ABSTRACT

GARNER, ALAN BRAD. High-Density Grass Carp Stocking Effects on a Reservoir Invasive Plant, Water Quality, and Native Fishes. (Under the direction of Dr. Thomas J. Kwak).

Stocking grass carp *Ctenopharyngodon idella* is a commonly applied technique used to control nuisance aquatic vegetation in reservoirs. Factors that influence the degree of aquatic vegetation control are stocking density, regional climate, abundance and species composition of the aquatic plant community, and relative grass carp feeding preferences for the plant species. We evaluated high-density grass carp stocking in a reservoir for control of parrot-feather (*Myriophyllum aquaticum*, an invasive aquatic plant that is not preferentially consumed by grass carp) and the associated effects on water quality and native fishes. Lookout Shoals Lake, a piedmont North Carolina reservoir, was stocked with triploid grass carp at a density of 100 fish per vegetated hectare. Parrot-feather biomass in the lake was significantly reduced three months after grass carp stocking, compared to biomass in in-situ exclosures. During the second year after grass carp stocking, parrot-feather biomass in the lake compared to biomass in in-situ exclosures indicated continued control, but unexplained lack of growth within most experimental exclosures precluded biomass analyses. Increases in ambient water chlorophyll *a*, reactive phosphorus, and nitrate-nitrite concentrations were measured after grass carp stocking. We evaluated the native fish community using seasonal shoreline electrofishing before and after grass carp stocking. Total catch (by number or biomass) for all fish species in aggregate at shoreline transects was not significantly different after grass carp stocking. Catch rates of largemouth bass *Micropterus salmoides*, bluegill *Lepomis macrochirus*, and redbreast sunfish *Lepomis auritus* were not significantly different after grass carp stocking, but yellow perch *Perca flavescens* catch rates were significantly lower. The biological significance of fish distribution changes and long-term effects on lake biota remain undetermined. Our results demonstrate that intensive grass carp stocking can control an invasive aquatic plant that is not preferentially consumed by grass carp, and reveal associated changes in water quality and fish distributions.

High-Density Grass Carp Stocking Effects on a Reservoir Invasive Plant,
Water Quality, and Native Fishes

by
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Biography

I was raised, along with one younger brother, by two wonderful parents who imparted compassion, discipline, humility, and a love of nature. My father is a wildlife biologist who enjoys hunting and fishing, and he introduced my brother and me to these activities at an early age. I started fishing with a cane pole for “bream” which I later learned were likely bluegill or redear sunfish. It wasn’t long before I graduated to bass fishing, of which I plan to be a life-long participant and student, especially smallmouth bass fishing. The natural world has always been an important part of my life. Activities like hunting, fishing, or just plain rambling have instilled in me an appreciation and respect for the natural world.

The outdoor activities of my youth heavily influenced my career plans. I entered the Aquaculture, Fisheries, and Wildlife Biology undergraduate program at Clemson University and studied the science of the natural world. I have since worked as a fisheries technician and consultant. I am currently pursuing a Master of Science degree in Fisheries, and Wildlife Sciences at North Carolina State University, of which this thesis is part of the requirements. Future plans include securing a position with a state or federal agency near the Rocky Mountains.

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Chapter I - High-Density Grass Carp Stocking Effects on a Reservoir Invasive Plant and Water Quality

Introduction

Stocking grass carp *Ctenopharyngodon idella* is an increasingly employed control technique for the management of nuisance aquatic vegetation and offers several advantages over mechanical or chemical controls in aquatic plant management, including low cost, labor saving, and low risk to public health (Sutton 1977; Rottman 1977; Bailey 1978; Allen and Wattendorf 1987; Cassani 1995; Li and Moyle 1999). Despite these advantages, aquatic vegetation control has not been universally achieved in all grass carp applications. Factors that influence the degree of vegetation control when using grass carp are stocking density (Fowler and Robson 1978), regional climate (Van Dyke et al. 1984; Bonar et al. 1993), abundance and species composition of the aquatic plant community (Prowse 1971; Opuszynski 1972; Van Dyke et al. 1984), and relative grass carp feeding preferences for the plant species (Fowler and Robson 1978). Application of this technique typically involves varying grass carp stocking densities in an effort to achieve the desired amount of vegetation control. High-density grass carp stocking to control persistent nuisance aquatic vegetation is receiving increased attention as an application, but scientific evaluation has been limited. Of particular interest are the degree of vegetation control and the potential impacts to ecosystem processes associated with this control technique.

Aquatic plants perform important ecological functions, including provision of food and cover for fish and wildlife, as well as aesthetic appeal (Cross 1969; Florschutz 1972; Vinogradov and Zolotova 1974; Strange et al. 1975; Voigts 1976; Hall and Werner 1977; Gasaway and Drda 1977; Savino and Stein 1982; Schramm and Jirka 1989). Aquatic plant species provide an array of benefits in a balanced ecosystem where their growth is primarily controlled by natural processes, such as regional climate, herbivory, and interspecific competition (Holm et al. 1969; Frye 1972; Aulbach-Smith and de Kozlowski 1996). Conversely, invasive plant species often spread unhindered by these same control mechanisms and become a nuisance (Frye 1972; Aulbach-Smith and de Kozlowski 1996). When alien plants naturalize or native plants are released from natural controls, the ecosystem is always impacted, and in some cases irrevocably (Cronk 1995).

Reservoirs are especially susceptible to nuisance aquatic plant infestations. Stabilized water levels characteristic of reservoir operations are conducive to aquatic plant growth (Leslie et al. 1987), but reservoirs are typically constructed in regions where natural lakes and native aquatic vegetation do not occur (Smart et al. 1998). Reservoir ecosystems are typically in relatively early stages of ecological succession, and establishment of native aquatic vegetation is a slow process, unlike that for nuisance species which invade rapidly and fill the available niche (Boyd 1971; Doyle et al. 1997; Smart et al. 1998).

Infestations of nuisance aquatic vegetation can impact the reservoir water resource in a number of ways. They can suppress other aquatic plants (Mitchell 1969; Aiken et al. 1979; Madsen et al. 1991; de Winton and Clayton 1996; Boylen et al. 1999; Ali and Soltan 2006), clog municipal and agricultural water sources and power generation facilities (Holm et al. 1969; Vinogradov and Zolotova 1974); degrade water quality (Roach and Wickliff 1934; Wickliff and Roach 1937, Boyd and Tucker 1998), hinder fishing and boating activities, and provide habitat for nuisance insect species (Aliyev and Bessmertnaya 1968; Orr and Resh 1992). Management of nuisance aquatic plants is a common problem for reservoir managers.

Stocking grass carp is an important biological management technique for nuisance aquatic plant growth. Grass carp are herbivorous fish native to China and Siberia and can grow to more than 50 kg and survive more than 20 years (Lin 1935; Nikol'skii 1954; Avault 1965; Cross 1969; Bailey and Boyd 1972; Jenkins and Burkhead 1993). Diploid grass carp were introduced to the U.S. for aquatic macrophyte control in 1963 and have been prohibited in many states because they may reproduce and impact native aquatic plant and fish communities (Stevenson 1965; Sills 1970; Sutton 1977; Pflieger 1978; Allen and Wattendorf 1987; Cassani 1995). Since the development of sterile triploid grass carp in 1983, which alleviated some concern over the long-term impacts from establishment of viable grass carp populations, grass carp have become widely used as a control agent for aquatic nuisance vegetation (Bain 1993; Cassani 1996).

While the ability of grass carp to control many aquatic plants has been demonstrated, they exhibit selective herbivory in aquatic systems with diverse macrophyte community structures. Field and laboratory studies indicate a grass carp feeding preference for succulent plant species and an aversion to fibrous plant species (Avault 1965; Prowse 1971; Chapman and Coffey 1971; Vinogradov and Zolotova 1974; Fowler and Robson 1978; Van Dyke et al. 1984; Wiley et al. 1986; Pine and Anderson 1991; Catarino et al. 1997). Grass carp feeding preference for a particular plant species may also vary by region, as soil and water quality characteristics affect plant texture or taste (Chapman and Coffey 1971; Leslie et al. 1987). Region-specific research of feeding preferences and stocking densities is necessary for effective aquatic plant management using grass carp (Leslie et al. 1987).

Feeding preference for the target plant relative to other aquatic vegetation by grass carp is an important consideration when determining stocking densities for aquatic plant control (Fowler and Robson 1978). Grass carp may not consume a less preferred target plant, opting instead for a preferred plant which can eventually lead to the spread of the target vegetation (Edwards 1974; Vinogradov and Zolotova 1974; Fowler and Robson 1978; Leslie et al. 1987). This is of particular concern if the vegetation preferred by grass carp is native or endemic to the aquatic system (Allen and Wattendorf 1987). Grass carp introduction in this scenario could adversely affect the species composition of the plant community (Vinogradov and Zolotova 1974; Fowler and Robson 1978).

High-density grass carp stocking is a potential management technique for controlling target plant species that are food sources not preferred by grass carp (Fowler and Robson 1978; Pine and Anderson 1991; Catarino et al. 1997). However, achieving control of aquatic vegetation not preferentially consumed by grass carp with intensive stocking could result in elimination of all aquatic vegetation in the system (Shireman et al. 1985; Kohler and Courtenay 1986; Allen and Wattendorf 1987; Bain 1993; Cassani 1995). Elimination of invasive aquatic vegetation would clearly limit its spread (Cronk 1995), but the adverse impact to native and endemic plant species is of concern (Allen and Wattendorf 1987). The potential elimination of all aquatic plant species should be considered in relation to impacting the uses of the waterbody (Leslie et al. 1987).

High-density grass carp stocking and the associated effects to the aquatic plant community may impact water quality. Excessive turbidity and accelerated eutrophication are two issues of concern. Shoreline and littoral macrophytes can reduce nutrient inflow

(Mickle and Wetzel 1978) and stabilize sediments of reservoirs (Dieter 1990; James and Barko 1994; Madsen et al. 2001; Bachmann et al. 2004). Increased turbidity could result from runoff, shoreline erosion, and destabilization of sediments due to a decline in shoreline and littoral vegetation. These same factors and the release of nutrients from grass carp excreta can also accelerate eutrophication (Stanley 1974; Mitzner 1978; Cassani 1996).

Field investigations in reservoirs where grass carp have controlled nuisance aquatic vegetation have detected some moderate changes in water quality. Nutrient concentrations have decreased, and water clarity has increased, following aquatic plant infestations (Canfield et al. 1983). Aquatic vegetation control utilizing grass carp has resulted in increases in nutrient parameters and decreased water clarity (Canfield et al. 1983; Leslie et al. 1983). Conversley, decreases in nutrient concentrations and increased water clarity have followed aquatic vegetation control utilizing grass carp (Mitzner 1978). Water depth, retention time, nutrient loading, relative abundance of vegetation to lake volume, and rate of vegetation removal influence the impact grass carp may have on lake and reservoir water quality (Mitzner 1978; Leslie et al. 1983; Shireman et al. 1985; Cassani 1996). The mechanisms producing these conflicting results remain unclear and demonstrate that additional research is needed in this area.

Controlling aquatic plant species that are food sources non-preferred by grass carp with stocking densities of grass carp that are higher than normally recommended is an unproven practice, and the effects of such a practice on reservoir aquatic plant communities and water quality remain largely unknown. Greater insight into the efficacy and the indirect effects of this practice is needed for the advancement of sound aquatic plant management. With this research project, I attempt to address two key issues related to high-density grass carp stocking in a reservoir setting: (1) the degree of control of aquatic plant species considered difficult to control with grass carp and (2) possible impacts to water quality. The objectives of this study were to evaluate the efficacy of a high-density triploid grass carp stocking to control nuisance levels of parrot-feather *Myriophyllum aquaticum*, a non-native nuisance aquatic plant that is not a food source preferred by grass carp and detect associated changes in the water quality of a piedmont North Carolina reservoir.

Study Area and Aquatic Plant Management

Lookout Shoals Lake is a 528-ha reservoir located between Lake Hickory and Lake Norman on the Catawba River in the western piedmont of North Carolina (Figure 1). The lake is jointly managed by Duke Energy and the North Carolina Wildlife Resources Commission. It was impounded in 1915 by the construction of the Lookout Shoals Dam and Hydroelectric Station for power production, which is the primary use of the reservoir; municipal water source and recreation are secondary uses (Duke Power 1999; NCDWQ 2003). Parrot-feather and Brazilian elodea *Egeria densa* are two invasive aquatic plant species that dominate the submersed aquatic plant community of Lookout Shoals Lake (Figure 2). Both are introduced aquatic plants from South America that often form large, extensive mats on the water surface that interfere with ecological processes and human uses of the reservoir (Weatherby 1932; Sutton 1985; Blackburn et al. 1969; Aulbach-Smith and de Kozlowski 1996).

Water-level management and aquatic herbicides have been utilized to control parrot-feather and Brazilian elodea abundances and distributions. Brazilian elodea has been successfully controlled with winter drawdowns of Lookout Shoals Lake, which have restricted the infestation to the riverine section from the Oxford Hydroelectric Dam, that impounds Lake Hickory, downstream about five km to the upstream impounded area of Lookout Shoals Lake. Parrot-feather has infested approximately 90 ha of the littoral zone of the upper reservoir due to the ineffectiveness of winter drawdowns at controlling its spread and has been selectively controlled with aquatic herbicides (Ken Manuel, Duke Energy, personal communication). However, recent installation of a municipal water facility has resulted in increased restrictions on the use of aquatic herbicides, and other control options were sought for the management of this invasive aquatic plant. Parrot-feather is not considered to be a food source preferred by grass carp (Avault 1965; Pine and Anderson 1991; Catarino et al. 1997); thus, control of this nuisance aquatic plant with grass carp was questionable. This presented an experimental opportunity to evaluate a high-density grass carp stocking management technique in an attempt to control a nuisance aquatic plant species that was not a food source preferred by grass carp.

In May 2005, Lookout Shoals Lake was stocked with approximately 9,200 grass carp (100 fish per vegetated ha). This stocking rate is twice the recommended rate for North Carolian public waters (Rice et al. 1999), and characterized as high relative to those applied in previous studies (Bailey 1978; Fowler and Robson 1978; Leslie et al. 1983; de

Kozlowski 1994; Hanlon et al. 2000). These fish were approximately 350 mm in total length and 525 g in weight.

Methods

Experimental Exclosure and Vegetation Sampling

To determine grass carp efficacy as a control agent of parrot-feather, eight square exclosures approximately 6-m wide and 2-m high were installed prior to grass carp stocking to allow vegetation growth within the exclosures without grass carp influence (Figure 3). They were constructed in shallow water, less than 2 m deep, in areas of known parrot-feather occurrence. Galvanized pipe was driven into the substrate at the exclosure corners, and plastic 1.3-cm square mesh fencing was stretched around and held in place by vinyl-coated galvanized cable at the top and reinforcement bar weights at the bottom. An area of equal size and morphology adjacent to each exclosure was delineated to sample for comparison of parrot-feather biomass within the corresponding exclosure.

Harvested plant biomass sampling was conducted for sensitivity in detecting changes in parrot-feather abundance to evaluate the degree of control (Madsen and Bloomfield 1993). Throughout the growing season, vegetation was harvested from six rectangular quadrats within each exclosure and the area adjacent to the exclosure. The quadrats were 1 m wide by 6 m long and extended the length of the exclosure perpendicular to the reservoir shoreline. One quadrat from inside each exclosure and one quadrat from outside each exclosure were sampled monthly from July through September of 2005 and 2006 after grass carp stocking. Rooted vegetation was completely harvested by hand and by the same observer. The vegetation from each quadrat was then spun in a mesh bag to remove excess water, and fresh weight was measured (±1 g). This process was repeated until the weight stabilized. The first vegetation samples (July 2005) from inside the exclosures were used to verify an adequate sample size using the equation suggested by Madsen (1993) for estimating adequate sample sizes needed for aquatic vegetation biomass sampling. Evidence of non-normality was detected in the parrot-feather biomass samples which was not remedied by data transformation. Thus, a Wilcoxon signed rank test was used to detect differences in parrot-feather biomass from inner and outer quadrats. All statistical comparisons were considered significant at a probability of less than 0.05 (alpha).

Water Quality Sampling

Lookout Shoals water quality was monitored monthly from July 2005 through April 2007. Surface samples and measurements were collected from two mainlake locations in the middle and upper lake areas and from inside and outside each of the eight experimental exclosures (Figure 2). Measurements of temperature, dissolved oxygen, and pH were made using a Hydrolab® model MS5 multi-probe datasonde and Surveyor 4a display unit. Surface samples for ammonia, nitrate, nitrite, and reactive phosphorus were filtered immediately under low pressure through a 0.45-µ glass fiber filter to remove suspended solids. Surface samples were placed on ice immediately following collection or collection and filtration and maintained at less than 4^{o} C until analyzed. Samples were analyzed within 24 h of collection for total alkalinity (phenolphthalein and total method) and total hardness (ethylenediaminetetraacetic acid titration), and within 48 h of collection for ammonia (salicylate method), nitrate (calcium reduction method), nitrite (diazotization method), reactive phosphorus (PhosVer 3 method), using a Hach® CEL/850 test kit. Chlorophyll *a* samples were delivered to the Center for Applied Aquatic Ecology, North Carolina State University, within 24 h of collection, where they were analyzed using the EPA 445.0 (revision 1.2) method (EPA 1997).

Lookout Shoals Lake water quality was monitored during the months of June, July, and August of 1997 and 2002 by the North Carolina Division of Water Quality (NCDWQ 2003). One of these locations corresponds to one (midlake) of the two mainlake locations monitored for our research. This location provides a comparison of water quality parameters before (1997, 2002) and after (2005-2007) grass carp stocking. One-way analysis of variance (JMP 7, Statistical Analysis Software, Cary, North Carolina) was utilized to detect differences among years for water quality determinations available from the NCDWQ (NCDWQ 2003) and those conducted for this study, and among years and seasons for the mainlake water quality determinations from this study. A one-way analysis of variance was also used to detect differences among seasons and years between the inner and outer exclosure water quality determinations with blocking by exclosure.

Results

Vegetation

Only three submersed aquatic plant taxa were encountered in the sampling for this research; they were Chara (*Chara* spp.), Brazilian elodea, and parrot-feather (Table 1).

Chara and Brazilian elodea were only encountered in quadrats inside the exclosures, while parrot-feather was encountered inside and outside the exclosures. Parrot-feather biomass was highly variable among exclosures and sampling occasions (Table 2). Analysis of July 2005 parrot-feather biomass indicated no significant difference in the mean parrot-feather biomass from the quadrats available to the grass carp (0.02 kg/m^2) and the quadrats from inside the exclosures (0.05 kg/m^2) two months after grass carp were stocked and one month into the parrot-feather growing season (Figure 4). A significant reduction in parrot-feather biomass was observed outside the exclosures, relative to that inside, three and four months after grass carp were stocked. Mean parrot-feather biomass from the quadrats available to the grass carp was less than 0.01 kg/m^2 in both August and September, whereas mean parrot-feather biomass from the quadrats in the exclosures was 0.60 kg/m^2 and 0.54 kg/m^2 in August and September, respectively. These results demonstrate an average difference in parrot-feather biomass inside and outside of exclosures over 50-fold.

In the 2006 sampling season, no vegetation was encountered in any quadrat available to grass carp (Table 2). One exclosure (number 7) contained more parrot-feather biomass during all three months than in the corresponding months of 2005. Among all exclosures, no growth was observed in five of the eight quadrats in July, four of the eight quadrats in August, and seven of the eight quadrats in September. This absence of parrot-feather precluded statistical analyses of abundance in 2006. However, on average, parrot-feather biomass inside the exclosures was 0.06 kg/m^2 during July, 0.18 kg/m^2 during August, and 0.17 kg/m^2 during September, relative to no plant biomass outside of the exclosures.

Water quality

No significant differences were detected before and after grass carp stocking for temperature, pH, and secchi depth, or dissolved oxygen, ammonia, and nitrate-nitrite concentrations from existing data and the sampling conducted during this study (Table 3). There was a significant increase in chlorophyll *a* concentrations from 2002 and 2005 to 2006, but this change did not coincide temporally with grass carp stocking.

No consistent or ecologically significant changes between seasons in physical water quality parameters were detected during the two years following grass carp stocking (Table 4). A relatively high degree of variability in secchi depth readings was found for all seasons during the two years following grass carp stocking. Some interesting changes were observed in chemical parameters, with elevated concentrations of reactive phosphorus and nitrate-nitrite throughout the first year after the grass carp were stocked. Mean nitrate-nitrite concentrations were four times greater during the fall and winter of 2005 and 2006 and the spring of 2006, than those in the same seasons one year later; mean phosphorus concentrations were two to five times greater throughout the first year following grass carp stocking than those in the second year. No significant differences were detected in mean chlorophyll *a* concentrations; maximum detected concentrations were disparate, 9.9 mg/L during summer 2005 versus 5.4 mg/L during summer 2006, 17.5 mg/l during fall and winter of 2006 and 2007 versus 2.9 mg/L during fall and winter of 2005 and 2006, and 8.3 mg/L during spring 2006 versus 1.2 mg/L during spring 2007. Slightly elevated levels of total alkalinity and hardness were observed during the fall and winter of 2006 and 2007, relative to those of the previous year. Stable ammonia concentrations and typical seasonal variation in dissolved oxygen concentrations were observed over the entire course of the study.

Significant differences were detected between the uplake and midlake sampling locations for the water quality parameters (Table 5). Consistently elevated levels were detected in reactive phosphorus concentrations during summer 2005 and secchi depth readings during summer 2006 at the uplake location. Consistently elevated levels of total hardness were detected at the midlake location during fall and winter of 2005 and 2006. However, no significant differences were detected for temperature, dissolved oxygen, pH, chlorophyll *a*, ammonia, nitrate-nitrite, or total alkalinity between the locations throughout the course of the study.

No consistent or ecologically significant differences were detected inside versus outside of the experimental exclosures in any season for temperature or dissolved oxygen, ammonia, nitrate-nitrite, total alkalinity, and total hardness concentrations (Tables 6-8). Consistently elevated pH was observed inside the exclosures during summer 2005 (Table 6). Increased maximum concentrations were detected in reactive phosphorus during summer 2005 and in chlorophyll *a* during summer 2005 and 2006 inside the exclosures versus outside.

Discussion

My analysis of parrot-feather abundance indicated that control was achieved shortly after grass carp stocking. One factor that may have influenced the degree of control is the timing of the stocking. Grass carp were stocked in May, one month prior to the initiation of the parrot-feather growing season and two months prior to its peak growth period in Lookout Shoals Lake. In this case, the plant biomass of a major infestation did not require depletion to achieve reasonable control; rather, grass carp could consume less abundant vegetation as it grew. Grass carp presence at the initiation of the parrot-feather growing season may be a key factor contributing to the level of control seen. Also, parrot-feather new growth may be more palatable to grass carp facilitating early and continued control as the growing season progressed (Prowse 1971; Leslie et al. 1994). This factor (i.e., timing) may be of equal importance as density of grass carp for controlling parrot-feather.

Lakewide visual estimation and quadrat sampling indicated a lack of plant biomass and continued vegetation control in 2006 and 2007. The finding of little to no vegetation growth inside the exclosures during 2006 was unexpected and may be due to herbivore intrusion, sedimentation effects, or annual variation in plant growth. Doyle et al. (1997) reported herbivory and sedimentation as deterrents to the establishment of aquatic plants in exclosures in two Texas reservoirs. Evidence of sedimentation and intrusion by painted turtles *Chrysemys picta* was observed among all exclosures during my study. Painted turtles are known to be omnivorous (Martof et al. 1980) and may have sought the vegetation food source within the exclosures as the vegetation outside the exclosures was depleted. Further, other vertebrate or invertebrate organisms may have been attracted to the vegetation in the exclosures, as similar physical structure became scarce throughout the lake. No clear explanation for the interannual differences in results were revealed even after careful field observation.

Of the significant changes detected in the water quality parameters of Lookout Shoals Lake after grass carp stocking, only those for nutrient and chlorophyll *a* concentrations pose ecological significance. The elevation in Chorophyll *a* during the second year after grass carp were stocked could be indicative of a shift in nutrient availability to the phytoplankton community resulting from grass carp excreta or resuspension of sediments (Boyd 1971; Shireman et al. 1985). The elevated nitrogen and phosphorus levels observed beginning in the summer of 2005 and through the spring of 2006 were followed by a four-fold increase in chlorophyll *a* concentrations during the summer of 2006. Chlorophyll *a* serves as an index of primary production (Brylinsky and Mann 1971). In lakes with extensive littoral areas, aquatic macrophytes can comprise a significant portion of the primary production as they sequester nutrients and alter photic conditions early in the growing season in competition with phytoplankton communities (Boyd 1971). Aquatic macrophytes also have added competitive advantages over phytoplankton by accessing nutrients in the substrate as well as stabilizing those substrates preventing nutrient recycling to the water (Bachmann et al. 2004). However, attributing this increase in phytoplankton production to the loss of littoral vegetation is not clear. Lookout Shoals Lake water quality is influenced by watershed inputs and releases from upstream impoundments (NCDWQ 2003), and our ability to detect any grass carp influence on the water quality of Lookout Shoals is confounded by these factors and a low water retention time.

Management Implications

Invasive aquatic vegetation is an ongoing problem for water resource managers. The triploid grass carp is an increasingly useful biological control agent of aquatic vegetation that can decrease cost and risk to public health (Sutton 1977; Bailey 1978; Li and Moyle 1999), and my research demonstrates that high-density grass carp stocking can be included in that assessment for controlling aquatic plant species considered to be food sources non-preferred by grass carp. I observed control of an invasive aquatic plant previously considered difficult to control with grass carp, presumably due to the high-density component of the stocking. The changes observed in reservoir water quality following vegetation control were moderate and are likely of minimal concern for management and water use.

In addition to stocking density, timing of grass carp stocking may influence the degree of vegetation control. For control of invasive aquatic vegetation that is not preferred by grass carp, managers should consider stocking soon after infestation or early in the season before vegetation biomass reaches peak levels.

When deciding on a management strategy for the control of invasive aquatic plants, one must consider the impacts of control techniques versus the costs of no control. This study did not address possible impacts to native or endemic vegetation because the aquatic plant community of the study reservoir was dominated by invasive plant species.

The sparse native littoral aquatic vegetation community of Lookout Shoals Lake was not extensively monitored, but there is evidence of an impact to the macroalgae following grass carp stocking (Table 1). My results confirm research on grass carp effects in other water bodies that indicate full control of aquatic plants may result with high-density grass carp stocking (Bailey and Boyd 1972; Martyn et al. 1986; Hanlon et al. 2000), and managers should expect full control and depletion of aquatic vegetation with this strategy. The changes observed in the aquatic plant community and water quality of Lookout Shoals raise questions regarding the possible long-term effects on littoral and shoreline aquatic plant and fish species. Future research may address those issues, as well as broader ecosystem-level effects.

The results of this study and other studies addressing the effectiveness and ecological impacts of utilizing grass carp for aquatic plant control will aid managers in developing aquatic plant management plans. Relating the effectiveness and potential ecological impacts of various control techniques is crucial in this process.

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Table 1. Total submersed vegetation biomass (g/m^2) from inside and outside eight experimental exclosures on Lookout Shoals Lake during July, August, and September of 2005 and 2006, following the May 2005 stocking of grass carp.

	2005			2006		
Species	July	August	September	July	August	September
			Inside			
Chara spp.	223	142	45	140	0	0
Brazilian elodea	1	0	0	0	0	0
Parrot-feather	416	4,784	4,349	462	1,459	1,330
			Outside			
Parrot-feather	180	26	12	0	0	0

Table 2. Parrot-feather biomass (g/m^2) from inside (In) and outside (Out) eight experimental exclosures on Lookout Shoals Lake during July, August, and September of 2005 and 2006.

	2005						2006					
	July		August		September		July		August		September	
Exclosure	In	Out	In	Out	In	Out	In	Out	In	Out	In	Out
1	27	2	1,643	0	385	0	0	0	0	0	0	0
2	2	2	3	0	8	0	0	0	0	0	0	0
3	37	2	178	2	197	0	13	0	2	0	0	0
4	0	0	23	0	48	0	117	0	2	0	0	0
5	245	33	1,660	0	2,025	0	0	0	0	0	0	0
6	38	113	580	2	447	0	0	0	2	0	0	0
7	65	28	687	22	1,177	12	332	0	1,453	0	1,330	0
8	2	0	10	0	62	0	0	0	0	0	0	0

Table 3. Mean, standard error (in parentheses), and range of water quality measurements from Lookout Shoals Lake monthly (June-August) in 1997 and 2002 (NCDWQ 2003) prior to grass carp stocking, and in 2005-2006 after grass carp stocking. Within each parameter, different superscript letters indicate significant differences.

	Year			
	1997	2002	2005	2006
Temperature (^oC)	25.3^a (3.2)	26.0^a (1.9)	24.7^a (<0.1)	27.8^a (1.2)
	19.7-30.9	22.3-28.1	24.7-24.8	26.0-30.0
pH	6.5^a (0.3)	6.9^a (0.2)	6.7^a (0.2)	7.4^a (0.3)
	6.1-7.2	6.6-7.2	6.5-6.8	7.1-7.9
Dissolved oxygen (mg/L)	7.0^a (0.8)	6.3^a (0.9)	5.5^a (1.0)	7.0^a (0.5)
	5.6-8.3	4.6-7.3	4.5-6.5	6.1-7.6
Secchi depth (m)	1.5^a (0.3)	1.5^a (0.5)	1.2^a (0.6)	1.4^a (0.1)
	1.0-2.0	1.0-2.4	0.6-1.8	1.3-1.5
Chlorophyll *a* (µg/L)	-	3.5^a (1.5)	2.1^a (0.4)	8.7^b (0.6)
	-	0.4-5.0	1.7-2.5	8.0-9.9
Ammonia (mg/L NH_3-N)	0.05^a (0.02)	0.03^a (0.01)	0.07^a (0.02)	0.02^a (0.01)
	0.01-0.07	0.02-0.04	0.05-0.08	<0.01-0.03
Nitrate-nitrite (mg/L NO_3^--N, NO_2^--N)	0.25^a (0.03)	0.07^a (0.04)	0.31^a (0.20)	0.27^a (0.15)
	0.22-0.30	0.02-0.15	0.11-0.52	0.06-0.56

Table 4. Mean, standard error (in parentheses), and range of water quality measurements during spring (April-June), summer (July-September), fall-winter (October, November, January, March) taken at uplake and midlake locations on Lookout Shoals Lake 2005-2007. Within each parameter and season, different superscript letters indicate significant differences.

	Spring		Summer		Fall-Winter	
	2006 (*N*=6)	2007 (*N*=2)	2005 (*N*=6)	2006 (*N*=6)	2005-2006 (*N*=8)	2006-2007 (*N*=8)
Temperature (^{o}C)	19.1^{a} (2.0)	15.6^{a} (0.8)	24.7^{a} (0.1)	26.4^{a} (1.0)	12.1^{a} (1.7)	12.0^{a} (1.7)
	14.5-26.0	14.8-16.3	24.1-25.1	23.5-30.0	6.8-18.8	8.0-19.5
Dissolved oxygen (mg/L)	8.0^{a} (0.3)	9.8^{b} (0.3)	5.4^{a} (0.3)	5.6^{a} (0.5)	8.4^{a} (0.5)	9.7^{a} (0.7)
	6.8-8.9	9.5-10.0	4.4-6.5	4.4-7.4	6.4-10.3	7.4-13.0
pH	7.0^{a} (<0.1)	7.5^{b} (0.1)	6.6^{a} (<0.1)	7.2^{b} (0.2)	7.2^{a} (<0.1)	7.5^{b} (0.1)
	6.8-7.1	7.4-7.5	6.5-6.8	6.7-7.9	7.0-7.3	7.2-8.1
Secchi depth (m)	1.9^{a} (0.2)	2.7^{b} (0.2)	1.5^{a} (0.3)	1.6^{a} (0.1)	2.3^{a} (0.2)	1.9^{a} (0.4)
	1.5-2.5	2.5-2.8	0.6-2.5	1.3-2.0	1.4-3.0	0.9-3.3
Chlorophyll *a* (µg/L)	3.9^{a} (1.2)	1.1^{a} (0.2)	2.5^{a} (0.6)	4.3^{a} (1.5)	1.7^{a} (0.3)	4.4^{a} (2.3)
	1.2-8.3	0.9-1.2	1.1-5.4	1.5-9.9	0.7-2.9	0.7-17.5
Ammonia (mg/L NH_3-N)	0.05^{a} (0.01)	0.05^{a} (0.01)	0.06^{a} (0.01)	0.04^{a} (0.02)	0.06^{a} (0.01)	0.05^{a} (0.01)
	0.01-0.1	0.04-0.05	<0.01-0.1	<0.01-0.11	<0.01-0.1	<0.01-0.09
Nitrate-nitrite (mg/L NO_3^--N, NO_2^--N)	0.63^{a} (0.02)	0.15^{b} (0.01)	0.47^{a} (0.12)	0.54^{a} (0.18)	0.72^{a} (0.15)	0.16^{b} (0.02)
	0.56-.68	0.14-0.16	0.11-0.86	0.06-1.11	0.09-1.31	0.05-0.25
Reactive phosphorus (mg/L PO_4^{3-})	0.17^{a} (0.02)	0.03^{b} (0.02)	0.24^{a} (0.03)	0.09^{b} (0.02)	0.23^{a} (0.01)	0.11^{b} (0.01)
	0.11-.26	0.01-0.05	0.12-0.32	0.06-0.17	0.18-0.30	0.06-0.16
Total alkalinity (mg/L as $CaCO_3$)	11.3^{a} (0.50)	12.3^{a} (0.3)	11.5^{a} (0.3)	10.7^{a} (0.6)	11.6^{a} (0.2)	13.7^{b} (0.3)
	9.5-12.5	12.0-12.5	10.5-11.9	8.5-12.5	11.0-12.5	12.5-14.5
Total hardness (mg/L as $CaCO_3$)	10.5^{a} (0.6)	12.3^{a} (0.3)	11.1^{a} (0.3)	10.9^{a} (0.5)	11.7^{a} (0.3)	13.6^{b} (0.6)
	8.5-11.5	12.0-12.5	10.4-11.7	9.0-12.5	10.6-12.8	11.5-16.5

Table 5. Mean, standard error (in parentheses), and range of water quality measurements during spring (April-June), summer (July-September), fall-winter (October, November, January, March) taken at uplake and midlake locations on Lookout Shoals Lake 2005-2007. Within each parameter, season, and year, different superscript letters indicate significant differences.

	Summer 2005		Fall-Winter 2005-2006		Spring 2006		Summer 2006		Fall-Winter 2006-2007	
	Uplake	Midlake	Uplake	Midlake	Uplake	Midlake	Uplake	Midlake	Uplake	Midlake
Temperature (^oC)	24.5^a (0.3)	24.8^a (0.1)	12.2^a (2.7)	12.0^a (2.4)	19.0^a (2.9)	19.2^a (3.5)	25.9^a (1.2)	27.0^a (1.9)	12.2^a (2.6)	11.8^a (2.5)
	24.1-25.1	24.7-25.0	6.8-18.8	7.3-17.8	15.5-24.7	14.5-26.0	23.8-28.0	23.5-30.0	8.3-19.5	8.0-18.4
Dissolved oxygen (mg/L)	5.2^a (0.5)	5.6^a (0.6)	8.3^a (0.8)	8.5^a (0.5)	7.7^a (0.6)	8.3^a (0.3)	5.1^a (0.5)	6.1^a (0.8)	9.8^a (1.2)	9.7^a (1.0)
	4.4-5.9	4.5-6.5	6.4-10.3	7.3-9.9	6.8-8.9	7.6-8.7	4.4-6.1	4.7-7.4	7.6-13.0	7.4-12.2
pH	6.5^a (<0.1)	6.6^a (0.1)	7.2^a (0.1)	7.2^a (0.1)	7.0^a (0.1)	7.0^a (<0.1)	7.0^a (0.2)	7.4^a (0.2)	7.6^a (0.2)	7..5^a (0.2)
	6.5-6.6	6.5-6.8	7.0-7.3	7.0-7.3	6.8-7.1	7.0-7.1	6.7-7.2	7.2-7.9	7.3-8.1	7.2-8.0
Secchi depth (m)	1.7^a (0.6)	1.3^a (0.4)	2.4^a (0.3)	2.2^a (0.3)	2.0^a (0.3)	1.8^a (0.2)	1.9^a (0.1)	1.4^b (<0.1)	1.7^a (0.4)	2.1^a (0.6)
	0.6-2.5	0.6-1.8	1.5-3.0	1.4-2.7	1.5-2.5	1.5-2.1	1.7-2.0	1.3-1.4	0.9-2.5	0.9-3.3
Chlorophyll *a* (µg/L)	1.8^a (0.4)	3.2^a (1.1)	1.5^a (0.3)	1.8^a (0.5)	3.3^a (1.8)	4.5^a (2.0)	1.8^a (0.2)	6.8^a (2.2)	5.1^a (4.1)	3.8^a (2.9)
	1.1-2.5	1.7-5.4	0.7-2.2	1.0-2.9	1.2-6.8	1.8-8.3	1.5-2.2	2.5-9.9	0.7-17.5	0.7-12.4
Ammonia (mg/L NH_3-N)	0.05^a (0.03)	0.07^a (0.01)	0.06^a (0.02)	0.06^a (0.02)	0.05^a (0.02)	0.06^a (0.02)	0.07^a (0.02)	0.01^a (0.01)	0.05^a (0.02)	0.06^a (0.02)
	<0.01-0.10	0.05-0.08	<0.01-0.10	0.03-0.09	0.01-0.09	0.03-0.10	0.04-0.11	<0.01-0.02	<0.01-0.09	0.01-0.08
Nitrate-nitrite (mg/L NO_3^--N, NO_2^--N)	0.06^a (0.21)	0.40^a (0.15)	0.63^a (0.17)	0.82^a (0.26)	0.64^a (0.04)	0.61^a (0.03)	0.63^a (0.21)	0.50^a (0.33)	0.16^a (0.04)	0.16^a (0.02)
	0.14-0.86	0.11-0.57	0.15-0.88	0.09-1.31	0.56-0.68	0.56-0.64	0.30-1.01	0.06-1.11	0.05-0.25	0.11-0.20
Reactive phosphorus (mg/L PO_4^{3-})	0.28^a (0.04)	0.19^b (0.04)	0.24^a (0.03)	0.22^a (0.01)	0.18^a (0.04)	0.17^a (0.02)	0.08^a (0.01)	0.11^a (0.03)	0.10^a (0.01)	0.13^a (0.02)
	0.19-0.32	0.12-0.26	0.18-0.30	0.20-0.24	0.11-0.26	0.14-0.19	0.07-0.10	0.06-0.17	0.06-0.13	0.09-0.16
Total alkalinity (mg/L as $CaCO_3$)	11.8^a (<0.1)	11.1^a (0.4)	11.8^a (0.4)	11.5^a (0.3)	11.5^a (0.8)	11.0^a (0.8)	11.7^a (0.4)	9.7^a (0.6)	13.6^a (0.4)	13.8^a (0.5)
	11.8-11.9	10.5-11.9	11.0-12.5	11.0-12.0	10.0-12.5	9.5-12.0	11.0-12.5	8.5-10.5	12.5-14.5	12.5-14.5
Total hardness (mg/L as $CaCO_3$)	11.3^a (0.4)	10.9^a (0.4)	11.5^a (0.5)	11.9^b (0.4)	10.7^a (0.8)	10.3^a (0.9)	11.8^a (0.4)	10.0^a (0.6)	14.0^a (1.0)	13.3^a (0.7)
	10.6-11.7	10.4-11.7	10.6-12.6	11.1-12.8	9.0-11.5	8.5-11.5	11.0-12.5	9.0-11.0	11.5-16.5	11.5-15.0

Table 6. Mean, standard error (in parentheses), and range of water quality measurements during summer (July-September) 2005 and 2006 samples taken inside and outside eight experimental exclosures on Lookout Shoals Lake. Within each parameter and year, different superscript letters indicate significant differences.

	Summer 2005		Summer 2006	
	Inside	Outside	Inside	Outside
Temperature (^{o}C)	24.7^{a} (0.2)	24.7^{a} (0.2)	26.2^{a} (0.5)	26.1^{a} (0.5)
	23.6-26.5	23.7-26.9	23.0-29.6	23.0-29.6
Dissolved oxygen (mg/L)	5.8^{a} (0.2)	5.5^{a} (0.2)	5.6^{a} (0.1)	5.5^{a} (0.1)
	4.2-9.1	4.1-9.0	4.7-6.8	4.7-7.0
pH	6.7^{a} (<0.1)	6.6^{b} (<0.1)	7.1^{a} (0.1)	7.1^{a} (<0.1)
	6.5-6.9	6.4-6.9	6.6-7.5	6.6-7.5
Chlorophyll *a* (µg/L)	15.3^{a} (4.0)	10.0^{a} (2.1)	11.4^{a} (1.7)	10.4^{a} (1.5)
	0.9-94.6	0.7-33.2	1.7-35.3	1.2-21.4
Ammonia (mg/L NH_3-N)	0.04^{a} (0.01)	0.05^{a} (0.01)	0.05^{a} (0.01)	0.05^{a} (0.01)
	<0.01-0.10	<0.01-0.15	<0.01-0.12	<0.01-0.15
Nitrate-nitrite (mg/L NO_3^--N, NO_2^--N)	0.32^{a} (0.05)	0.36^{a} (0.05)	0.46^{a} (0.07)	0.46^{a} (0.07)
	0.01-0.69	0.02-0.87	0.07-0.97	0.06-1.12
Reactive phosphorus (mg/L PO_4^{3-})	0.27^{a} (0.03)	0.26^{a} (0.02)	0.13^{a} (0.01)	0.15^{a} (0.02)
	0.10-0.87	0.10-0.48	<0.01-0.29	0.05-0.38
Total alkalinity (mg/L as $CaCO_3$)	11.8^{a} (0.1)	11.6^{a} (0.1)	11.0^{a} (0.2)	11.0^{a} (0.2)
	10.8-12.8	10.8-12.6	9.5-12.5	9.5-12.5
Total hardness (mg/L as $CaCO_3$)	11.4^{a} (0.1)	11.5^{a} (0.1)	11.1^{a} (0.2)	11.1^{a} (0.3)
	10.4-12.2	10.7-12.2	8.5-13.0	8.5-13.0

Table 7. Mean, standard error (in parentheses), and range of water quality measurements during fall-winter (October and November 2005; January, March, October, and November 2006; and, January, and March 2007) samples taken inside and outside eight experimental exclosures on Lookout Shoals Lake. Within each parameter and year group (2005-2006 and 2006-2007), different superscript letters indicate significant differences.

	Fall-winter 2005-2006		Fall-winter 2006-2007	
	Inside	Outside	Inside	Outside
Temperature (^oC)	11.5[a] (0.6)	11.5[a] (0.6)	11.3[a] (0.7)	11.3[a] (0.7)
	7.1-18.5	7.0-18.4	5.8-19.6	5.8-19.6
Dissolved oxygen (mg/L)	9.0[a] (0.3)	8.6[a] (0.2)	9.6[a] (0.3)	9.4[a] (0.3)
	6.6-11.5	6.6-10.7	6.7-12.6	6.7-12.6
pH	7.1[a] (<0.1)	7.1[a] (<0.1)	7.5[a] (0.1)	7.5[a] (0.1)
	6.9-7.4	6.9-7.4	6.9-8.4	7.1-8.2
Chlorophyll *a* (µg/L)	5.3[a] (1.0)	4.8[a] (0.9)	5.9[a] (1.4)	4.6[a] (1.0)
	1.2-29.6	0.7-28.3	0.6-37.3	0.7-20.4
Ammonia (mg/L NH_3-N)	0.04[a] (<0.01)	0.05[a] (0.01)	0.07[a] (0.01)	0.07[a] (0.01)
	0.01-0.08	<0.01-0.11	<0.01-0.21	<0.01-0.22
Nitrate-nitrite (mg/L NO_3^--N, NO_2^--N)	0.57[a] (0.06)	0.58[a] (0.05)	0.19[a] (0.03)	0.21[a] (0.02)
	0.03-1.13	0.03-0.98	0.03-0.89	0.04-0.77
Reactive phosphorus (mg/L PO_4^{3-})	0.23[a] (0.01)	0.22[a] (0.01)	0.12[a] (0.01)	0.12[a] (0.01)
	0.14-0.38	0.10-0.46	0.01-0.26	0.02-0.28
Total alkalinity (mg/L as $CaCO_3$)	11.2[a] (0.1)	11.7[a] (0.1)	13.4[a] (0.2)	13.5[a] (0.2)
	10.4-11.8	10.5-13.0	12.0-16.0	12.0-16.0
Total hardness (mg/L as $CaCO_3$)	11.8[a] (0.1)	11.9[a] (0.2)	13.75[a] (0.3)	13.9[a] (0.3)
	10.4-13.0	10.9-13.5	10.5-18.0	11.0-17.0

Table 8. Mean, standard error (in parentheses), and range of water quality measurements during spring (April, May, and June 2006 and April 2007) samples taken inside and outside eight experimental exclosures on Lookout Shoals Lake. Within each parameter and year, different superscript letters indicate significant differences.

	Spring 2006 (*N*=24)		Spring 2007 (*N*=8)	
	Inside	Outside	Inside	Outside
Temperature (^oC)	18.8^a (1.0)	18.9^a (1.0)	16.0^a (0.7)	15.9^a (0.6)
	14.3-25.7	14.3-25.8	13.7-18.8	13.9-18.4
Dissolved oxygen (mg/L)	7.7^a (0.2)	7.5^a (0.2)	9.1^a (0.2)	8.9^a (0.3)
	5.2-9.6	5.7-9.0	8.3-10.0	7.7-9.9
pH	6.9^a (<0.1)	6.9^a (<0.1)	7.4^a (<0.1)	7.2^a (0.1)
	6.7-7.1	6.7-7.0	7.1-7.5	6.5-7.5
Chlorophyll *a* (µg/L)	7.2^a (1.0)	7.1^a (1.0)	4.2^a (1.1)	4.7^a (1.3)
	1.6-19.4	1.4-18.5	1.4-10.5	1.3-11.2
Ammonia (mg/L NH_3-N)	0.06^a (0.01)	0.05^a (0.01)	0.05^a (0.01)	0.04^a (0.01)
	<0.01-0.20	<0.01-0.20	<0.01-0.10	<0.01-0.10
Nitrate-nitrite (mg/L NO_3^--N, NO_2^--N)	0.65^a (0.03)	0.59^a (0.03)	0.16^a (0.01)	0.16^a (0.01)
	0.41-1.17	0.30-1.07	0.13-0.20	0.11-0.20
Reactive phosphorus (mg/L PO_4^{3-})	0.17^a (0.01)	0.14^a (0.01)	0.12^a (0.03)	0.05^a (0.02)
	0.06-0.33	0.04-0.26	<0.01-0.26	<0.01-0.13
Total alkalinity (mg/L as $CaCO_3$)	11.9^a (0.2)	11.9^a (0.2)	12.8^a (0.2)	12.6^a (0.2)
	10.0-13.0	10.0-13.0	12.0-14.0	12.0-14.0
Total hardness (mg/L as $CaCO_3$)	11.1^a (0.2)	11.2^a (0.2)	12.6^a (0.1)	12.6^a (0.1)
	9.5-12.5	9.0-12.5	12.5-13.5	12.5-13.5

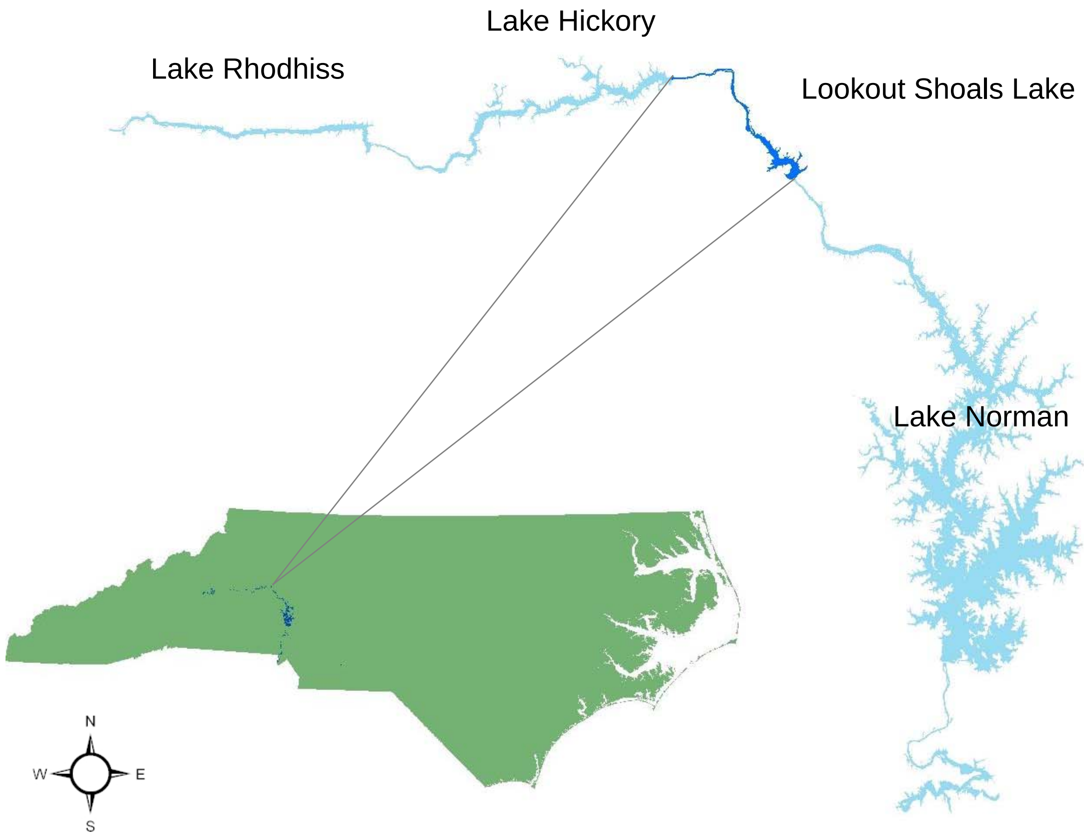

Figure 1. Location of study site, Lookout Shoals Lake, in relation to other Catawba River reservoirs within North Carolina.

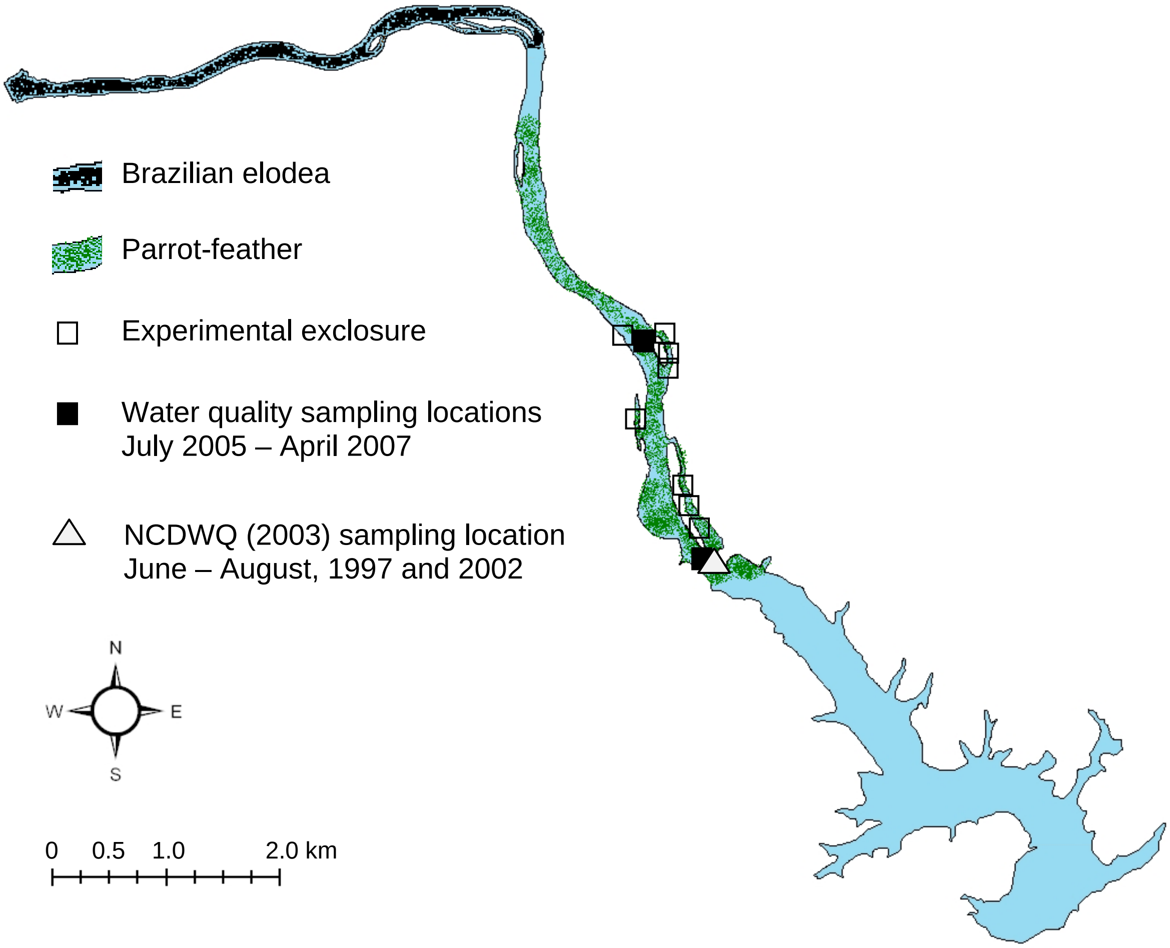

Figure 2. Parrot-feather and Brazilian elodea approximate coverage, and experimental exclosure and water quality sampling locations on Lookout Shoals Lake, North Carolina.

Figure 3. Experimental exclosures (6 m × 6 m × 2 m, 1.3-cm plastic mesh). The upper photograph was taken during exclosure construction with a 1-m drawdown of Lookout Shoals Lake. Lower photograph depicts normal lake level.

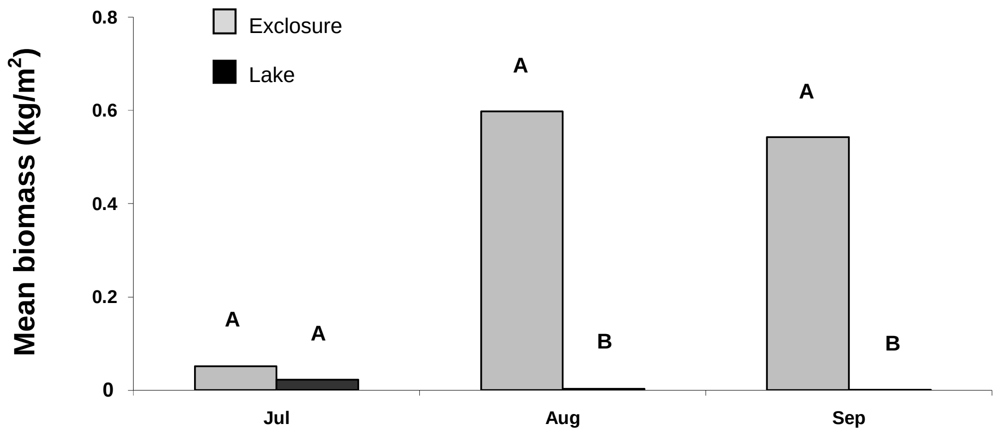

Figure 4. Mean biomass (kg/m^2) of parrot-feather collected from eight exclosures during July, August, and September on Lookout Shoals Lake, 2005. Different letters indicate significant differences within months.

Chapter II - High-Density Grass Carp Stocking Effects on Reservoir Fish Populations

Introduction

Stocking grass carp *Ctenopharyngodon idella* is a commonly applied technique used to control nuisance aquatic vegetation in reservoirs. Factors that influence the degree of aquatic vegetation control are stocking density (Fowler and Robson 1978), regional climate (Van Dyke et al. 1984; Bonar et al. 1993), abundance and species composition of the aquatic plant community (Prowse 1971; Opuszynski 1972; Van Dyke et al. 1984), and relative grass carp feeding preferences for the plant species (Fowler and Robson 1978). Among these factors, manipulation of grass carp stocking densities is the typical method available to aquatic plant managers to achieve control of some types of aquatic vegetation. High-density grass carp stocking may help managers control aquatic plant species not preferentially consumed by grass carp and difficult to control with this technique. High stocking densities may result in the rapid elimination of aquatic vegetation preferentially consumed by grass carp, forcing the fish to consume aquatic plant species they would not ordinarily consume, or risk starvation (Fowler and Robson 1978; Pine and Anderson 1991; Catarino et al. 1997). Achieving control of vegetation that is not preferentially consumed by grass carp using high stocking densities could result in elimination of all aquatic vegetation in the system (Shireman et al. 1985; Kohler and Courtenay 1986; Allen and Wattendorf 1987; Bain 1993; Cassani 1995). Complete elimination of invasive vegetation would limit its spread (Cronk 1995), but little is known regarding the impacts that such a management strategy may exert on reservoir fish communities.

Aquatic plants perform valuable ecological functions that ultimately support fish and wildlife species, as well as provide aesthetic appeal. Aquatic vegetation and the associated invertebrate communities provide important food sources for fish and waterfowl (Cross 1969; Vinogradov and Zolotova 1974; Voigts 1976; Schram and Jirka 1989), spawning and nursery habitat for fish (Muncy 1962; Franklin and Smith 1963; Holland and Huston 1984), and are influential in fish predator-prey relationships (Hall and Werner 1977; Mittlebach 1981; Savino and Stein 1982). However, reservoirs typically have little or no native aquatic vegetation (Smart et al. 1998) and the resident fish populations should have few established relationships with aquatic plants.

Native plant species provide an array of benefits in a balanced ecosystem where their growth is primarily controlled by natural adaptive processes (Holm et al. 1969; Frye 1972; Aulbach-Smith and de Kozlowski 1996). Conversely, invasive plants often spread unhindered by these same natural controls and become a nuisance (Frye 1972; Aulbach-Smith and de Kozlowski 1996). Invasive aquatic vegetation can provide similar benefits as non-invasive vegetation, but without natural control, invasive plants can suppress other aquatic plants (Mitchell 1969; Aiken et al. 1979; Madsen et al. 1991; de Winton and Clayton 1996; Boylen et al. 1999; Ali and Soltan 2006), and adversely affect fish populations by excluding fish from spawning sites (Keast 1984) and providing excessive cover for prey fish species (Swingle and Smith 1941; Smith and Swingle 1942; Heman et al. 1969; Frye 1972; Barnett and Schneider 1974).

Management of invasive aquatic plants typically becomes a management dilemma when these plants interfere with human activities and water use. These plants may become so dense as to clog municipal and agricultural water sources and power generation facilities (Holm et al. 1969; Vinogradov and Zolotova 1974), hinder fishing and boating activities, and provide habitat for nuisance insect species (Aliyev and Bessmertnaya 1968; Orr and Resh 1992). Management of nuisance aquatic plants is a common challenge for water resource managers.

Stocking grass carp is a technique for the management of nuisance aquatic vegetation and offers several advantages over mechanical or chemical controls in aquatic plant management including low cost, labor saving, and low risk to public health (Sutton 1977; Rottman 1977; Bailey 1978; Allen and Wattendorf 1987; Cassani 1995; Li and Moyle 1999). Grass carp are herbivorous fish native to China and Siberia and can grow to more than 50 kg and survive more than 20 years (Lin 1935a; Nikol'skii 1954; Avault 1965; Cross 1969; Bailey and Boyd 1972; Jenkins and Burkhead 1993). Diploid grass carp were introduced to the U.S. for aquatic macrophyte control in 1963 and have been prohibited in many states because they may reproduce and impact native aquatic plant and fish communities (Stevenson 1965; Sills 1970; Sutton 1977; Pflieger 1978; Allen and Wattendorf 1987; Cassani 1995). Since the development of sterile triploid grass carp in 1983, which alleviated some concern over the long-term impacts from establishment of viable grass carp populations, grass carp have become widely used as a control agent for aquatic nuisance vegetation (Bain 1993; Cassani 1996).

Application of this technique typically involves varying grass carp stocking densities in an effort to achieve the desired amount of vegetation control. Thus, high-density grass carp stocking to control persistent nuisance aquatic vegetation is receiving increased attention as a management application, but scientific evaluation has been limited. Of special concern would be the efficacy for vegetation control and effects on native fishes.

High grass carp densities and the potential loss of aquatic vegetation have raised concerns with fisheries managers regarding the effects on the native fish community. Grass carp in Asia have been reported to be omnivorous, consuming plants, insects, and fish (Lin 1935b), but studies conducted in the U.S. indicate that grass carp do not compete directly with native fishes for food resources (Kilgen and Smitherman 1971; Bailey and Boyd 1972; Terrell and Fox 1974; Rottman and Anderson 1976; Colle et al. 1978; Lembi et al. 1978; Lewis 1978). Thus, some concern remains that high grass carp densities could cause significant habitat or physical disturbances that may impact resident fish populations (Vinogradov and Zolotova 1974; Kohler and Courtenay 1986).

Results from previous research on the effect of vegetation removal by grass carp on resident fish communities are inconclusive. In a seven-year study in Lake Marion, South Carolina, Killgore et al. (1998) found no negative impacts on the littoral fishery following aquatic vegetation removal by grass carp. Bailey (1978) observed little change in the total standing crop and condition factors of fish in 31 Arkansas lakes following complete elimination of submersed vegetation. In Lake Baldwin, Florida, after removal of submersed macrophytes by grass carp, Shireman et al. (1985) observed an increase in harvestable size largemouth bass and relative weight values equal to or higher than those seen during high vegetation coverage; fluctuations in numbers and standing crop of all harvestable fish species appeared to be unrelated to macrophyte abundance. However, Bettoli et al. (1993) noted biomass declines for 8 of 17 fish species in Lake Conroe, Texas, after vegetation removal by grass carp. The mechanisms producing these variable results are unclear, and demonstrate the need for additional research.

Stocking densities of grass carp that are higher than typically recommended may be a cost-effective means to control aquatic plants that are not preferentially consumed by grass carp, but additional insight into the effects of such a practice on reservoir fish communities is needed for informed aquatic plant and ecosystem management. The objective of this study was to evaluate the effects of a high-density triploid grass carp stocking on the abundance and distribution of native fish populations in Lookout Shoals Lake, a piedmont North Carolina reservoir.

Study Area and Management

Lookout Shoals Lake is a 528-ha reservoir located between Lake Hickory and Lake Norman on the Catawba River in the western piedmont of North Carolina (Figure 1). The lake is jointly managed by Duke Energy (e.g., water levels and aquatic macrophytes) and the North Carolina Wildlife Resources Commission (e.g., the fishery). It was impounded in 1915 by the construction of the Lookout Shoals Dam and Hydroelectric Station for power production, which is the primary use of the reservoir; municipal water supply and recreation are secondary uses (Duke Power 1999; NCDWQ 2003). Parrot-feather *Myriophyllum aquaticum* and Brazilian elodea *Egeria densa* are two invasive aquatic plant species that dominate the submersed aquatic plant community of Lookout Shoals Lake (Figure 2). Both were introduced from South America and often form large, extensive mats on the water surface that interfere with ecological processes and human uses of the reservoir (Weatherby 1932; Sutton 1985; Blackburn et al. 1969; Aulbach-Smith and de Kozlowski 1996).

Water-level management and aquatic herbicides have been previously utilized in attempts to control parrot-feather and Brazilian elodea abundances and distributions. Brazilian elodea has been successfully controlled with winter drawdowns of this reservoir, which have restricted the infestation to the riverine section from the Oxford Hydroelectric Dam, that impounds Lake Hickory, downstream about five km to the upstream impounded area of Lookout Shoals Lake. Parrot-feather has infested approximately 90 ha of the littoral zone of the upper reservoir due to the ineffectiveness of winter drawdowns at controlling its spread and has been selectively controlled using aquatic herbicides (Ken Manuel, Duke Energy, personal communication). Recent installation of a municipal water intake facility on Lookout Shoals Lake, however, has resulted in increased restrictions on the use of aquatic herbicides, and other control options were sought for the management of this invasive aquatic plant. While grass carp are a common control option for nuisance aquatic plants, parrot-feather is not a food source preferred by grass carp (Avault 1965; Pine and Anderson 1991; Catarino et al. 1997). This presented an experimental opportunity to research a high-density grass carp stocking management technique to control a nuisance aquatic plant species that was not a food source preferred by grass carp.

In May 2005, Lookout Shoals Lake was stocked with approximately 9,200 grass carp (100 fish per vegetated ha). This stocking rate is twice the recommended rate for North Carolina public waters (Rice et al. 1999), and is characterized as high relative to those applied in previous studies (Bailey 1978; Fowler and Robson 1978; Leslie et al. 1983; de Kozlowski 1994; Hanlon et al. 2000). These fish were approximately 350 mm in total length and 525 g in weight.

Methods

The impact of a high-density grass carp stocking to control parrot-feather on fish populations in Lookout Shoals Lake was evaluated over time using shoreline electrofishing catch data collected prior to and after grass carp were stocked. Fish populations associated with eight, 100-m shoreline transects (Figure 2) were sampled seasonally during spring, summer and fall of 2004-2007 using a boat-mounted electrofisher operating at 4-6 A, 120 pulses/sec, DC. These transects are located in the upper area of Lookout Shoals Lake, where the parrot-feather infestations were extensive and any effects of grass carp were likely to occur. Sampling these transects began in summer 2004 and continued through spring 2005, prior to grass carp stocking and from summer 2005 to spring 2007 after grass carp had been stocked. All sizes and species of fish collected were identified to species, and individually weighed (±1 g) and measured (±1 mm total length, TL).

In addition, fish population data from ten, 300-m shoreline transects (Figure 2) sampled during spring using similar techniques as described above were used for historical comparisons. These data were collected by Duke Energy biologists in 1994-1997, 2000, and 2005-2007 from shoreline transects representing available habitat types, situated throughout the middle and lower lake. All sizes and species of fish collected were identified to species, counted and weighed in aggregate (±1 g).

Catch rates for all species in aggregate, largemouth bass *Micropterus salmoides*, bluegill *Lepomis macrochirus*, redbreast sunfish *Lepomis auritus*, and yellow perch *Perca flavescens* were calculated as number of fish per 100 m of shoreline (number/100 m) and biomass of fish per 100 m of shoreline (kg/100 m). Mean lengths were calculated and distributions plotted for all bluegill and yellow perch collected from the fall samples of the seasonal shoreline electrofishing transects. Mean relative weights were calculated for largemouth bass greater than 150 mm TL, bluegill greater than 80 mm TL, and yellow perch greater than 100 mm TL, collected from the seasonal shoreline electrofishing transects, using equations from Henson (1991), Hillman (1982), and Willis et al. (1991), respectively. Initially, a factorial analysis of variance with year, season, and transects as main effects was conducted for catch rates of all fish species in aggregate, bluegill, largemouth bass, redbreast sunfish, and yellow perch from the seasonal transects. Year, season, and transects were significant main effects with no interaction. Then, catch rates were analyzed by season for annual differences before and after grass carp stocking. Evidence of non-normality was detected in catch rates which was not remedied by data transformation. Thus, a Kruskal-Wallis test was utilized to detect differences among years within seasons and before and after grass carp stocking, and a Nemenyi multiple comparison test was used to detect differences between years within each season (Zar 1996, Statistical Analysis Software 2007). All statistical comparisons were considered significant at a probability of 0.05 (alpha).

Results

Catch for all years combined from the eight, 100-m seasonal transects totaled 5,276 fish representing 29 species (Table 1). Most common species by number were bluegill (38%), yellow perch (24%), largemouth bass (14%), and redbreast sunfish (8%). Catch for all years from the ten, 300-m annual shoreline electrofishing transects totaled 7,462 fish representing 31 species and one hybrid complex (Table 2). Most common species by number were bluegill (47%), yellow perch (15%), largemouth bass (13%), and redbreast sunfish (12%).

100-m seasonal shoreline electrofishing transects

Analyses of catch rate by number (number/100 m) from the eight, 100-m seasonal shoreline electrofishing transects (Figure 3) detected no significant differences for all fish species in aggregate before and after grass carp stocking. Single-species analyses of bluegill, largemouth bass, redbreast sunfish, and yellow perch indicated significant differences in some species and seasons after grass carp stocking. Bluegill numbers were significantly higher during the second spring after grass carp stocking (2007). Mean number of bluegill was 15 fish/100 m (3.7 SE) during spring 2006 versus 53 fish/100 m (19.7 SE) during spring 2007. Largemouth bass numbers showed a decreasing trend during summer and fall with statistical significance detected in the second year after grass carp stocking (2007). Mean number of largemouth bass was 18 fish/100 m (4.7 SE) during summer 2004 versus 11 fish/100 m (1.9 SE) during summer 2006. Redbreast sunfish displayed a downward trend during summer, and a significant reduction was detected during fall after grass carp stocking. Mean number of redbreast sunfish was 9 fish/100 m (2.5 SE) during fall 2004 versus 3 fish/100 m (1.2 SE) during fall 2006. Catch rates of yellow perch decreased significantly during spring and summer during the second year after grass carp stocking, and yellow perch numbers were significantly lower in both fall samples after grass carp stocking. Mean number of yellow perch was 33 fish/100 m (9.0

SE) during spring 2005 versus 10 fish/100m (3.9 SE) during spring 2007, 25 fish/100 m (8.2 SE) during summer 2004 versus 8 fish/100 m (3.3 SE) during summer 2006, and 37 fish/100 m (11.2 SE) during fall 2004 versus 2 fish/100 m (0.6 SE) during fall 2006.

Biomass (kg/100 m) of fish species sampled from the eight, 100-m seasonal shoreline transects followed similar trends as those of numerical catch rates (Figure 4). No significant differences in biomass were detected for bluegill and largemouth bass before and after grass carp stocking, but significant differences were revealed for all species in aggregate, redbreast sunfish, and yellow perch. The elevated biomass of all species total observed during the second spring and first fall after grass carp stocking was due to the occurrence of common carp *Cyprinus carpio* in the catch that occurs sporadically in reservoirs. No significant differences in biomass of redbreast sunfish were detected in spring; a significant decreasing trend during summer was detected in the second summer after grass carp stocking. Mean biomass of redbreast sunfish was 0.26 kg/100 m (0.09 SE) during summer 2004 versus 0.06 kg/100 m (0.02 SE) during summer 2006. Redbreast sunfish biomass was significantly reduced in both fall samples after grass carp stocking. Mean biomass of redbreast sunfish was 0.26 kg/100 m (0.09 SE) during fall 2004 versus 0.06 kg/100 m (0.02 SE) during fall 2006. A decreasing trend occurred in the biomass of yellow perch during spring, which was significant during the second spring after grass carp stocking. No difference was detected in yellow perch biomass between the summer before grass carp stocking and the first summer after stocking; a significant reduction was detected during the second summer after stocking. Mean biomass of yellow perch was 0.56 kg/100 m (0.08 SE) during summer 2004 versus 0.23 kg/100 m (0.10 SE) during summer 2006.

Yellow perch showed a significant reduction in biomass in both fall samples after grass carp stocking. Mean biomass of yellow perch was 0.92 kg/100 m (0.27 SE) during fall 2004 versus 0.14 kg/100 m (0.04 SE) during fall 2006.

Mean lengths and length distributions of bluegill and yellow perch showed some variability before and after grass carp stocking (Figures 5 and 6). Mean length of bluegill during fall 2004, before grass carp stocking, was 78 mm (2.0 SE). After grass carp stocking, mean length of bluegill was 104 mm (2.5 SE) during fall 2005 and 93 mm (1.7 SE) during fall 2006. Mean lengths of yellow perch during fall 2004, before grass carp stocking, was 121 mm (2.4 SE). After grass carp stocking, mean length of yellow perch was 160 mm (7.5 SE) during fall 2005 and 179 mm (8.0 SE) during fall 2006.

Relative weight values of bluegill, largemouth bass, and yellow perch showed statistically significant differences for each species during at least one season throughout the course of this study (Figure 7). Largemouth bass relative weight increased during the second spring after grass carp stocking. Relative weight of largemouth bass was 94 (1.7 SE) during spring 2005 versus 102 (2.6 SE) during spring 2007. Bluegill relative weight increased during the first summer after grass carp stocking. Relative weight of bluegill was 91 (1.2 SE) during summer 2004 versus 95 (0.9 SE) during summer 2005. Yellow perch relative weight decreased during the second summer and the first and second fall after grass carp stocking. Relative weight of yellow perch was 76 (0.5 SE) during summer 2004 versus 73 (1.0 SE) during summer 2006, and 70 (0.4 SE) during fall 2004 versus 67 (1.2 SE) during fall 2006.

300-m annual shoreline electrofishing transects

Catch rate by number (number/100 m) and biomass (kg/100 m) from the ten, 300-m annual shoreline electrofishing transects for all fish species total, largemouth bass, bluegill, redbreast sunfish, and yellow perch indicated no significant differences before (1994-2005) and after (2006-2007) grass carp stocking (Figures 8-12). Within the time frame of this study (2005-2007), the number of yellow perch reflect a similar reduction as observed in the eight, 100-m seasonal shoreline electrofishing transects. Mean number of yellow perch was 9 fish/100m (1.9 SE) during 2005 versus 1.2 fish/100 m (0.80 SE) during 2007. However, this decline observed in annual data is not significantly different from the historical numbers (1994-2000) of Lookout Shoals Lake.

Biomass followed similar trends as the numerical catch rates during annual sampling. No significant differences in biomass were detected for all fish species in aggregate, largemouth bass, bluegill, redbreast sunfish, or yellow perch before (1994-2005) and after (2006-2007) grass carp stocking. Similar to the numerical catch rates, within the time frame of this study (2005-2007), biomass of yellow perch declined as observed in the eight 100-m seasonal shoreline transects. Mean biomass of yellow perch was 0.16 kg/100 m (0.05 SE) during 2005 versus 0.02 kg/100 m (0.01 SE) during 2007. However, as with yellow perch numbers, these reductions in biomass were not significantly different from historical levels (1994-2000).

Discussion

Analysis of the fish community of Lookout Shoals Lake revealed seasonal and annual variability for most species common to southeastern U.S. reservoir fish communities. There were two marked changes in the fish community that I detected over the course of this study following grass carp stocking: increases in the number of bluegill and decreases in the number and biomass of yellow perch. These changes occurred in both the lower and upper lake areas, and associated research indicated that grass carp effectively removed all aquatic vegetation in these areas (see chapter 1).

The mechanisms associated with an increase in bluegill numbers may be related to nutrient dynamics and trophic relationships. There was a significant increase in the number of bluegill during the spring of 2007 in the area of Lookout Shoals Lake where the parrot-feather infestation was eliminated. This increase in number of bluegill was distributed across the range of length classes (Figure 5). Bluegill numbers were also slightly elevated in the lower lake area in the spring of 2007. Macrophytes can sequester nutrients that could otherwise be allocated to phytoplankton (Bachmann et al. 2004; Boyd 1971). The chlorophyll *a* concentrations in Lookout Shoals Lake increased two-fold from the summer of 2005 to the summer of 2006 (see chapter 1). Chlorophyll *a* provides a an index of phytoplankton primary production (Brylinsky and Mann 1971) and can serve as a predictor of zooplankton biomass (McCauley and Kalff 1981). Jones and Hoyer (1982) found mean summer chlorophyll *a* concentrations to be a strong indicator of annual sport fish production in midwestern lakes and reservoirs.

Mittlebach (1981) found bluegill prey encounter rates to be higher in open water than in vegetated habitat, and bluegill forage size-selectively on large zooplankton (Bartell 1982) and maximize prey biomass consumed (Werner and Hall 1974). Small bluegill may seek cover in littoral vegetation where predation risk is lower (Mittlebach 1981; Savino and Stein 1982; Werner and Hall 1988; Hayse and Wissing 1996) and foraging efficiency is reduced (Werner et al. 1983). Large zooplankton may utilize aquatic macrophytes as a refuge from planktivorous fish (Lauridsen and Lodge 1996). A decrease in zooplankton refugia coupled with an increase in zooplankton production, may have increased bluegill foraging efficiency. However, the loss of macrophyte cover would also render small bluegill more vulnerable to predation, but increased phytoplankton densities (as indicated by elevated chlorophyll a concentration) could provide compensatory protection from predators by reducing visibility in the water column (Gardner 1981, Sweka and Hartman 2001, De Robertis et al. 2003). A potential decrease in interspecific competition from the reduction in yellow perch numbers coupled with higher zooplankton densities may explain the increase in the bluegill population during spring 2007.

There was a marked decrease in the numbers of yellow perch seen in the first fall following vegetation removal that carried through the spring of 2007 in both areas of the lake. The loss of vegetative habitat may have impacted the yellow perch population in two ways. The invasive vegetation may have provided structure advantageous to yellow perch spawning or to avoid predation.

Length distributions of yellow perch following grass carp stocking indicate recruitment failure (Figure 6). Loss of the major submersed aquatic vegetation infestation may have depleted yellow perch spawning habitat. Female yellow perch are known to select submersed brush and vegetation for attaching egg strands (Muncy 1962). This necessary structure may be limited in Lookout Shoals Lake as with many reservoirs due to lakefront development, fluctuating water levels, and basin clearing (Bryan and Scarnecchia 1992; Christensen et al. 1996; Summerfelt 1999).

Increased predation on yellow perch by largemouth bass following the elimination of the submersed vegetation may be a factor contributing to the reduction in the yellow perch population. Loss of vegetative cover with the resulting increase in predation vulnerability of small littoral fish species has been linked to reductions in those populations (Bettoli et al. 1993; Pothoven et al. 1999). However, the bluegill faced a similar situation and their dynamics did not follow the same trend as yellow perch. Prey behavior and predator feeding preferences may address this issue. It is possible the predatory fish species of Lookout Shoals Lake, primarily largemouth bass, preyed selectively on yellow perch over bluegill. Research on predator-prey relationships clearly indicates that yellow perch are more vulnerable than bluegill to piscivore predation. Seaburg and Moyle (1964) found that northern pike *Esox lucius*, walleye *Sander vitreus*, and largemouth bass selectively preyed upon yellow perch over small centrarchid and cyprinid species in two western Minnesota lakes. Margenau et al. (1998) also found a preference for yellow perch over bluegill in the diet of northern pike in one Wisconsin lake. Similarly, Clady (1974) found adult largemouth bass and yellow perch selectively preyed upon small yellow perch in a northern Michigan lake.

Bluegill are flexible in their habitat use and foraging habits (Werner and Hall 1977; Ehlinger and Wilson 1988) and could influence the decline in yellow perch numbers. Schneider (1997) attributed competition for large zooplankton and benthos as a causal mechanism when yellow perch biomass declined after introduction of bluegill. Hanson and Legget (1985) cited interspecific competition in favor of pumpkinseed *Lepomis gibbosus* to explain yellow perch reduced growth coincident with increased abundance of this centrarchid.

This short-term reduction in the numbers of yellow perch may not be indicative of the long-term dynamics of the yellow perch population of Lookout Shoals Lake. Intraspecific competition could have been elevated during the parrot-feather infestation beyond that of a typical stable condition. The dense vegetation may have provided cover sufficient to reduce predation to the point allowing for atypically high densities of yellow perch. This can suppress growth rates in a prey population (Tonn et al. 1992). Of the yellow perch sampled before vegetation removal, 67% were less than stock size (130 mm, Gabelhouse 1984) opposed to 47% after vegetation removal (Figure 6). This change in population size structure could be simply due to selective predation of smaller individuals by largemouth bass. This might also indicate an early stage in transition to a lower-density, faster-growing population of yellow perch. Lott (1991) found three characteristics indicative of high quality yellow perch populations to be fast growth, low density, and low and inconsistent recruitment. However, I did not detect an increase in the condition of yellow perch in this study; rather, relative weights decreased over time during summer and fall (Figure 7). Continued monitoring of the yellow perch population of Lookout Shoals Lake could provide data required to assess the long-term condition of the population.

Management Implications

In this study, as in other studies where grass carp have eliminated submersed aquatic plant species (see chapter 1), the effects on fish populations appear to be species specific. Aquatic vegetation provides critical fish spawning and nursery habitat (Muncy 1962; Franklin and Smith 1963; Holland and Huston 1984) and is influential in predator-prey relationships that structure fish communities (Hall and Werner 1977; Mittlebach 1981; Savino and Stein 1982). Changes observed in fish populations after vegetation removal certainly vary with region-specific variables such as fish and aquatic plant community compositions. Fish species that are dependent upon or associated with littoral vegetation during their life cycle, such as bluegill and yellow perch in my study, are most likely to be adversely affected when high densities of grass carp are introduced to the system.

My results suggest that the Lookout Shoals Lake sport fishery should not be greatly affected by this high-density grass carp stocking and subsequent vegetation removal. Creel surveys of two other Catawba River reservoirs, Lake Rhodhiss and Lake Hickory, indicate yellow perch are less sought by anglers than most other taxa (Duke Energy 2002a, 2002b). Among carp, suckers *Catostomidae*, catfish *Ictaluridae*, white bass *Morone chrysops*, striped bass *Morone saxatilis*, sunfish *Centrarchidae*, smallmouth bass *Micropterus dolomieu*, largemouth bass, crappie *Pomoxis* spp., and yellow perch, yellow perch are fourth in total catch after sunfish, crappie and largemouth bass, and eighth in total harvest before sucker and smallmouth bass in Lake Rhodhiss. Of golden shiner *Notemigonus crysoleucas*, catfish, white bass, striped bass, sunfish, largemouth bass, crappie, and yellow perch, yellow perch are sixth in total catch before white bass and golden shiner, and fifth in total harvest before white bass, largemouth bass, and golden shiner in Lake Hickory.

When invasive aquatic vegetation infests a system, fish species that can adapt favorably may benefit. Native fish populations in reservoirs of the North Carolina piedmont region are generally riverine species that survive and flourish to varying degrees in the impounded waterbody. These reservoirs typically support little endemic aquatic vegetation, and this is the case for Lookout Shoals Lake. The lotic-adapted fishes of reservoir systems have not evolved with dense stands of aquatic vegetation, so effects of invasive plant control may be less pronounced on this fish community, relative to those in natural lentic ecosystems, where plant-fish ecological relationships have been established and evolved over a considerable period of time. In this unique reservoir situation, the effects of high grass carp densities on the native fish community would contrast invasive aquatic plant infestation. In other aquatic systems, the effects of high-density grass carp stocking would differ significantly as fish and aquatic plant communities vary. Water resource managers should consider these factors when strategically planning aquatic plant management approaches.

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Table 1. Total numbers of fish collected by electrofishing eight, 100-m seasonal shoreline transects on Lookout Shoals Lake during spring, summer, and fall, 2004-2007.

	Spring			Summer			Fall		
Taxa	2005	2006	2007	2004	2005	2006	2004	2005	2006
Alewife *Alosa pseudoharengus*					28				
Black crappie *Pomoxis nigromaculatus*	4	2		2	15	2	7	11	3
Bluegill *Lepomis macrochirus*	119	119	423	162	137	190	223	184	427
Bluehead chub *Nocomis leptocephalus*			1						
Bowfin *Amia calva*		1		1	1			4	
Brown bullhead *Ameiurus nebulosus*					2				
Channel catfish *Ictalurus punctatus*					1			1	
Common carp *Cyprinus carpio*	4	1	14	2	4	4		14	
Eastern mosquitofish *Gambusia holbrooki*							1		
Flat bullhead *Ameiurus platycephalus*	1	3	2	1	2	2	2	3	5
Gizzard shad *Dorosoma cepedianum*				1	4	16	1	4	14
Golden shiner *Notemigonus crysoleucas*	3				4		24	1	
Grass carp *Ctenopharyngodon idella*			2		2			3	
Green sunfish *Lepomis cyanellus*	1		1	1	2	3	2		1
Largemouth bass *Micropterus salmoides*	45	39	39	147	130	91	117	74	58
Notchlip redhorse *Moxostoma collapsum*	1	8	5	22	21	14	4	24	1
Pumpkinseed *Lepomis gibbosus*			10	5			4		26
Rainbow trout *Oncorhynchus mykiss*			1						
Redbreast sunfish *Lepomis auritus*	54	61	28	68	43	27	63	30	30
Redear sunfish *Lepomis microlophus*	3		9	3	9	5	11	8	6
Snail bullhead *Ameiurus brunneus*				1		2			
Spottail shiner *Notropis hudsonius*	40		8	2		1	2	1	
Tessellated darter *Etheostoma olmstedi*	23	3		18	3	1	6	3	2
Threadfin shad *Dorosoma petenense*						107			4
Warmouth *Lepomis gulosus*	3	5	22	2	1	9	2	3	8
White catfish *Ameiurus catus*	14	10		8	18	3	5	2	
White perch *Morone americana*	78	2			7	1	3	1	
White sucker *Catostomus commersonii*	1	3	2	1	1	2	1	1	
Yellow perch *Perca flavescens*	262	142	80	199	165	63	299	32	19
Total	656	399	647	646	600	543	777	404	604
Total number of species	17	14	16	19	22	19	19	20	14

Table 2. Total numbers of fish collected by electrofishing at ten, 300-m annual shoreline transects on Lookout Shoals Lake during spring of 1994-1997, 2000, and 2005-2007.

Taxa	1994	1995	1996	1997	2000	2005	2006	2007
Alewife *Alosa pseudoharengus*						3		1
Black crappie *Pomoxis nigromaculatus*	8	6	1	4	14			2
Bluegill *Lepomis macrochirus*	380	590	548	268	181	562	227	735
Brassy jumprock *Scartomyzon* sp.	1							
Brown bullhead *Ameiurus nebulosus*	1	3						
Common carp *Cyprinus carpio*	8	6	2	2		6	7	
Fantail darter *Etheostoma flabellare*		1						
Flat bullhead *Ameiurus platycephalus*		6	1	1	4	1	3	1
Flathead catfish *Pylodictis olivaris*							1	
Gizzard shad *Dorosoma cepedianum*	11	29	16	2	2	9	15	2
Golden shiner *Notemigonus crysoleucas*		1	1	2	3	4	1	3
Grass carp *Ctenopharyngodon idella*							3	3
Green sunfish *Lepomis cyanellus*			8	1	5	3	5	6
Greenfin shiner *Cyprinella chloristia*					1	8		8
Unidentified hybrid sunfish *Lepomis* sp.		1	4		1		1	
Largemouth bass *Micropterus salmoides*	112	245	116	86	127	76	111	89
Notchlip redhorse *Moxostoma collapsum*	2	1	1	2		1	1	6
Piedmont darter *Percina crassa*		1						
Pumpkinseed *Lepomis gibbosus*	43	20	2			3		5
Quillback *Carpiodes cyprinus*		2						
Redbreast sunfish *Lepomis auritus*	148	303	95	58	89	45	25	119
Redear sunfish *Lepomis microlophus*	13	39	8	12	30	13	8	43
Spottail shiner *Notropis hudsonius*	6			1		98		
Striped bass *Morone saxatilis*	1							
Striped jumprock *Moxostoma rupiscartes*			1			1		
Tessellated darter *Etheostoma olmstedi*			1	2	5	2	2	8
Warmouth *Lepomis gulosus*	11	36	30	16	11	4	3	31
White bass *Morone chrysops*					1	1		
White catfish *Ameiurus catus*	43	61	36	21	8	44	13	7
White crappie *Pomoxis annularis*		5			1			
White perch *Morone americana*						1		
Yellow perch *Perca flavescens*	107	220	103	212	96	260	71	37
Total number	895	1,576	974	690	579	1,145	497	1,106
Total number of species	16	19	17	16	16	21	16	18

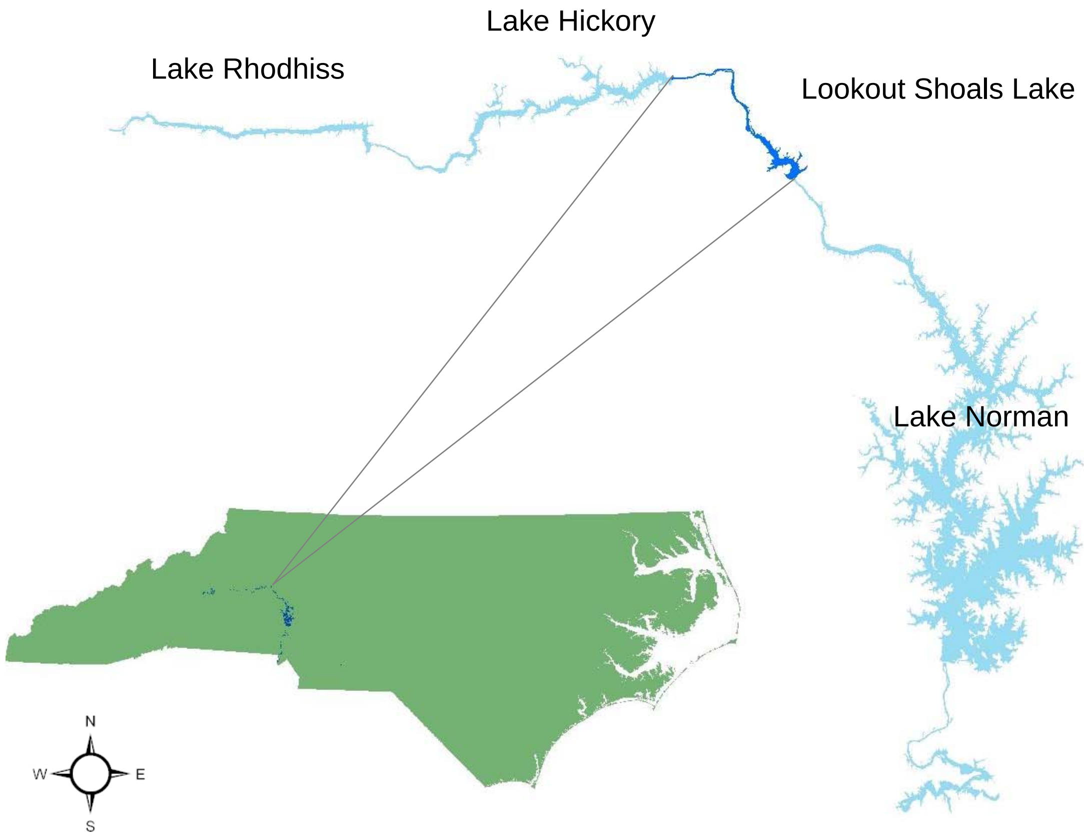

Figure 1. Location of study site, Lookout Shoals Lake, in relation to other Catawba River reservoirs within North Carolina.

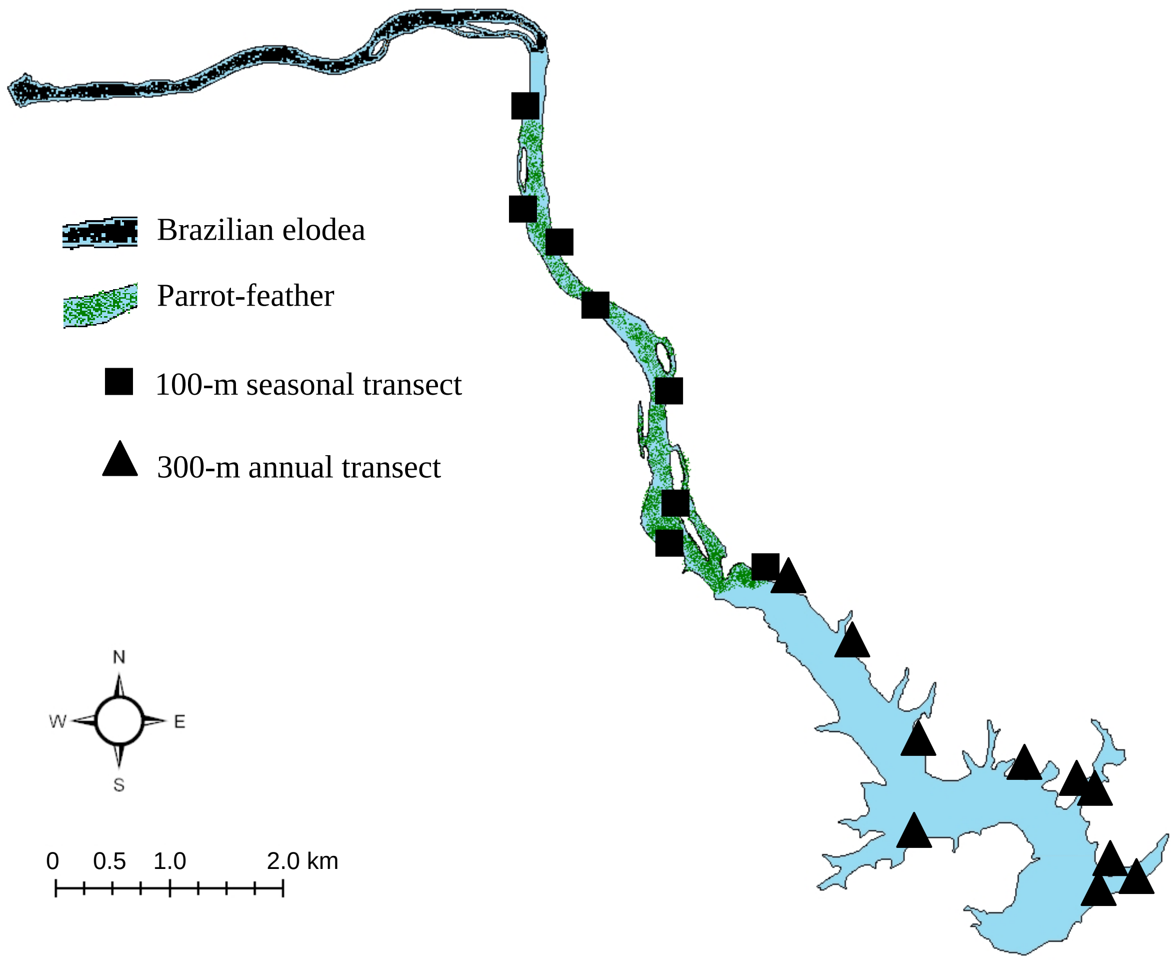

Figure 2. Parrot-feather and Brazilian elodea estimated coverage, locations of 100-m shoreline electrofishing transects sampled seasonally beginning summer of 2004 and ending spring of 2007, and 300-m shoreline electrofishing transects sampled annually during spring of 1994-1997, 2000, and 2005-2007 on Lookout Shoals Lake, North Carolina.

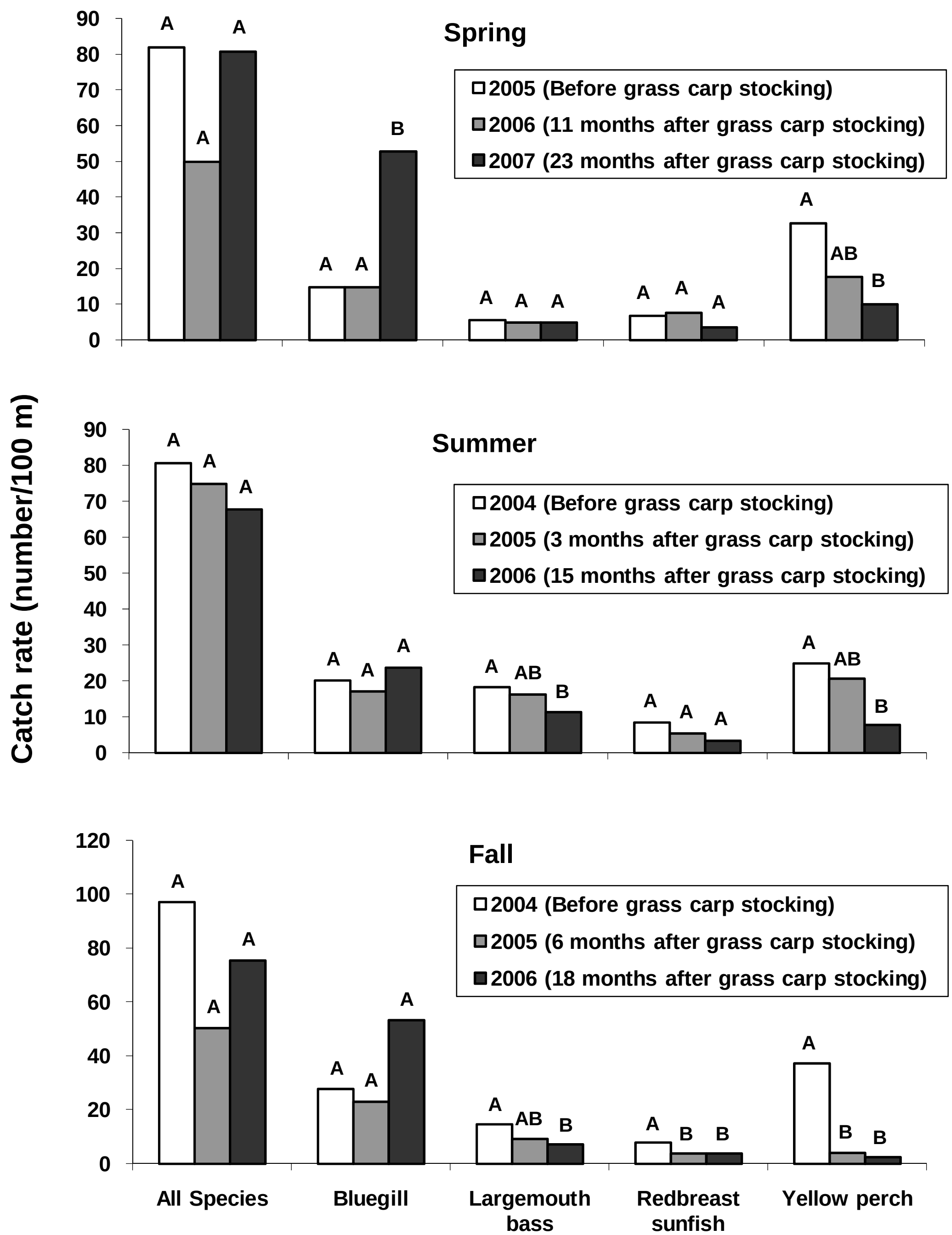

Figure 3. Catch rate by number (number/100 m) of all species in aggregate, bluegill, largemouth bass, redbreast sunfish, and yellow perch collected from eight, 100-m seasonal shoreline electrofishing transects during spring, summer and fall on Lookout Shoals Lake, 2004-2007. Common letters indicate catch rates that are not significantly different within all or each species.

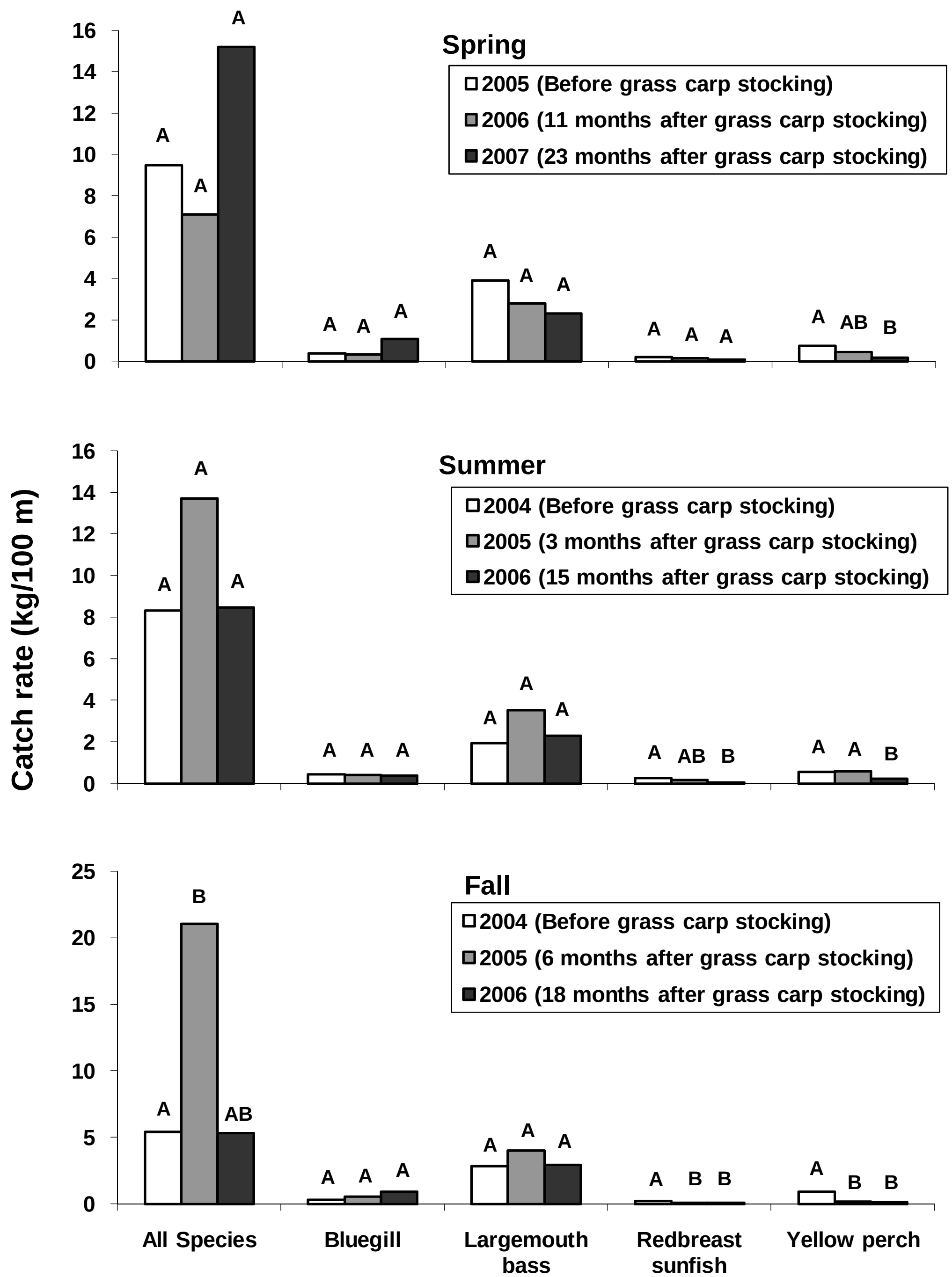

Figure 4. Biomass (kg/100 m) of all species in aggregate, bluegill, largemouth bass, redbreast sunfish, and yellow perch collected from eight, 100-m seasonal shoreline electrofishing transects during spring, summer and fall on Lookout Shoals Lake, 2004-2007. Common letters indicate biomass levels that are not significantly different within all or each species.

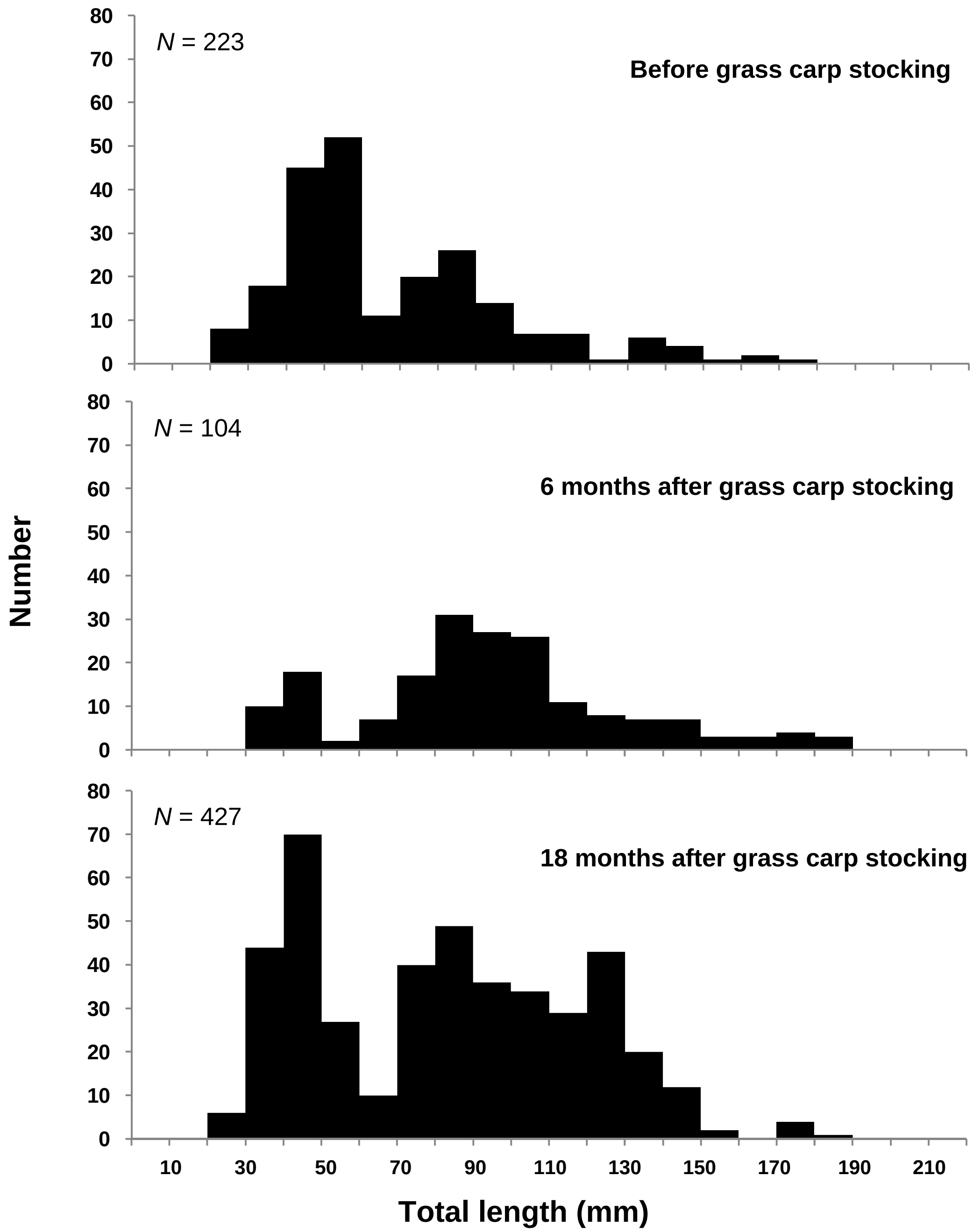

Figure 5. Length-frequency distributions of bluegill collected from eight, 100-m seasonal shoreline electrofishing transects during fall on Lookout Shoals Lake 2004-2006.

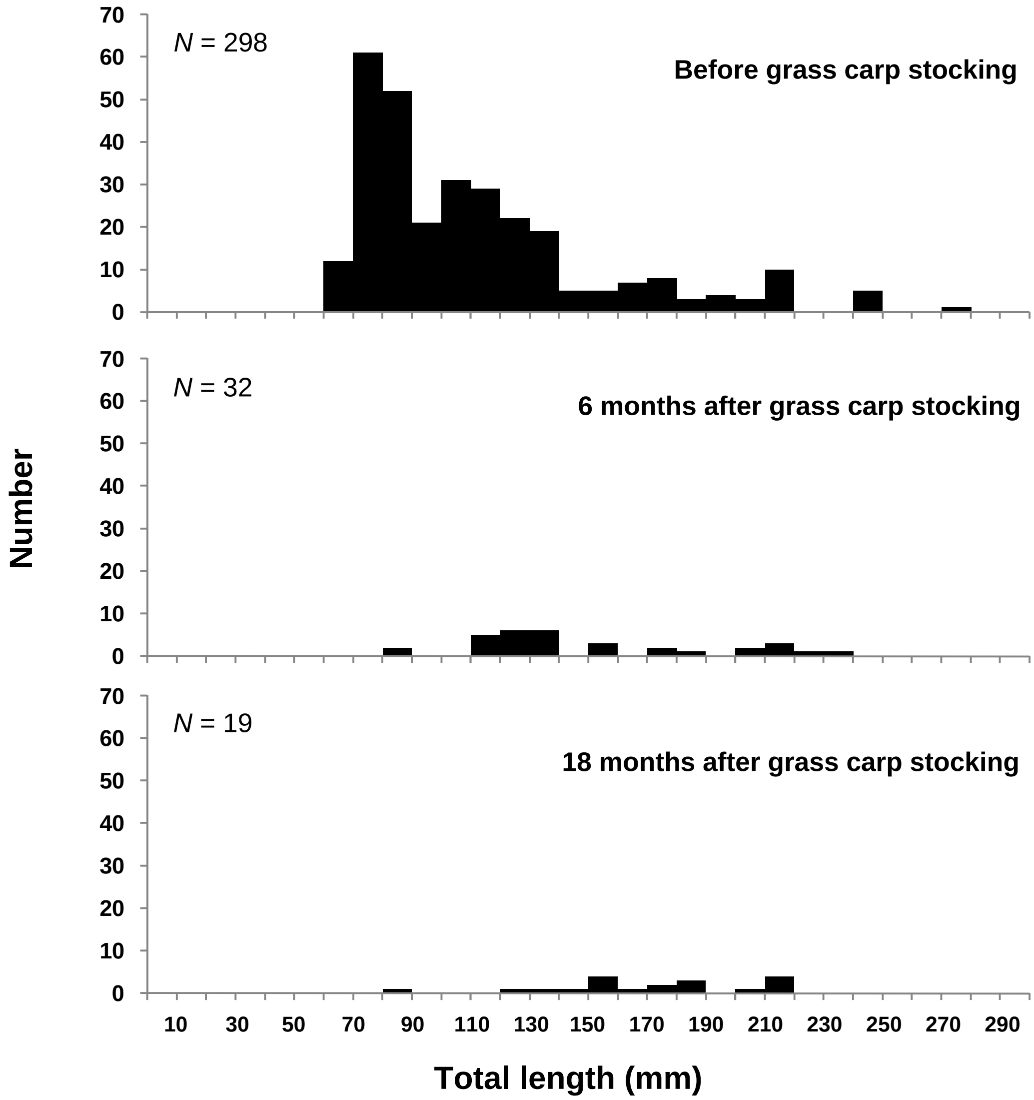

Figure 6. Length-frequency distributions of yellow perch collected from eight, 100-m seasonal shoreline electrofishing transects during fall on Lookout Shoals Lake 2004-2006.

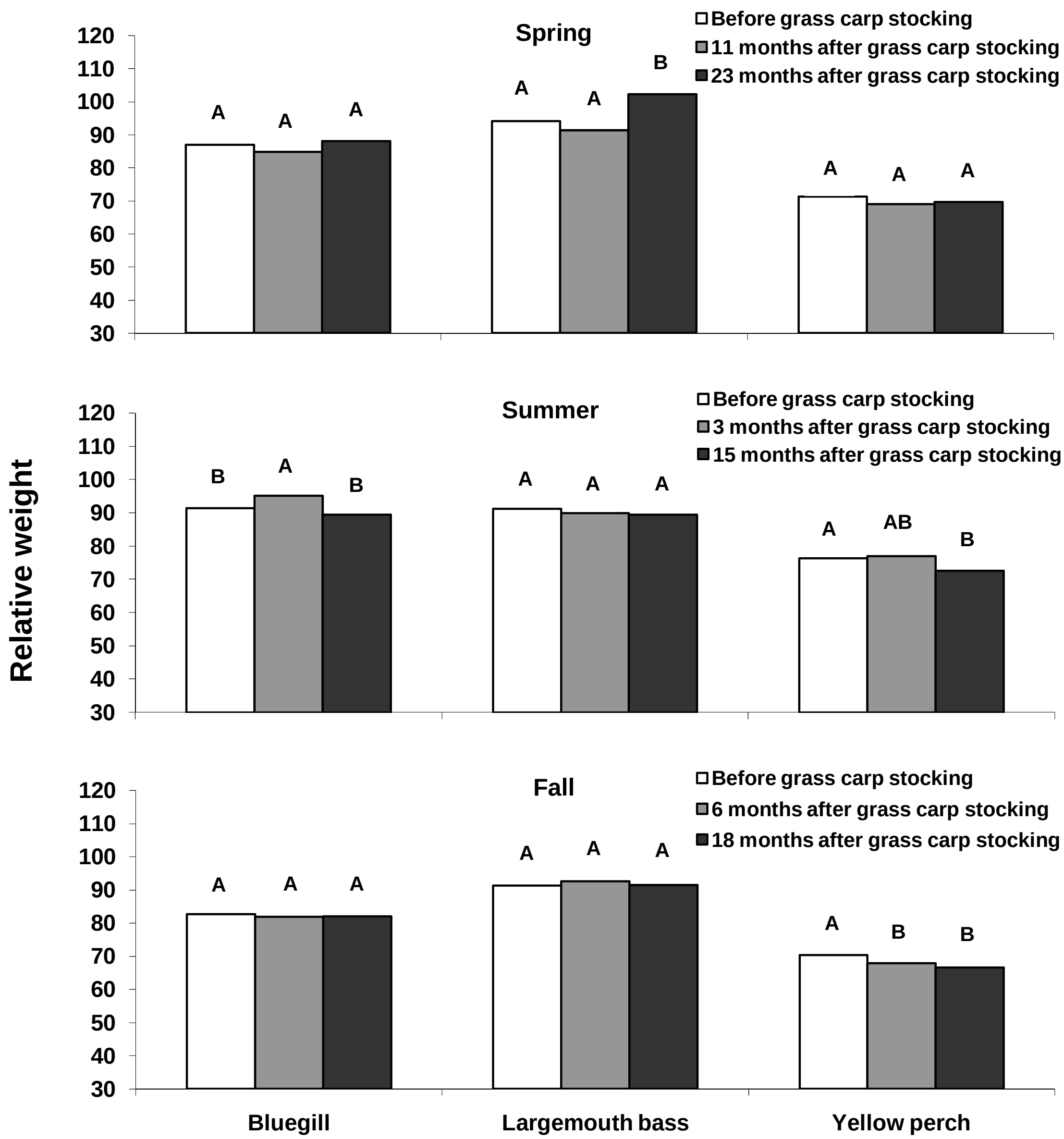

Figure 7. Relative weight of bluegill, largemouth bass and yellow perch collected from eight, 100-m seasonal shoreline electrofishing transects during spring, summer and fall on Lookout Shoals Lake 2004-2007. Common letters indicate relative weight values that are not significantly different within each species.

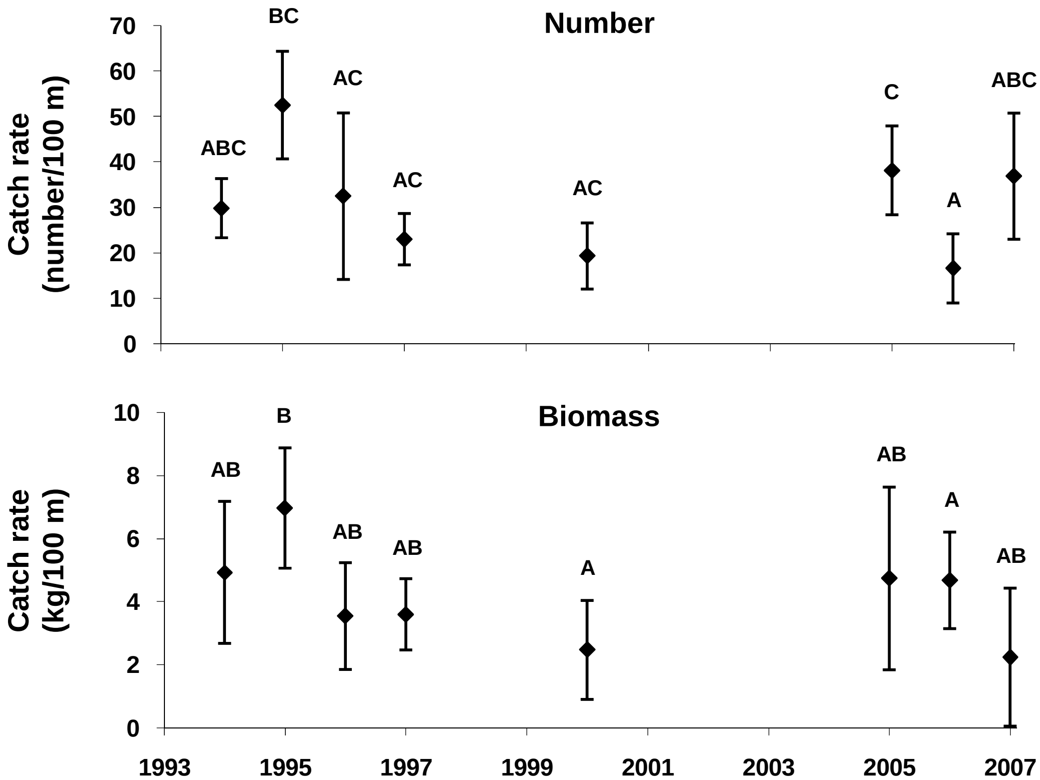

Figure 8. Catch rate by number (number/100-m, above) and biomass (kg/100-m, below) of all species collected from ten, 300-m annual shoreline electrofishing transects during spring on Lookout Shoals Lake, 1994-2007. Common letters indicate rates that are not significantly different.

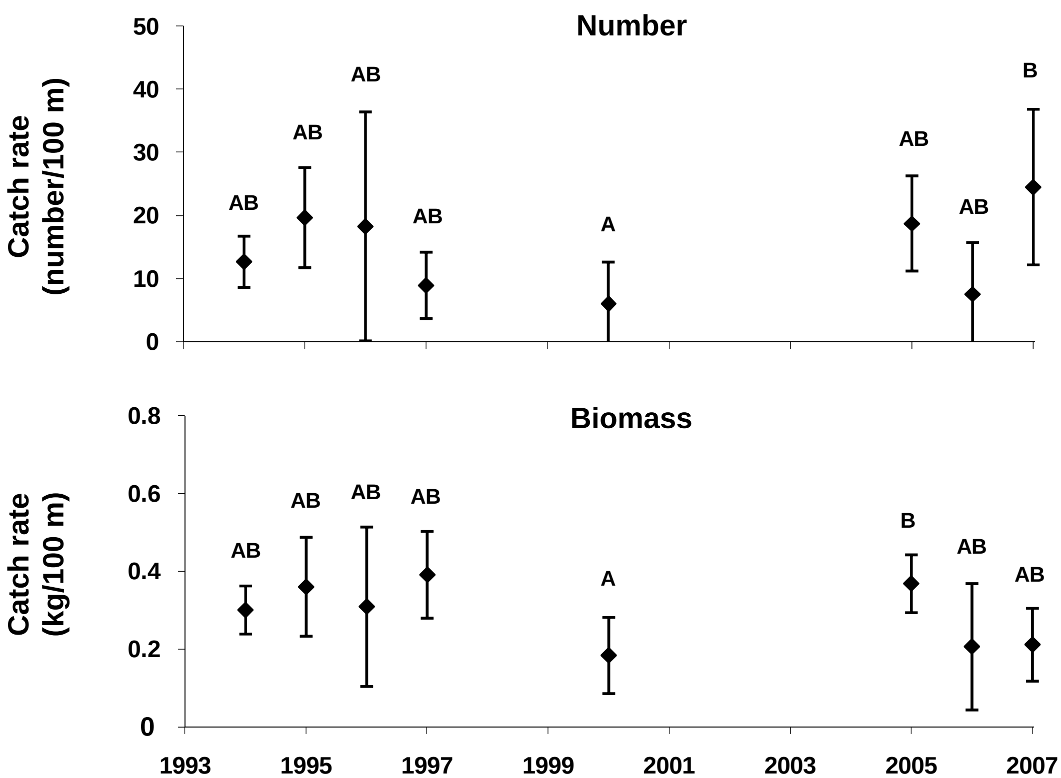

Figure 9. Catch rate by number (number/100-m, above) and biomass (kg/100-m, below) of bluegill collected from ten, 300-m annual shoreline electrofishing transects during spring on Lookout Shoals Lake, 1994-2007. Common letters indicate rates that are not significantly different.

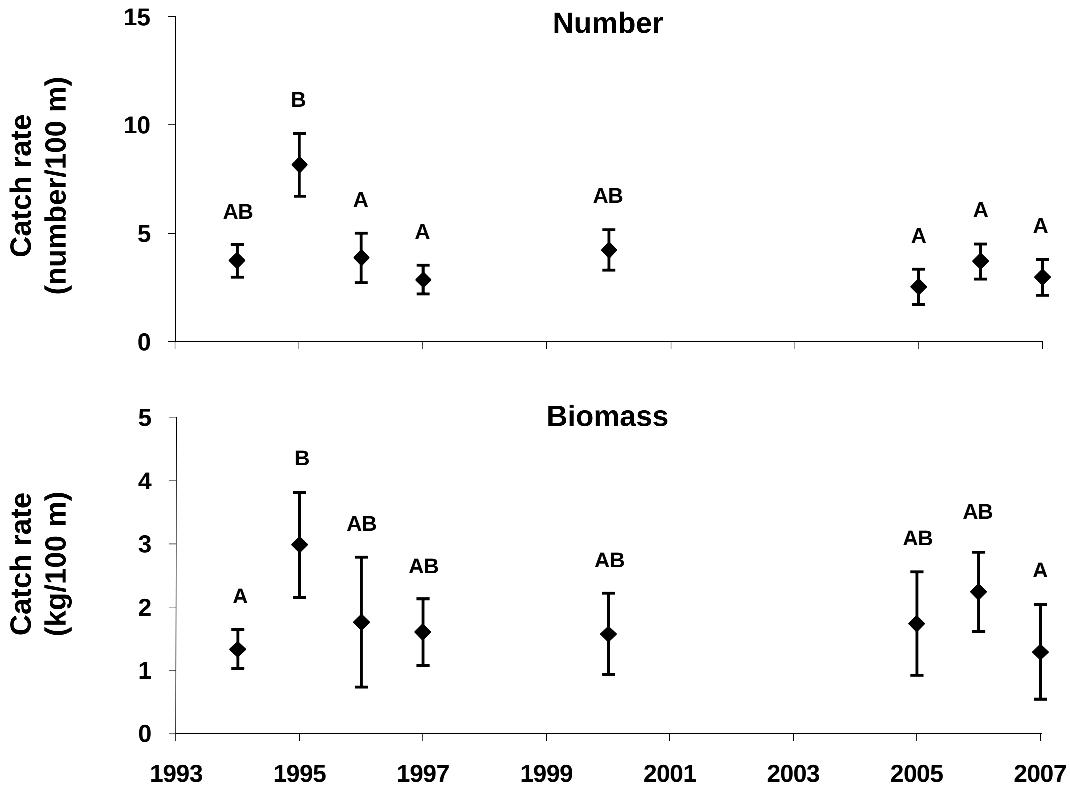

Figure 10. Catch rate by number (number/100-m, above) and biomass (kg/100-m, below) of largemouth bass collected from ten, 300-m shoreline electrofishing transects during spring on Lookout Shoals Lake, 1994-2007. Common letters indicate rates that are not significantly different.

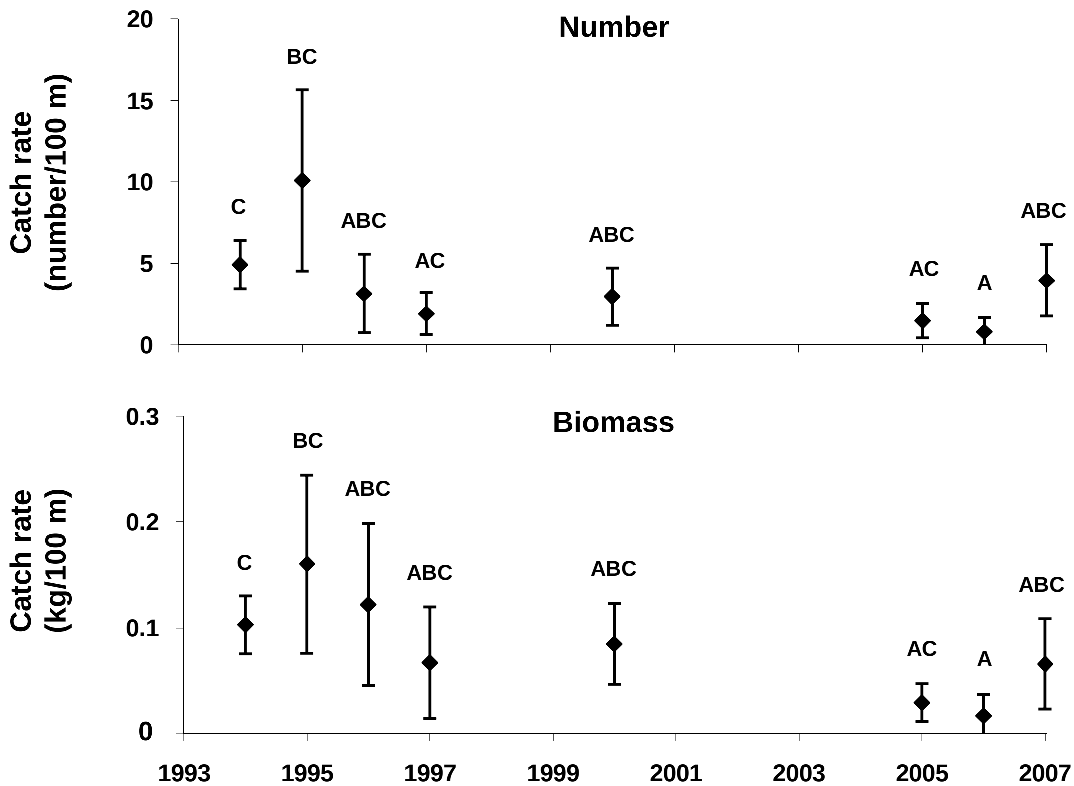

Figure 11. Catch rate by number (number/100-m, above) and biomass (kg/100-m, below) of redbreast sunfish collected from ten, 300-m shoreline electrofishing transects during spring on Lookout Shoals Lake, 1994-2007. Common letters indicate rates that are not significantly different.

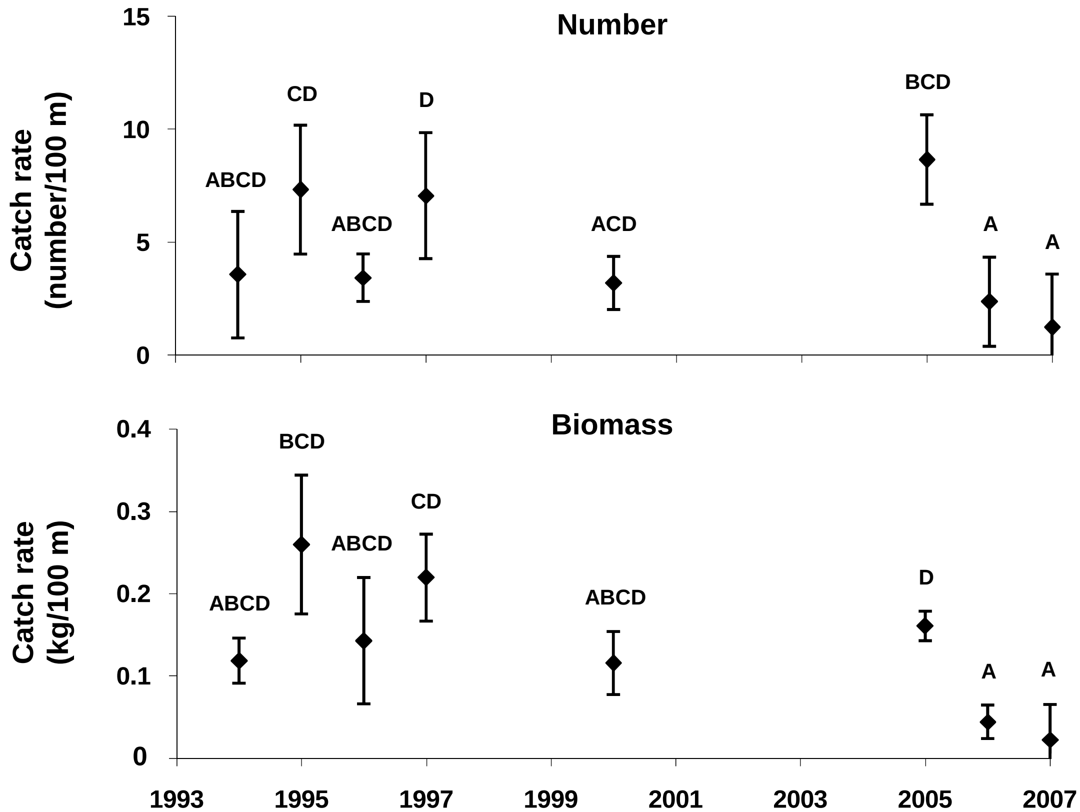

Figure 12. Catch rate by number (number/100-m, above) and biomass (kg/100-m, below) of yellow perch bass collected from ten, 300-m shoreline electrofishing transects during spring on Lookout Shoals Lake, 1994-2007. Common letters indicate rates that are not significantly different.