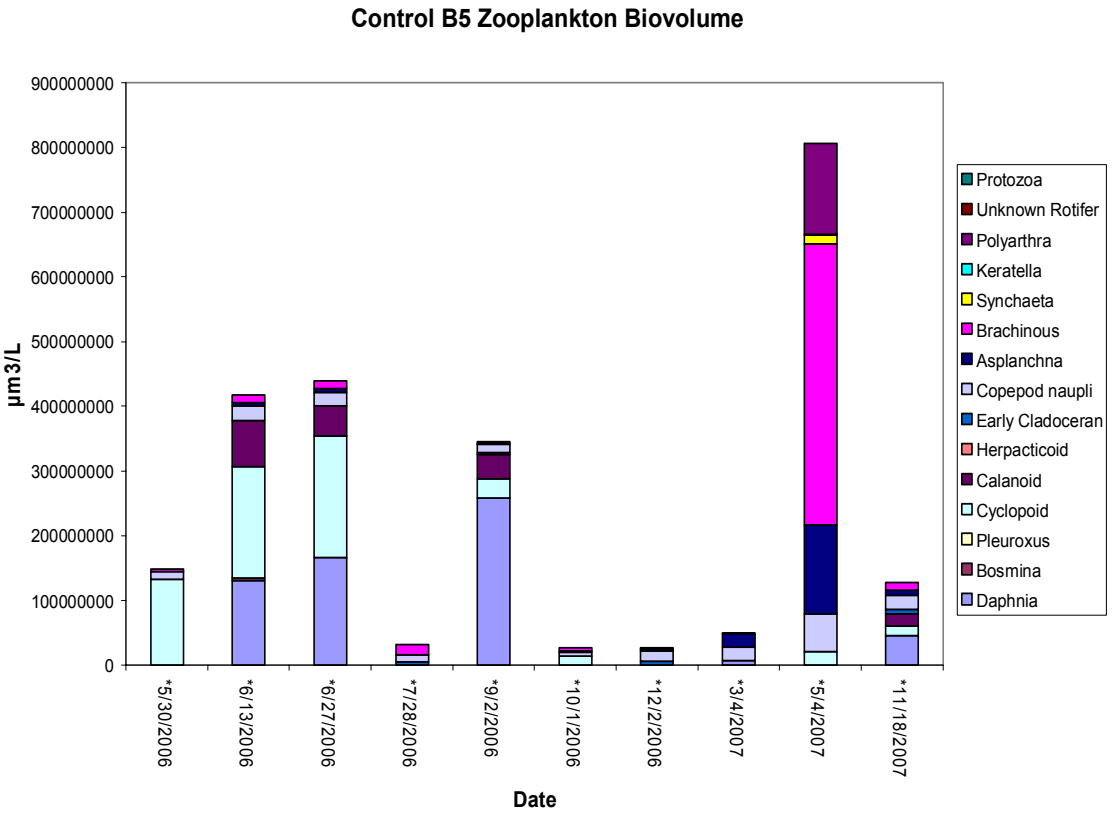

Figure B-18. Zooplankton biovolume for control pond B5

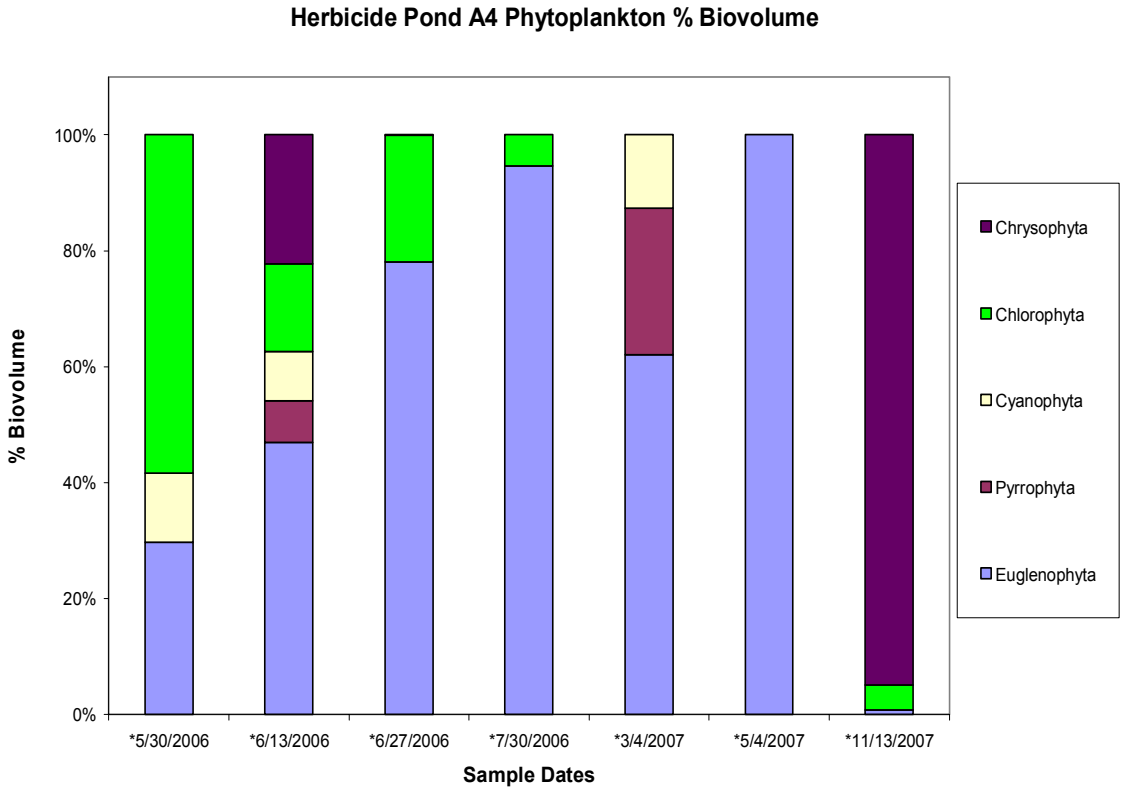

Figure B-19. Phytoplankton % biovolume for herbicide pond A4

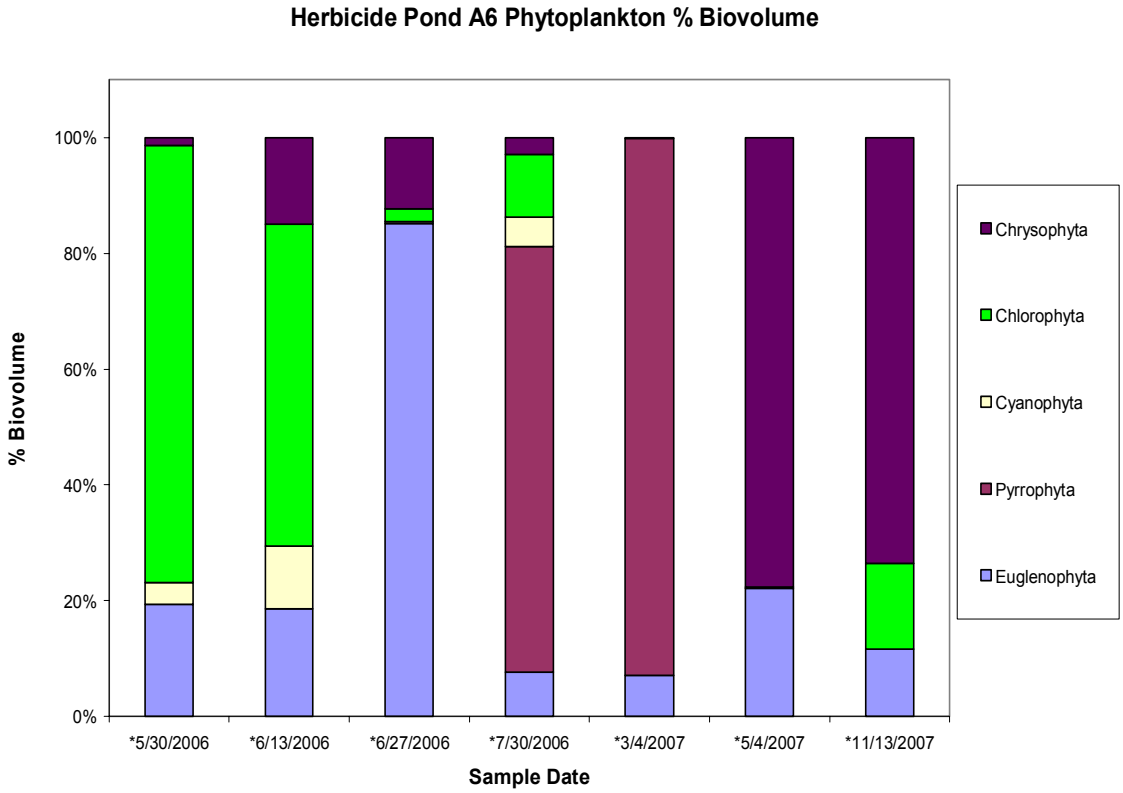

Figure B-20. Phytoplankton % biovolume for herbicide pond A6

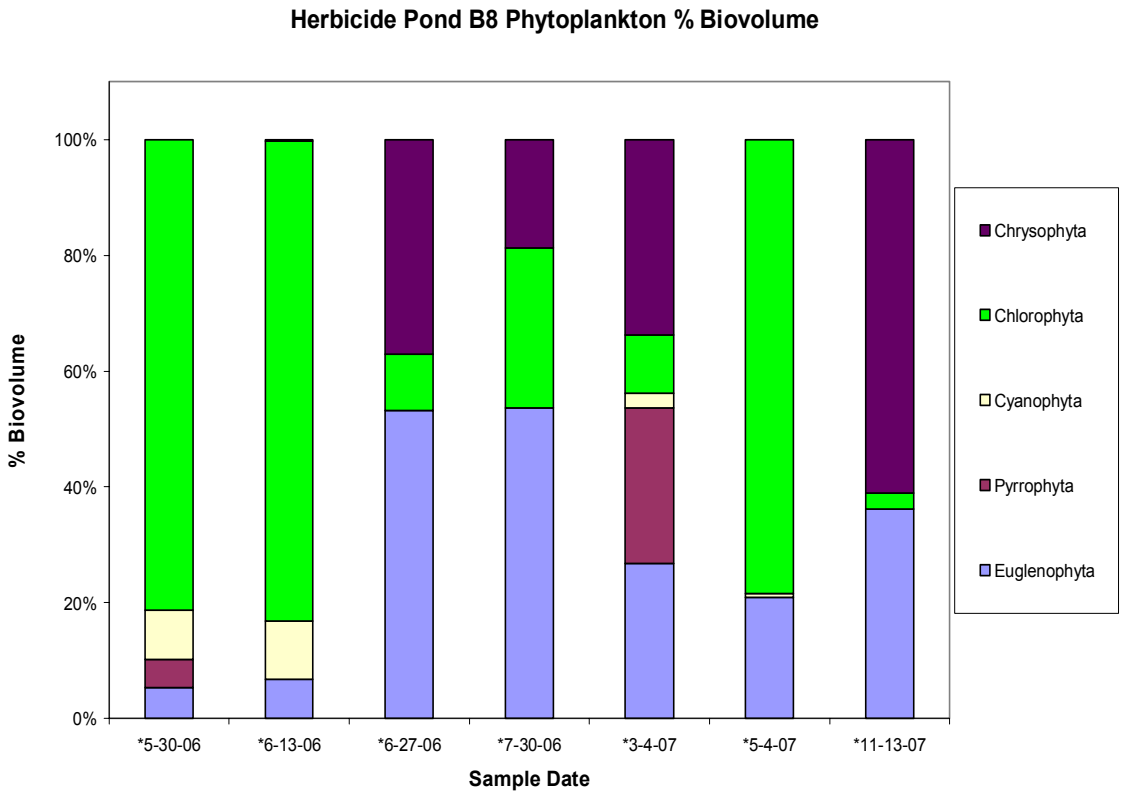

Figure B-21. Phytoplankton % biovolume for herbicide pond B8

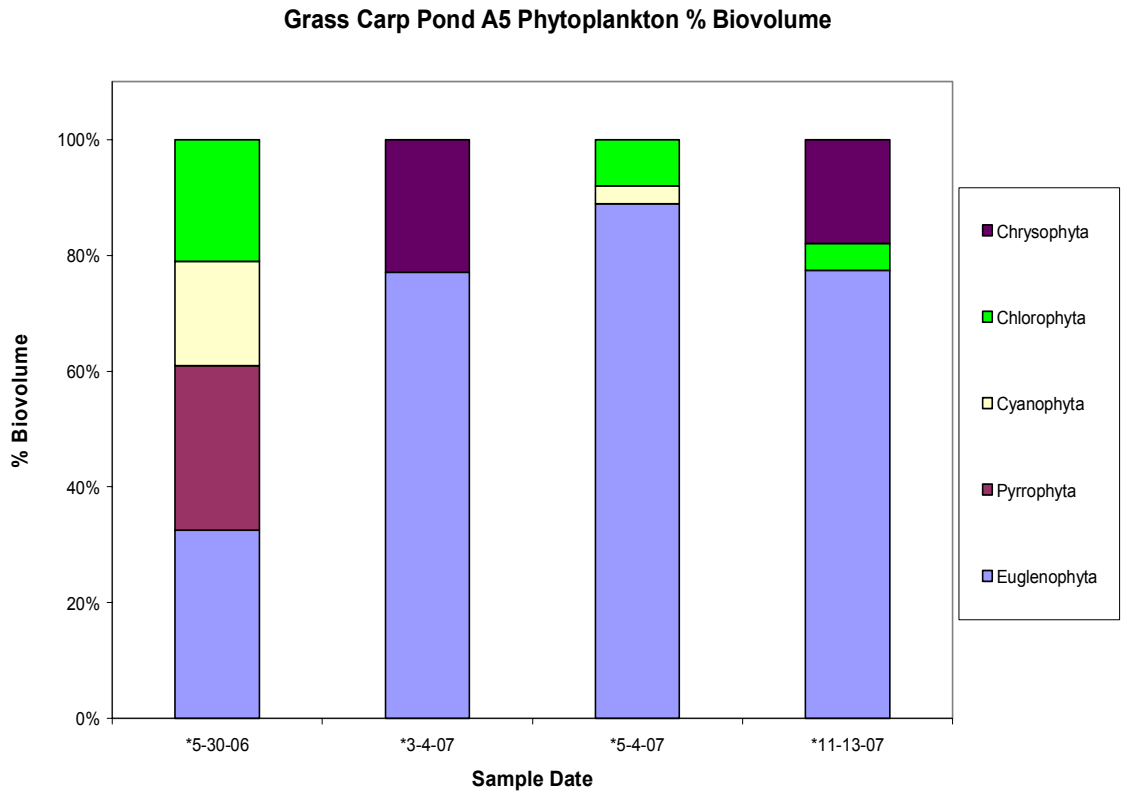

Figure B-22. Phytoplankton % biovolume for grass carp pond A5

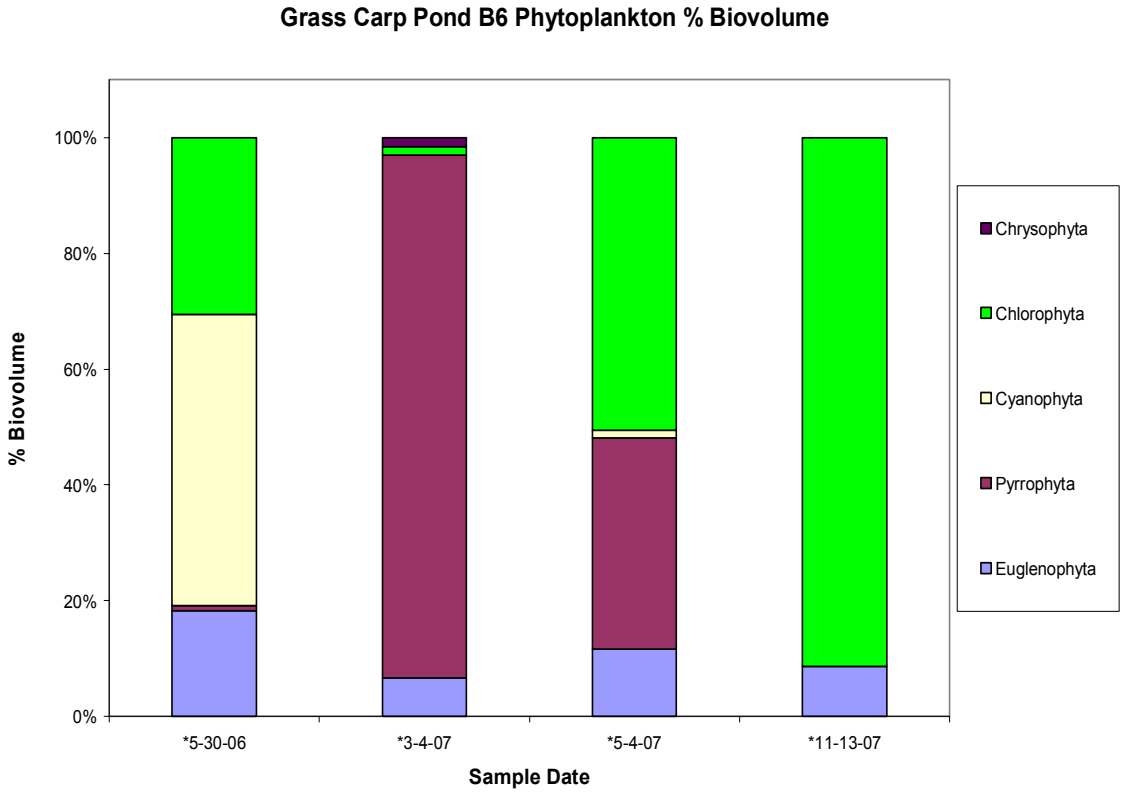

Figure B-23. Phytoplankton % biovolume for grass carp pond B6

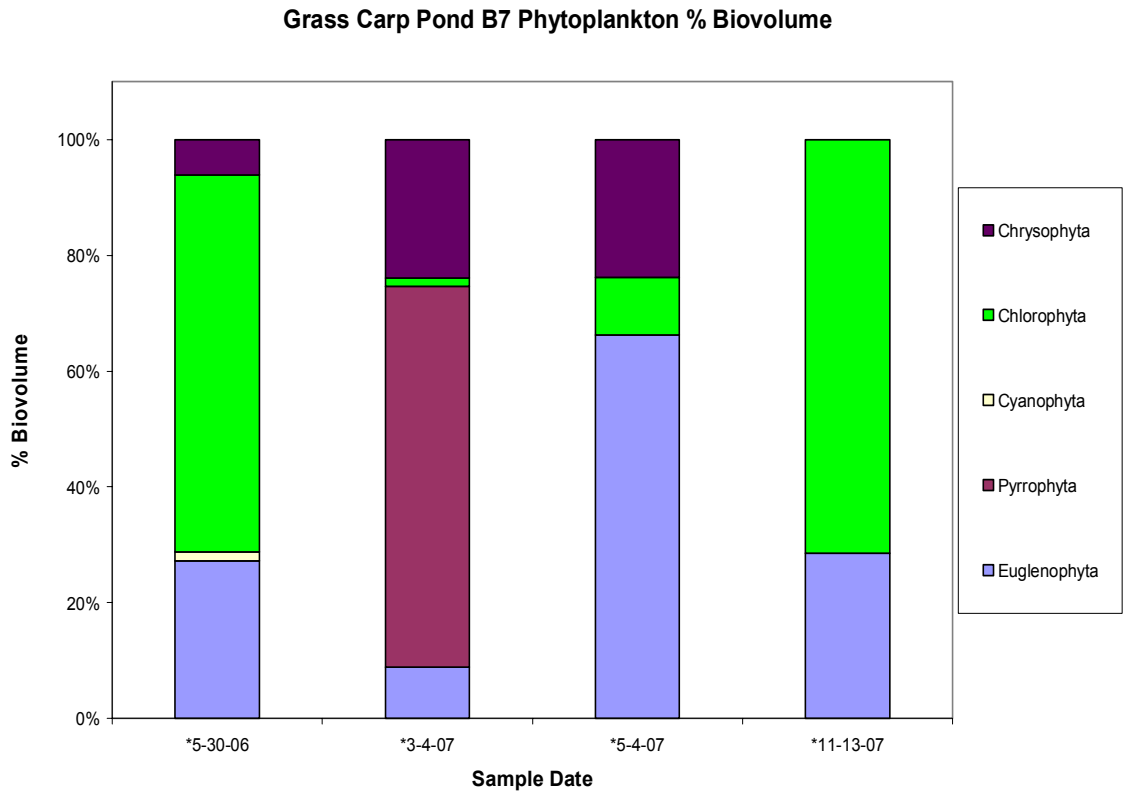

Figure B-24. Phytoplankton % biovolume for grass carp pond B7

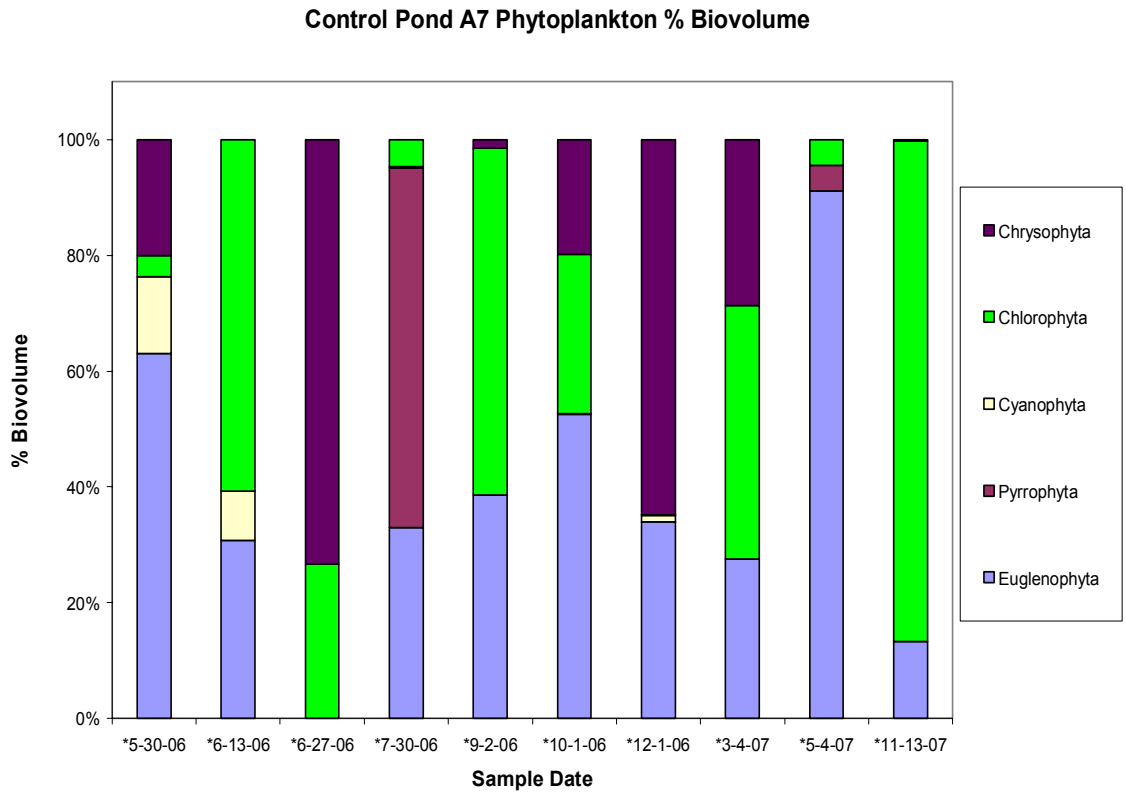

Figure B-25. Phytoplankton % biovolume for control pond A7

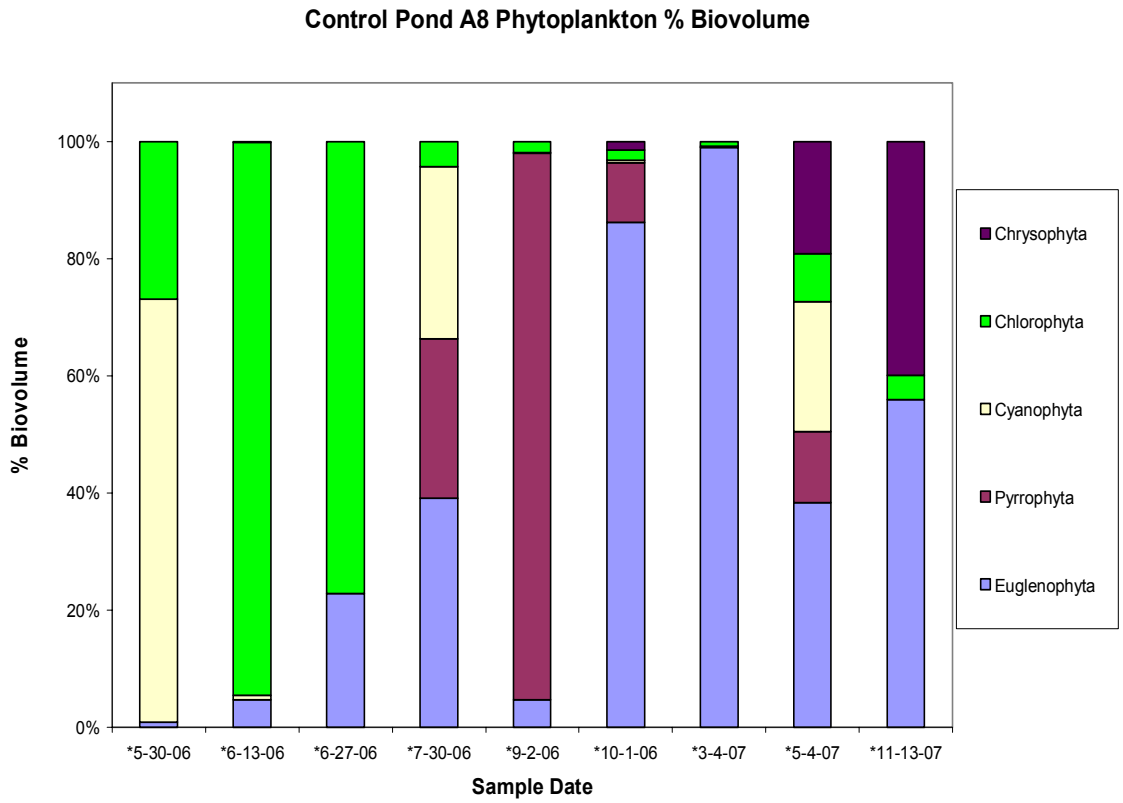

Figure B-26. Phytoplankton % biovolume for control pond A8

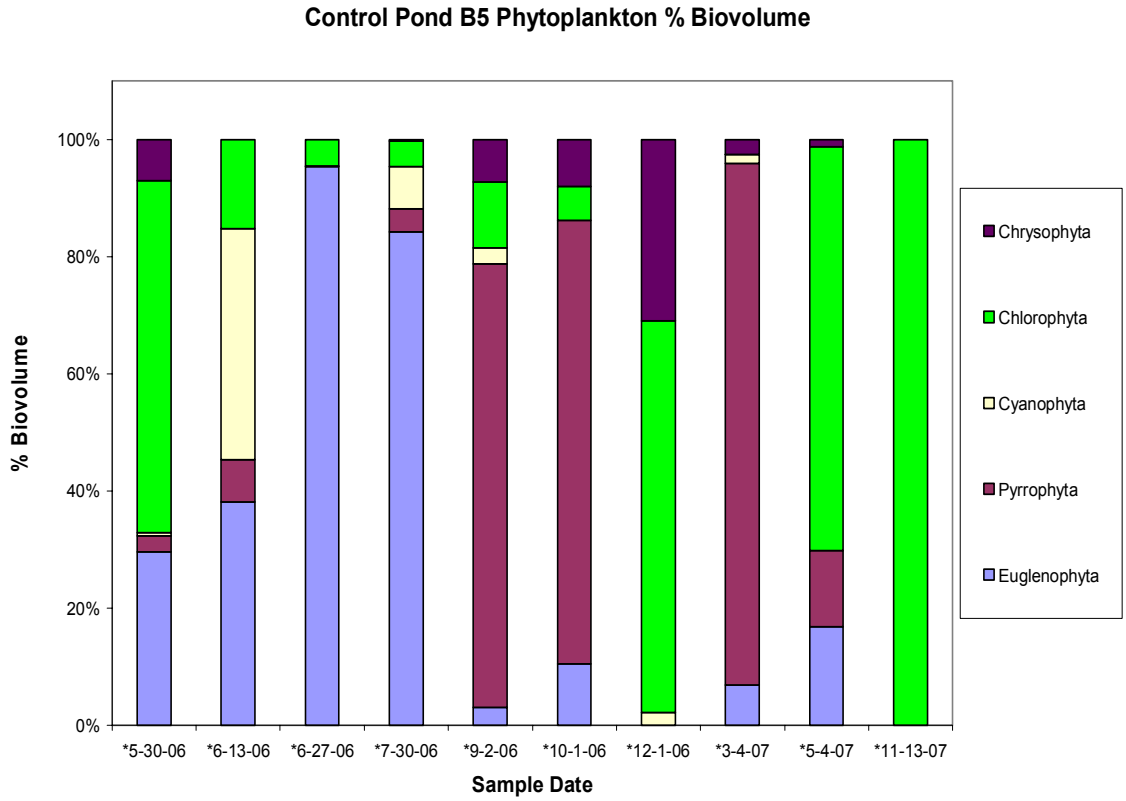

Figure B-27. Phytoplankton % biovolume for control pond B5

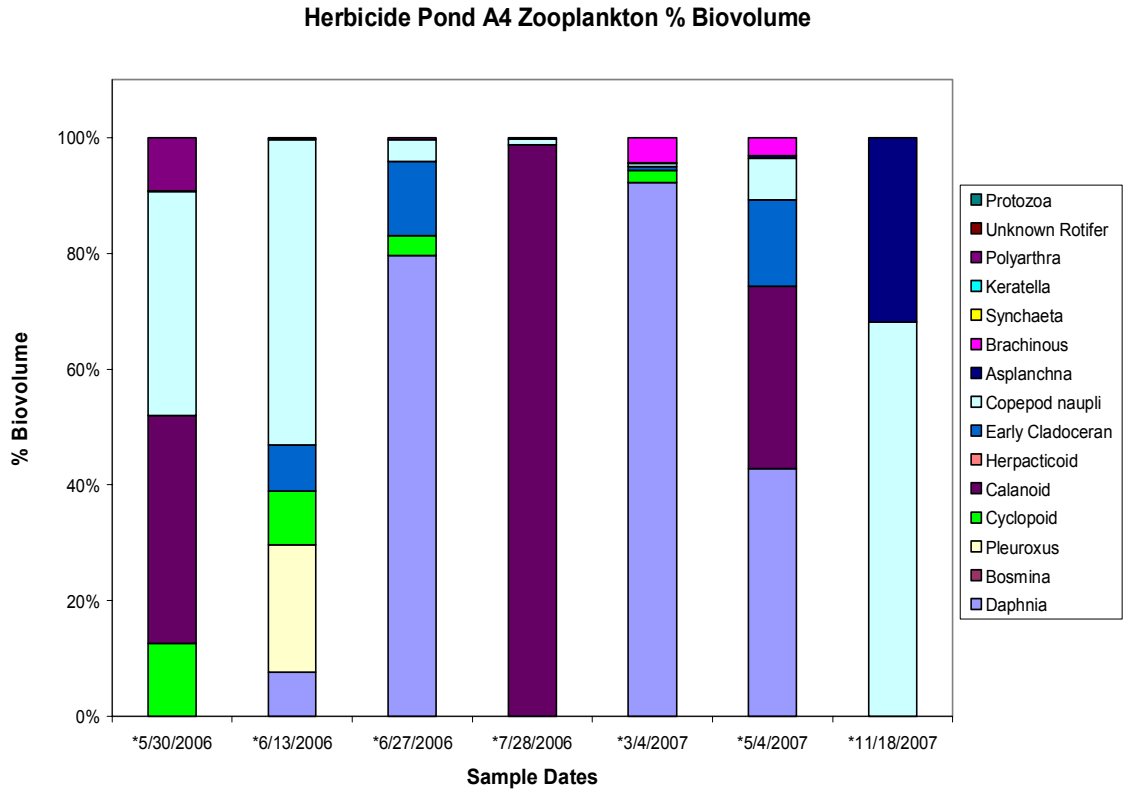

Figure B-28. Zooplankton % biovolume for herbicide pond A4

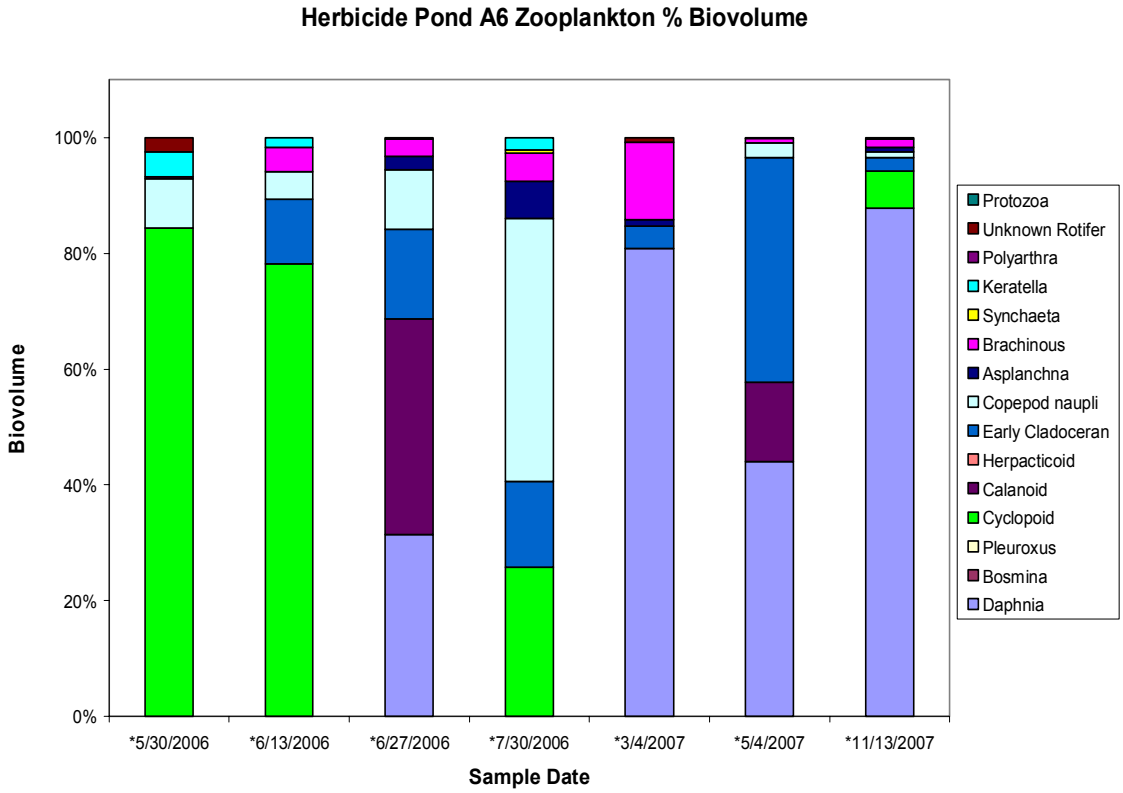

Figure B-29. Zooplankton % biovolume for herbicide pond A6

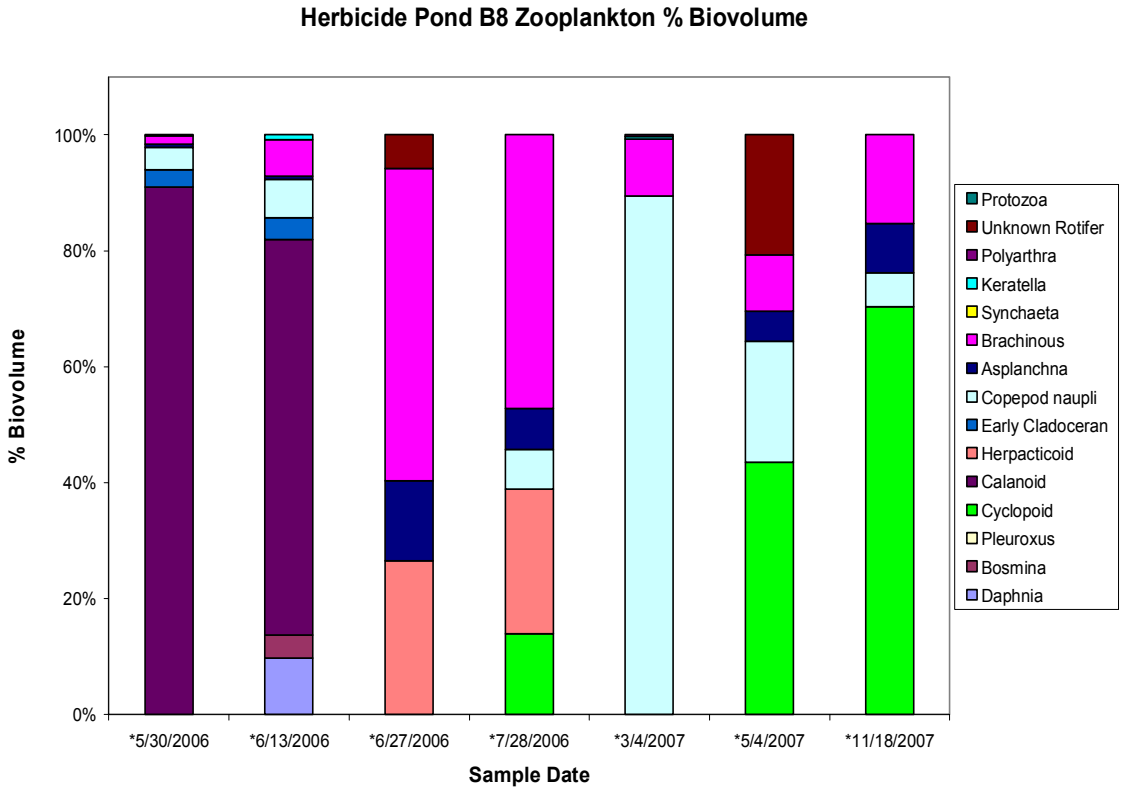

Figure B-30. Zooplankton % biovolume for herbicide pond B8

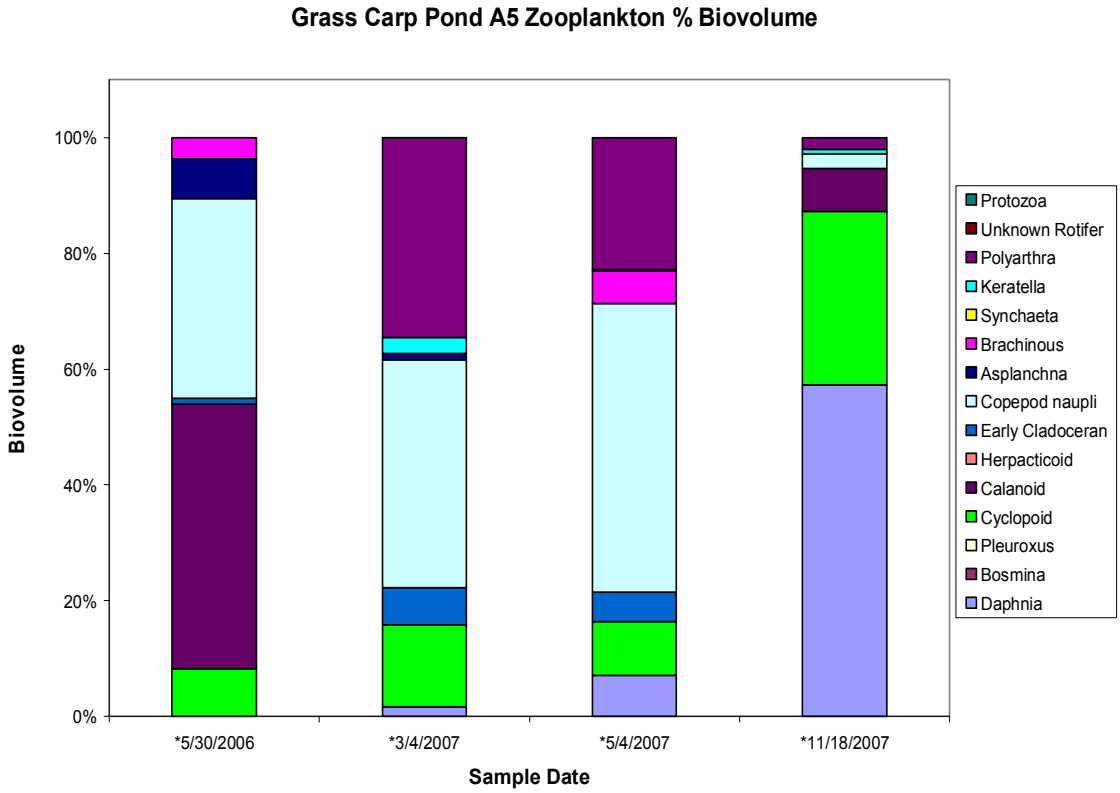

Figure B-31. Zooplankton % biovolume for grass carp pond A5

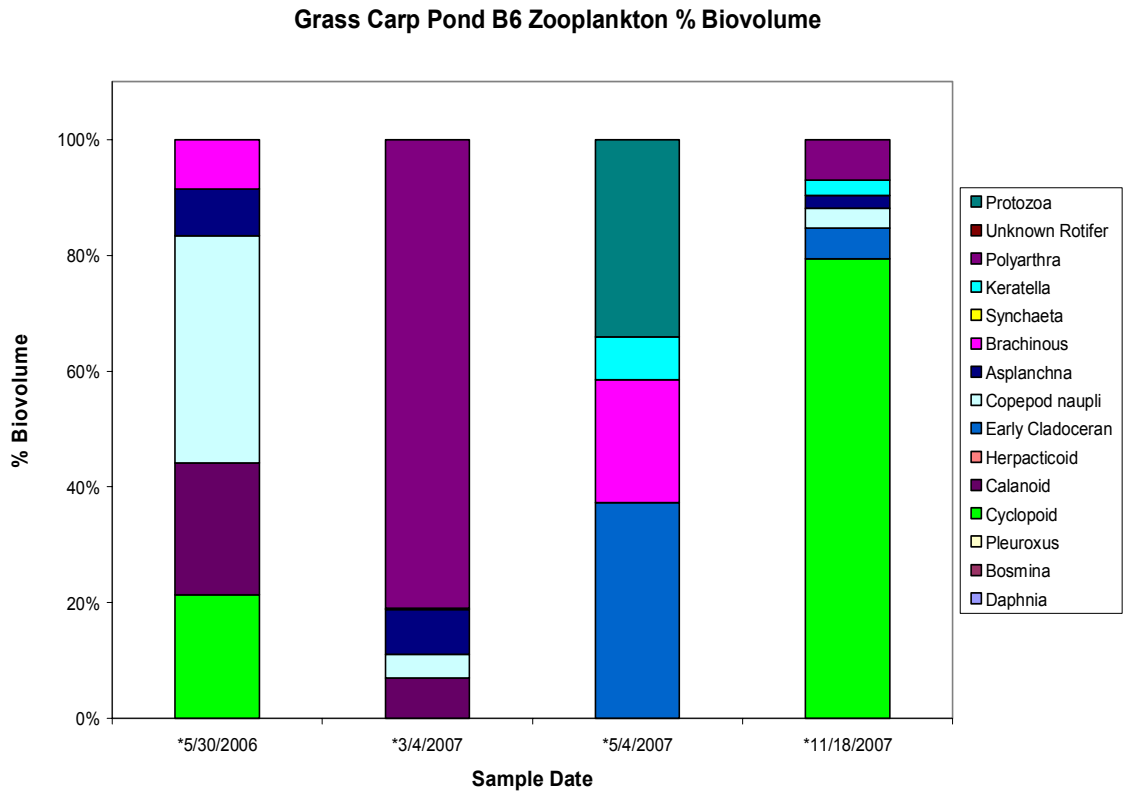

Figure B-32. Zooplankton % biovolume for grass carp pond B6

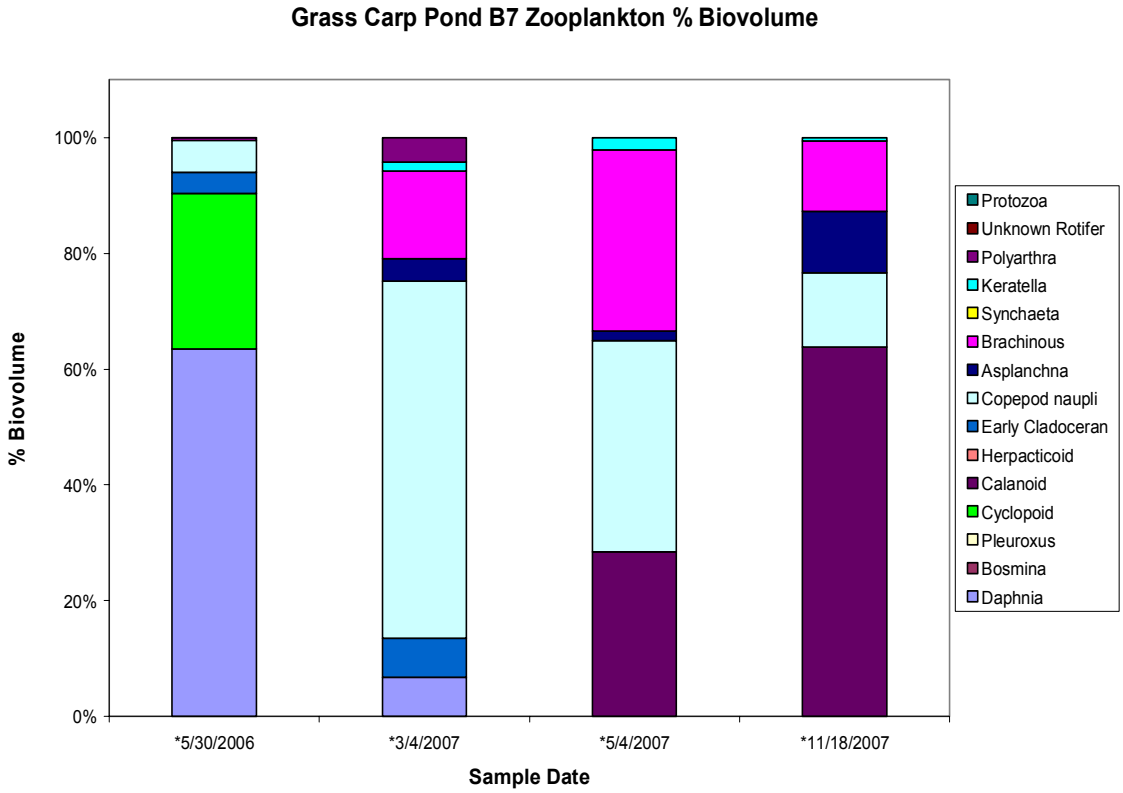

Figure B-33. Zooplankton % biovolume for grass carp pond B7

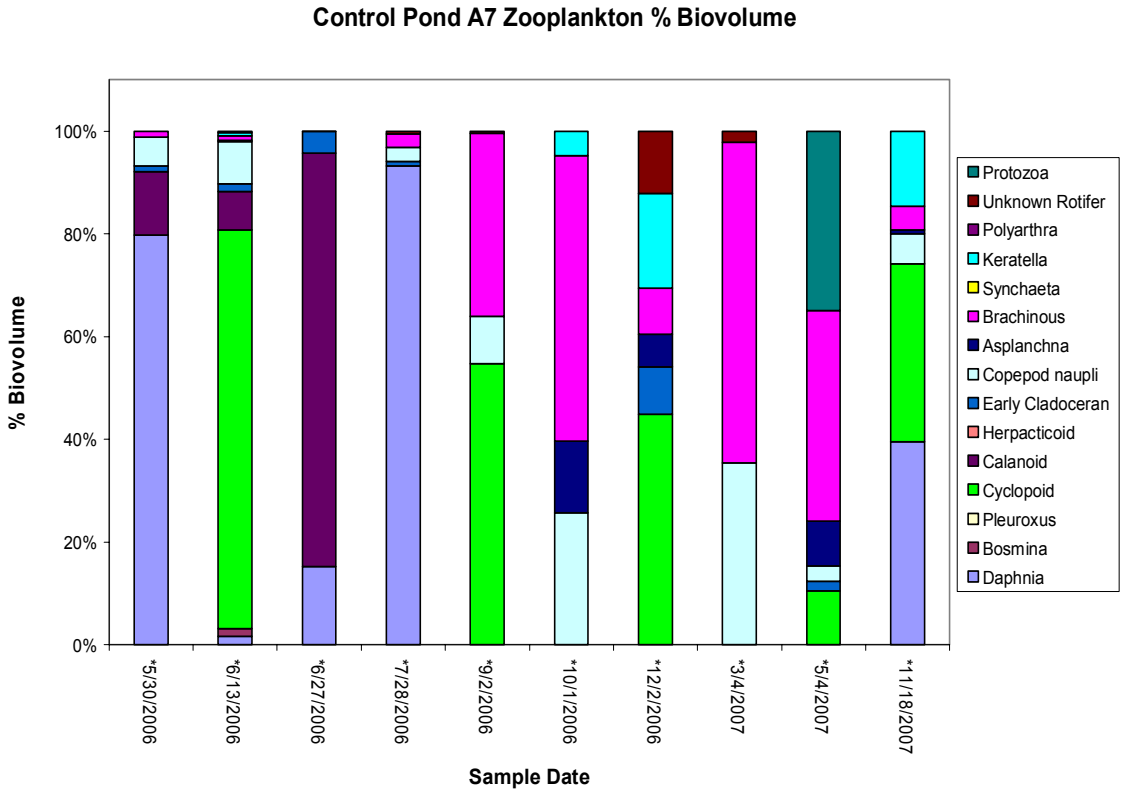

Figure B-34. Zooplankton % biovolume for control pond A7

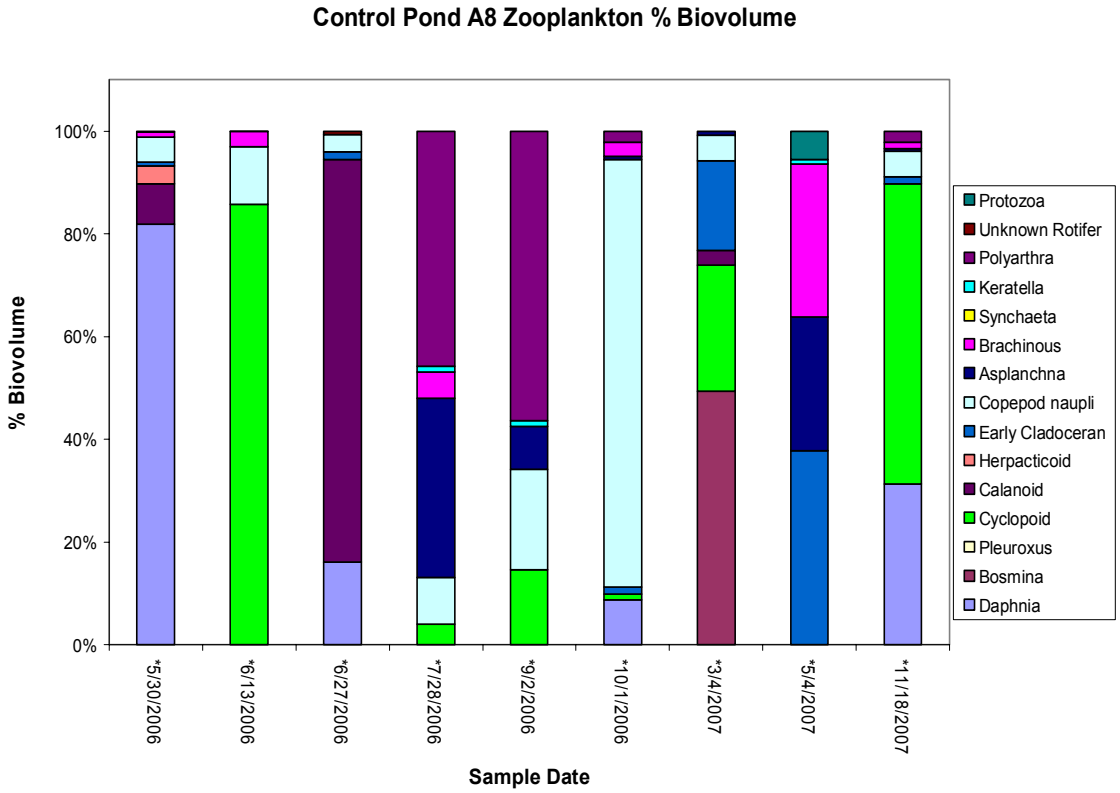

Figure B-35. Zooplankton % biovolume for control pond A8

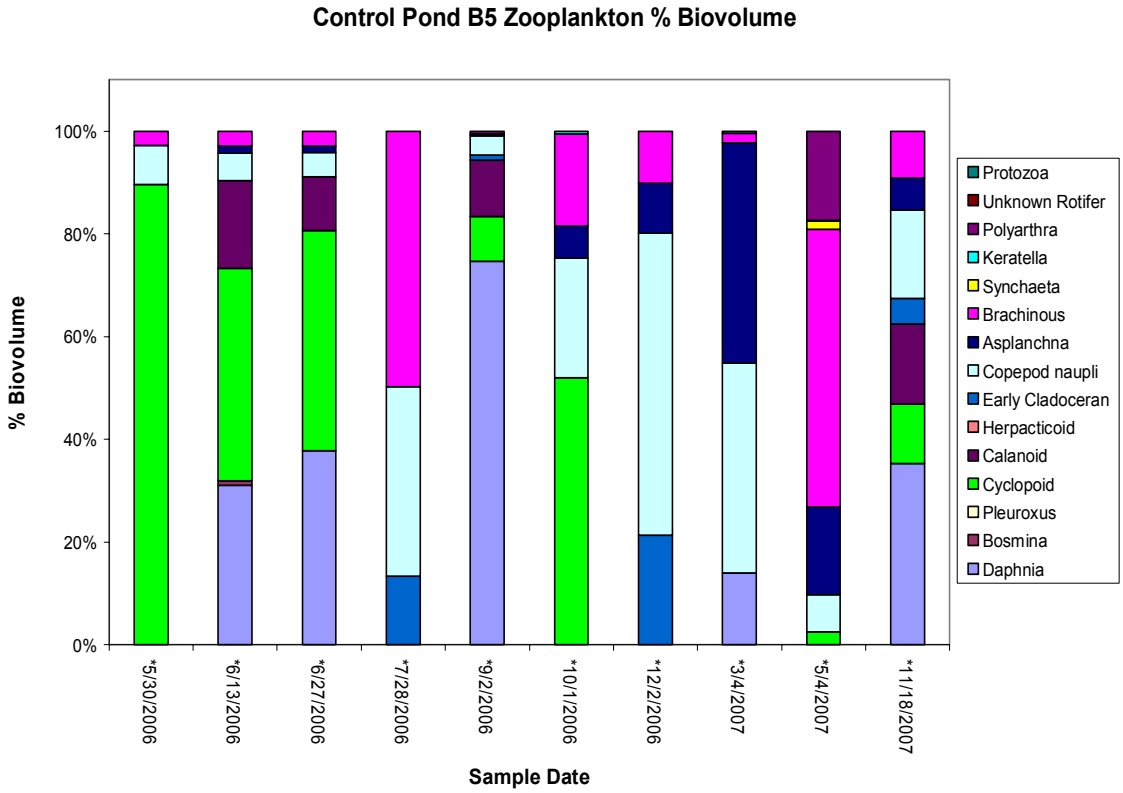

Figure B-36. Zooplankton % biovolume for control pond B5

VITA

Name:	Trevor Jason Knight
Address:	Department of Wildlife & Fisheries Sciences, 312 Nagle Hall, 2258 TAMU, College Station, TX 77843-2258
Email Address:	aggiebassin@hotmail.com
Education:	B.S., Fisheries Management, Delaware State University, 2005 M.S., Wildlife & Fisheries Sciences, Texas A&M University, 2009