OLIGOTROPHICATION EFFECTS FOLLOWING DIVERSION OF WASTE

EFFLUENT FROM AN EMBAYMENT OF LAKE MARTIN, ALABAMA

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OLIGOTROPHICATION EFFECTS FOLLOWING DIVERSION OF WASTE EFFLUENT FROM AN EMBAYMENT OF LAKE MARTIN, ALABAMA

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VITA

Richard Jason Dickey, son of Wayne and Linda (Pappas) Dickey, was born Friday, January 13th, 1978 in Oak Ridge, Tennessee. He graduated from Oak Ridge High School in 1996. In 2002, he received his Bachelors of Science in Wildlife and Fisheries Science from the University of Tennessee, Knoxville. During his undergraduate career, he gained internship experience with the Environmental Sciences Department at Oak Ridge National Laboratories and Cadmus Group, Inc. After graduation he spent one year working with the U.S. Fish and Wildlife Service in Panama City, Florida as a volunteer with Americorps/ Student Conservation Association. In August, 2003, he entered the Graduate School at Auburn University.

THESIS ABSTRACT

OLIGOTROPHICATION EFFECTS FOLLOWING DIVERSION OF WASTE EFFLUENT FROM AN EMBAYMENT OF LAKE MARTIN, ALABAMA

Richard Jason Dickey

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I examined the effects of nutrient diversion on water quality, phytoplankton, and macroinvertebrate and fish community structures over a 10-year period in Lake Martin, Alabama. In 2001, a pipeline began diverting treated effluent from Alexander City, Alabama out of Elkahatchee Creek Embayment (ECE) and into the mainstem of Lake Martin reservoir. Oligotrophication in ECE produced measurable responses within the biotic community, including: 1) decreased algal biomass; 2) a shifting black bass community; 3) decreased catch and condition of sunfish and crappie; and 4) altered structure and function of littoral macroinvertebrate communities. Elevated conductivity at the mouth of Dennis Creek Embayment (DCE), the reference site, suggested the encroachment of diverted effluent. No changes conclusive to nutrient enrichment were found with respect to: key nutrients, algal biomass, or macroinvertebrate assemblages.

However, some *Centrarchid* fish responded in a manner consistent with eutrophication. Members of *Lepomis* expressed higher condition following diversion. In addition, the black bass dynamic shifted to favor longer largemouth bass (*Micropterus salmoides*) with higher relative weights, while spotted bass (*Micropterus punctulatus*) were fewer, shorter, and less robust.

The aquatic communities in this study provided a measurable response to subtle changes in trophic status which underscores the sensitivity of the aquatic community. The ability to detect and predict these subtle changes enables reservoir managers to make more confident predictions and ultimately lead to improved design and management of reservoirs.

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I. INTRODUCTION

The Federal Water Pollution Control Act of 1972 addressed mounting concern in America over the widespread degradation of rivers, lakes, and estuaries due to cultural eutrophication. Decades of unabated nutrient enrichment had resulted in undrinkable, foul smelling water prone to summer fish kills and little commercial or recreational value (Carpenter et al. 1998). The obvious solution was to limit the inflow of key nutrients into these systems, namely phosphorous and nitrogen. This movement, termed cultural oligotrophication, aided in the recovery of nuisance conditions by limiting undesirable algal growth and greatly increasing water clarity (Ney 1996). However, the effect was bimodal. Fewer nutrients translated into lower productivity, particularly in reservoirs. By the mid 1990's, it had become apparent that aesthetically clearer water and fisheries productivity were not complementary (Schupp and Wilson 1993).

Establishing a healthy balance between clearer water and fisheries productivity has been complicated by the range of factors influencing trophic dynamics across the continent; these include, hydrology, nutrient input, regional geomorphology and climate (Ney, 1989; Axler *et al*. 1988.). As stewards, our challenge is to develop strategies for maintaining healthy and sustainable water resources. A sound understanding of reservoir nutrient dynamics is paramount for achieving this goal, both at the worldwide and regional scale.

In 1994, Auburn University conducted a limnological study of the Elkahatchee Creek embayment (ECE) on Lake Martin reservoir near Alexander City, Alabama. This study continued through 1997 and 1998. During this time, 8.5 million gallons/day (permitted discharge) of both treated domestic sewage and industrial (primarily textile) effluent were being discharged into Sugar Creek, a tributary of ECE, from the Alexander City Wastewater Treatment Plant (ACWTP). The purpose of the study that began in 1994 was to address rising concern over water quality and environmental condition in ECE. A nearby tributary, Dennis Creek Embayment (DCE), located directly across Lake Martin from ECE, was chosen as a reference site due its proximity, similarity in size and the lack of point source discharge. The objectives of the study were to: 1) determine water quality conditions in ECE and DCE; (2) compare water quality conditions of ECE and DCE with conditions existing in the mainstem of Lake Martin and; (3) document aquatic community (phytoplankton, benthos, and fish) responses to the prevailing environmental conditions during the growing season (Bayne et al. 1995).

In June 2001, a pipeline constructed for the ACWTP began diverting effluent out of ECE and into the thalweg on the mainstem of Lake Martin. The purpose of this thesis is to assess the effects of nutrient removal in ECE on water quality, phytoplankton biomass, and fish and benthic invertebrate communities by comparing results obtained in 2002 through 2004 with those reported in 1994, 1997, and 1998. Specifically, I propose to describe: 1) water quality conditions in ECE and DCE prior to and after nutrient diversion; 2) the effects of nutrient availability on phytoplankton biomass; 3) trends in the size, condition, and abundance of fish species, and; 4) changes in the density and diversity of benthic invertebrates.

II. LITERATURE REVIEW

Political and public perceptions of water quality often focus on water clarity with less emphasis on aquatic communities. As a result, water quality initiatives target nutrient reduction without considering the numerous impacts on reservoir biota. Recent studies have linked nutrient reductions with changes in: zooplankton and algal biomass, fish size and growth; and macroinvertebrate richness and diversity (Chambers et al. 2006; McCormick et al. 2004; Maceina and Bayne 2001; Allen et al. 1999, Hoxmeier and DeVries 1998, DiCenzo et al. 1995, Hendricks et al. 1995, Bayne et al. 1994; Blancher 1984).

Eutrophication can describe either the process of nutrient addition or the effects of added nutrients in the aquatic environment (Larkin and Northcote 1969). Cultural eutrophication describes changes resulting from anthropogenic influences. Increased levels of mainly phosphorous and nitrogen, stemming from agricultural, urban, domestic, and industrial effluent can bolster trophic status, sometimes to undesirable levels. Adverse effects of eutrophication in freshwater systems can range from benignly unpleasant tastes or odors to more serious issues such as water treatment inefficiencies, oxygen depletion, and fish kills (Carpenter et al. 1998).

Opposite of cultural eutrophication is the anthropogenic abatement and/or removal of nutrients, coined “cultural oligotrophication” (Ney 1996). Nutrient reductions most commonly occur from construction of upstream impoundments or point-source abatement programs, such as advanced wastewater treatment facilities or sewage diffusion (Bayne et al. 2003, Axler et al. 1988; Gloss et al. 1980). By decreasing nutrient loads, these efforts limit phytoplankton growth and, in turn, limit the primary productivity in the system. Expected outcomes of oligotrophication should include not only clearer water, but decreases in fish biomass and angler satisfaction as well (Yurk and Ney 1989).

Phosphorous (P) and nitrogen (N) are often the primary nutrients influencing productivity in freshwater ecosystems. Not all forms of P are readily available to plants and those that are get quickly utilized or bound in lake sediments. Moreover, sediment-bound P does not share the internal recycling efficiency of N (Boyd 2000). Therefore, P is often the least abundant and most limiting nutrient for productivity (Wetzel 1983). An assessment of 19 Alabama reservoirs determined that total phosphorous (TP) explained 49% of the variation in algal biomass (Maceina et al. 1996). The same study concluded that TP was nearly four times as influential as total nitrogen in predicting algal biomass. Canfield and Bachman (1981) reported total phosphorous (TP) to have accounted for 57% of the variation in chlorophyll *a* among 433 artificial lakes spread across the United States. Miller et al. (1974) found that P was limiting in 35 of 49 lakes with N limiting in the remaining eight.

Nitrogen limitation is usually predominant in marine systems but can also exist in eutrophic freshwater systems where P is abundant (Welch 1980). The ratio of total nitrogen to total phosphorous (TN:TP) has been a useful index comparing the relative influence of each nutrient across the trophic spectrum. For example, Welch (1980) reported that oligotrophic lakes had a TN:TP ≥ 16/1, indicating P limitation. Conversely, eutrophic lakes had a TN:TP < 16/1, indicating N limitation. Smith (1986) used this index to explain the relative dominance of nitrogen fixing blue-green algae to other phytoplankton types. Downing and McCauley (1992) proposed that the TN:TP ratio reflected the source of nutrient input into a system.

Nutrient cycling plays a critical role in the recovery time following nutrient abatement. The nitrogen budget in freshwater systems can be maintained at equilibrium by a variety of natural processes, i.e. atmospheric contribution, nitrogen fixation, mineralization, nitrification, and denitrification (Boyd 2000). Following a major nutrient reduction, this equilibrium can be reached in 0–5 years (Anderson et al. 2005).

Phosphorous however, lacks the recycling efficiency of N. Periods of anoxia or low dissolved oxygen trigger the release of P by lowering the oxidation-reduction potential at the sediment-water interface. Ultimately, these P inputs are either quickly utilized by plants or re-adsorbed and trapped again in lake sediments (Boyd 2000). Søndergaard et al. (1999) detected P release at sediment depths down to 25 cm among 265 shallow Danish lakes. Internal P loading can delay improvements in water quality for 10–25 years following nutrient abatement and is greatly influenced by initial loading rate, flushing schedule, depth, and the chemical characteristics of the sediment (Anderson et al. 2005; Jeppesen et al. 2005, Søndergaard et al 2003; Søndergaard et al. 1999; Van Der Molen et al. 1998). Edmondson and Lehman (1981) reported that P loading accounted for about 57% of the total P income during a 27-year period in Lake Washington, WA.

The relationship between productivity and the nutrient budget is of central importance to limnology. Classical theory asserts a trophic hierarchy whereby energy inputs are first expressed by producers and then flow upward along the foodchain (Elton 1927). Lindemann (1947) further theorized that the efficiency of energy flow followed a predictable pattern throughout natural lake succession; i.e. productivity increased from oligotrophy to eutrophy before eventually declining with lake senescence. Carlson (1977) later proposed the trophic state index (TSI) to quantitatively classify trophic status based on Secchi visibility, chlorophyll *a*, or TP measurements.

The effects of nutrient enrichment on plankton communities have been well documented. Generally, nutrient enrichment leads to: 1) an increasing dominance of Chlorophyta (green algae) and especially Cyanophyta (blue-green algae), and; 2) greater density and lower diversity of algal flora (Bold and Wynne 1985; Edmondson and Lehman 1981; Welch 1980; Schindler et al. 1973; Edmondson 1969). Bayne et al. (1990) recorded a shift in the phytoplankton community from a diatomaceous to a green and blue-green dominance during a 10-year period in West Point Lake, GA. In turn, zooplankton communities reflect this shift from more to less desirable algal species (Porter 1977). General community trends have shown that: 1) zooplankton abundance and biomass increase with trophy; 2) mean length of zooplankton declines with trophy, and; 3) dominance shifts from macrozooplankton to microzooplankton as systems become more productive (Harman et al. 1995; Bayne et al. 1992; Pace 1986; Blancher 1984; Canfield and Watkins 1984; Bays and Crisman 1982; Pace and Orcutt 1981).

Presence and distribution of benthic macroinvertebrates along the trophic gradient are regulated by the range of physical and chemical characteristics of each waterbody (Merritt and Cummins 1996). General trends of increased productivity, driven by increased organic enrichment, include: 1) increased standing stock overall; 2) increased densities of Chironomidae, Oligochaeta, Chaoboridae, Gastropoda, and Sphaeriidae; 3) fewer *Hexagenia* spp., and; 4) a greater composition of Oligochaeta to Chironomidae (Popp and Hoagland 1995; Welch 1980; Dermott et al. 1977; Peterka 1972; Jónasson 1969; Carr and Hiltunen 1965).

To date, several researchers have demonstrated a direct relationship between productivity and fish species richness, growth, biomass, and recruitment. McKenna (1995) associated greater diversity with more eutrophic conditions among three New York Finger Lakes. Phosphorous reductions in Lake Erie (1969–1996) led to both: 1) reduced species richness in the shallower and clearer West Basin and; 2) increased species richness in the deeper and more turbid Central Basin (Ludsin et al. 2001). A study of 65 Florida lakes, ranging from oligotrophic to hypereutrophic, found no correlation between species richness and trophic status (Bachman et al. 1996).

In 27 Alabama reservoirs, Hendricks et al (1995) found that increased trophic state corresponded to greater average weights of angler-caught black bass (*Micropterus spp*.). A later study of 32 Alabama reservoirs detected faster growth (mm TL) among largemouth bass and spotted bass as reservoir trophy increased (chlorophyll *a* ranged from 2 mg/m^3 to 27 mg/m^3) (Maceina et al. 1996). Allen et al. (1999) documented a positive linear relationship between trophic status and age-0 largemouth bass among nine Alabama reservoirs (chlorophyll *a* ranged from 1 mg/m^3 to 27 mg/m^3).

Crappie (*Pomoxis spp.)* have also been reported to exhibit faster growth (mm TL) at higher trophic levels. A comparison of three Alabama reservoirs, ranging in trophic status, documented faster crappie growth in the eutrophic reservoir than in the oligomesotrophic (Bayne et al. 1994). This trend was later confirmed in a study of 32 Alabama reservoirs; although, crappie growth did not increase when chlorophyll *a* levels exceeded 8 mg/m^3, the lower threshold for being considered eutrophic (Maceina et al. 1996).

These trends are not exclusive to piscivores. In 10 Alabama reservoirs, DiCenzo et al. (1996) found that gizzard shad (*Dorosoma cepedianum*) were more abundant in eutrophic systems; whereas, growth rates and condition were higher in the oligomesotrophic systems. These results confirmed an earlier study of four Alabama reservoirs where gizzard shad abundances increased with trophic status while average total length decreased (Bayne et al. 1994). This trend was also confirmed by Clayton and Maceina (2002) where gizzard shad exhibited faster growth and greater body condition but lower abundances in a reservoir with a lower trophic status. Thus for gizzard shad, increased abundances at higher trophic states may initiate density-dependent regulation which is less pronounced in more oligotrophic systems (Clayton and Maceina 2002). Moreover, with higher abundances, slower growth and smaller sizes gizzard shad make a more ideal forage fish in eutrophic, rather than oligotrophic systems (DiCenzo et al. 1996).

Fewer studies have addressed the direct impact of oligotrophication. Nutrient abatement programs in Lake Ontario reduced phosphorous by 25% which coincided with a 50% decrease in forage fish biomass, as well as decreased weights and poorer conditions of stocked salmonids (Great Lakes Fishery Commission 1992). Likewise, a 50% reduction in phosphorous load into Beaver Lake, Arkansas corresponded to a 55% decline in planktivorous fish and a 9% decline in piscivirous fish standing stocks (Ney 1996). During a 12 year study of Smith Mountain Lake, Virginia, Ney (1996) found that as TP declined 79%, annual growth rates of black bass and striped bass (*Morone saxatilis*) also declined by 12%.

Maceina and Bayne (2001) reported that nutrient reductions in West Point Reservoir, Alabama/Georgia resulted in fewer and smaller largemouth bass and smaller, yet more abundant spotted bass (chlorophyll *a* values declined from 40 µg/l to 9–17 µg/l). Greene and Maceina (2000) reported greater densities of age-0 largemouth bass than for spotted bass in eutrophic reservoirs, but greater densities and biomass of the latter in meso- and oligotrophic systems. Buynak et al. (1989) recorded greater catch of largemouth bass in the more eutrophic portions of Cave Run Lake, Kentucky and greater catch of spotted bass in the less productive portions.

Reservoir managers make decisions that affect a variety of stakeholders and multi-recreational users. A sound understanding of the nutrient-fishery dynamic is essential for striking the balance between achieving aesthetically clearer water and maximizing fish production. This thesis enhances this understanding by examining nutrient availability, phytoplankton biomass, fisheries productivity, and macroinvertebrate density and diversity as they directly pertain to oligotrophication in a southeastern reservoir.

III. MATERIALS AND METHODS

Study Sites

Lake Martin (Figure 1), a 15,850 ha reservoir was created on the Tallapoosa River system in 1926 to provide power generation, flood control, and recreation for east-central Alabama. The lake has a mean hydraulic retention time of 190 days and has been classified as low-ranged mesotrophic (TSI = 39) by calculating the trophic state index (TSI) based on chlorophyll *a* concentration (Bayne 1995).

Elkahatchee Creek Embayment (ECE) is located south of Alexander City (Figure 2). Until 2001, ECE had received treated municipal waste from Alexander City from a smaller tributary, Sugar Creek. The shoreline is predominantly forested with some shoreline residential development. Station E1 is located at mid-embayment downstream of the Sugar Creek confluence. Station E2 is located at the mouth near the mainstem of Lake Martin (Figure 2).

Dennis Creek Embayment (DCE) is located directly across the reservoir (Figure 2) from ECE. Dennis Creek receives no known point-source discharge though does contain some shoreline residential development. Station D1 is located in the upstream portion of DCE and Station D2 is at the mouth near the mainstem of Lake Martin (Figure 2).

Water Quality Sampling

Temperature and dissolved oxygen, photic zone, Secchi visibility and specific conductance were measured *in situ* in the water column at each station (Table 1). Water samples were collected using a submersible pump and hose apparatus. Composite samples were pumped from throughout the photic zone and stored in coolers on ice in 2-L Nalgene bottles until arrival at the laboratory for analysis. The following water quality variables were measured in the laboratory: pH, total alkalinity, total hardness, total phosphorous, soluble reactive phosphorous, total nitrogen, nitrate-nitrogen, nitrite-nitrogen, total ammonia nitrogen, turbidity, and total suspended solids (Table 1). Standard analytical methods were followed for all variables and holding times were well within recommended limits (APHA et al. 1998).

Water samples were collected monthly during the growing season (April–October) during 1994 and 2004 (Table 2). Only May and June were sampled during 1997. In 1998, June, July, and August were sampled. During 2002, samples were collected monthly from June through August. In 2003, the growing season was represented by April, June, and August. Growing season values were assessed for each water quality variable, divided into either a pre (1994–1998) or post (2002–2004) treatment category, and compared among sites and through time using a split-plot repeated measures analysis of variance (RMANOVA) (Maceina *et al.* 1994). In some cases the models indicated a significant station/treatment interaction ($P < 0.05$) signifying that the upstream and mouth stations had responded differently to nutrient diversion over time. For these instances *t*-tests were used to address the effects at each station separately.

Phytoplankton Sampling

Composite phytoplankton samples were collected throughout the photic zone as described above (Table 2). Chlorophyll *a* concentrations (corrected for phaeophyton) were measured in the laboratory according to protocol of the American Public Health Association (APHA 1998). Algal biomass was estimated from chlorophyll *a* concentrations and comparisons were made between sites and throughout time using RMANOVA. Chlorophyll *a* data were used to generate trophic scores using the trophic state index (TSI) (Carlson, 1977). Trophic scores were compared between sites and throughout time using RMANOVA. Stepwise regression and correlation analyses identified the variables that were significantly ($P < 0.05$) related with chlorophyll *a* concentrations. In some instances, $\log_{10}$ transformations were calculated to homogenize variances and linearize relationships among variables. Simple and multiple linear regressions were used to describe the nature and magnitude of these relationships. Partial squared correlation coefficients (pr^2) addressed the effects of each variable independently, relative to the effects of all others. Multicollinearity was considered problematic when condition index values exceeded 3.00 and when variance inflation factors (VIF) exceeded $1/(1\text{-}R^2)$.

Fish Sampling

Fish were collected twice from the upstream sites in ECE (E1) and DCE (D1). Fish were sampled during July and October 1994, 2002, and 2004 (Table 2). Only July was sampled during 2003. Fish were collected diurnally via boat electrofishing and sinking gill nets. Each site received 60 minutes pedal time of pulsed DC current using a Smith Root® boat electrofisher. Pedal time represented direct fishing effort distributed equally among available habitat types, excluding travel time. Transects were not performed. Sampling also involved a 4 hr soak period using three sinking gill nets (two 1.5 inch and one 2.5 inch bar mesh) checked at 2 hour intervals.

Fishes were kept in a livewell until transported to a central location where they were inspected for lesions/abnormalities, identified to species, and measured to the nearest. All fish were weighed to the nearest 0.1 gram with the exception of threadfin shad which were measured only to the nearest mm (TL). Total length was measured to the nearest mm. Fishes were released live when possible near the vicinity of capture. Unidentified species were preserved on ice and transported to the laboratory for identification.

Catch per effort (CPE) for electrofishing and gill netting effort was calculated as fish/hour and fish/net, respectively. Student's t-tests were used to compare CPE between treatments at each site. Relative weight (Wr) assessed fish condition for species having a standard equation (Anderson and Neumann 1996). Data were then pooled for sampling method and Student's t-tests were used to make comparisons between treatments at each site. For some species, a standard weight equation did not exist. Analysis of covariance (ANCOVA) was then used to detect elevation differences among $\log_{10}$ transformed weight-length regressions. Significant differences ($P < 0.05$) between y-intercepts signified that one population was heavier, or better conditioned, at a given length than the other. Condition was also assessed using ANCOVA by assessing differences in slope, rather than elevation, between treatments at each site. Finally, total length (TL) was compared between treatments at each site using Student's t-tests. Statistical analyses were considered significant at $\alpha = 0.05$ and were performed using SAS 9.1.3 (SAS Institute 2005).

Macroinvertebrate Sampling

Macroinvertebrates were collected during: summer (August) and fall (October) in 1994, summer (July) in 2002, and summer (August) and fall (November) in 2004 (Table 2). Summer samples were compared among all three years. Fall samples were compared only between 1994 and 2004. Macroinvertebrates were sampled using Hester-Dendy multiplate samplers and a petite ponar dredge. One Hester-Dendy sampler placed in DCE during October, 2004, was never recovered.

Hester-Dendy multiplate samplers (HD) consisted of nine masonite plates (7.6 cm x 7.6 cm) stacked and separated by 3 mm spacers. These plates imitated the natural reservoir substrate (e.g., rocks and woody debris) that reservoir invertebrates prefer while insuring a standardized colonization medium among stations. The total surface area (excluding edges) of each sampler was about 0.09 m^2. Three samplers were deployed in each embayment in the vicinity of stations E1 and D1. Each sampler was suspended above the substrate at depths of 1 to 3 m. This depth allowed adequate dissolved oxygen for supporting diverse communities. Samplers were given a 30-day colonization period, after which they were removed and transported on ice in water filled containers. In the laboratory, macroinvertebrates were: removed from the plates, sieved through a U.S.

Standard No.30 screen (0.6 mm mesh), preserved in 80% ethyl alcohol, and identified to the lowest practical taxon and counted. Densities were calculated as the mean number of organisms/m^2.

A petite ponar dredge (1.5 m x 1.5 m) was used to collect macroinvertebrates inhabiting the bottom sediment. Three dredge samples were taken from each embayment during July and October in the vicinity of stations E1 and D1. The samples were taken at depths greater than 3 m to ensure representation of anoxic sediments of the profundal zone, if present. Sieving (U.S. No. 30 sieve) and elutriation removed excess silt and debris from macroinvertebrate fauna. Samples were preserved in 80% ethyl alcohol, sorted, and identified to the lowest practical taxon. Densities were calculated as the mean number of organisms/m^2.

Community structure and function were assessed separately for HD and bottom dredge samples. Comparisons were made by season and between treatments using estimates of total density, species richness, diversity, evenness, and dominance by trophic level. Trophic level designations were classified based on feeding behaviors outlined by Merritt and Cummins (1996). Functional feeding groups (FFG) included: filter-collectors (FC), gather-collectors (GC), piercers (PI), predators (P), scrapers (S), and shredders (SH). Densities were compared using Student's t-tests and Analysis of Variance (ANOVA). Heterogeneity was assessed using the Shannon Diversity Index (H') and evenness (J') (Brower and Zar 1984). Student's t-tests assessed differences between treatments at each site. Variance was computed using the formula:

$$S^2 = \sum [\ f_i(\log_e f_i)^2 - [(\sum f_i(\log_e f_i)^2] / N] / N^2;$$

with degrees of freedom calculated as:

$$d.f. = (S^2{}_1 + S^2{}_2)^2 / [(S^2{}_1)^2 / N_1] + [(S^2{}_2)^2 / N_2].$$

Statistical analyses were considered significant at $\alpha = 0.05$ and were performed using SAS 9.1.3 (SAS Institute 2005).

IV. RESULTS

Water Quality

Soluble reactive phosphorous (SRP) declined ($t_{11.1}$ = -4.96; P < 0.0004) from a mean of 43 μg·L (± 8; ± *SE*) prior to nutrient diversion to 2 μg·L (± 0) following diversion at station E1, the mid-embayment portion of ECE (Figure 3). Station E2, located near the mouth of ECE, had a substantial, though less dramatic and statistically insignificant ($t_{11.1}$ = -1.94; P < 0.078), decline in mean SRP from 6 μg·L (± 3) during the pre to 1 μg·L (± 0) during the post (Figure 4). Soluble reactive phosphorous did not differ ($F_{1,1}$ = 0.01; P < 0.943) among treatment periods in DCE.

Total phosphorous (TP) also declined ($t_{11.2}$ = -7.59; P < 0.0001) at station E1 from a mean of 116 μg·L (± 12) during the pre to 27 μg·L (± 1) following nutrient diversion (Figure 5). At station E2, mean TP declined ($t_{12.6}$ = -5.72; P < 0.001) from 47 μg·L (± 4) during the pre to 21 μg·L (± 1) during the post (Figure 6) but did not differ ($F_{1,1}$ = 0.14; P < 0.769) between treatments in DCE. In ECE, the mean TN:TP ratio rose (t_{46} = 4.73; P < 0.0001) from 7.5 during the pre to 15.1 during the post. In DCE, this ratio remained similar ($t_{45.7}$ = 1.00; P < 0.325) between treatments and averaged 14.6 during the pre and 16.6 during the post.

Total alkalinity (TA) declined ($t_{13.9}$ = -5.26; P < 0.0001) at station E1 from a mean of 23 mg/L $CaCO_3$ (± 1) during the pre to 15 mg/L $CaCO_3$ (± 1) during the post

(Figure 7). At station E2, mean TA declined (t_{30} = -2.99; $P < 0.0056$) from 16 mg/L $CaCO_3$ (± 1) during the pre to 13 mg/L $CaCO_3$ (± 1) during the post (Figure 8) but did not differ ($F_{1,1}$ = 6.04; $P < 0.246$) between treatments in DCE.

Specific conductance declined at only the mid-embayment ECE station but increased at both DCE stations following nutrient diversion. At E1, mean conductance declined ($t_{11.4}$ = -8.37; $P < 0.0001$) from 119 µmhos/cm (± 8) to 52 µmhos/cm (± 1) following diversion (Figure 9). At station D1, mean conductance increased (t_{25} = 2.29; $P < 0.0309$) from 40 µmhos/cm (± 1) to 43 µmhos/cm (± 1) following diversion. At station D2, mean conductance increased ($t_{28.3}$ = 4.38; $P < 0.0001$) from 41 µmhos/cm (± 1.3) to 52 µmhos/cm (± 2.2) following diversion (Figure 10).

Mean pH decreased in ECE ($F_{1,1}$ = 164.10; $P < 0.0496$) from 7.39 (± 0.16) to 6.65 (± 0.07) following diversion but did not differ in DCE ($F_{1,1}$ = 17.74; $P < 0.1484$) (Figure 11). Hardness, total suspended solids, turbidity, compensation point, total nitrogen, nitrite-nitrogen, nitrate-nitrogen, ammonia-nitrogen, and Secchi disk visibility showed no significant ($P > 0.05$) changes following nutrient diversion in either embayment.

Phytoplankton

Chlorophyll *a* (CHLA) concentrations were higher ($F_{1,1}$ = 404.69; $P < 0.0316$) in ECE before effluent diversion and averaged 13 mg·m^3 (± 2; ± SE), as compared to the post average of 9 mg·m^3 (± 2) (Figure 12). Trophic state declined from pre (TSI = 54) to post (TSI = 51) but remained statistically similar ($F_{1,1}$ = 20.77; $P < 0.1375$) (Figure 13).

Prior to nutrient diversion in ECE, pH, total nitrogen, turbidity, total suspended solids, compensation point, and Secchi visibility were all significant ($P < 0.05$) correlates with CHLA (Table 3). Of these, total nitrogen (TN) was the best predictor of CHLA and explained 78% of the variability in the model:

$$\log_{10}\text{CHLA} = 0.189 + 5.22\ (\log_{10}\text{TN}).$$

Following nutrient diversion in ECE, TP, total suspended solids, compensation point, and Secchi visibility were all significant ($P < 0.05$) correlates with CHLA (Table 4). Of these, TP was the best predictor and explained 18% of the variability in the model:

$$\log_{10}\text{CHLA} = 0.860 + 6.32(\log_{10}\text{TP}).$$

In DCE, CHLA concentrations remained similar ($F_{1,1} = 0.42$; $P < 0.6336$) between treatments and averaged 5 mg·m^3 (± 1) before nutrient diversion compared with 5 mg·m^3 (± 1) after diversion (Figure 12). Trophic state rose slightly between pre (Mean TSI = 46) and post treatments (Mean TSI = 47), but remained statistically similar ($F_{15,41} = 0.79$; $P < 0.6845$) (Figure 13). Soluble reactive phosphorous, TIN, total suspended solids, and compensation point were all significant correlates with CHLA (Table 5). However, none of these variables were effective predictors of CHLA. This was most likely attributable to the low degree of variation in CHLA in DCE throughout the study period (Figure 12).

Fish

A total of 5,271 fishes weighing 759.7 kg representing 7 families and 25 species were collected at both sites from 1994 to 2004 (Appendix I and II). Dominant taxa included: *Micropterus punctulatus, M. salmoides, Lepomis macrochirus, L. microlophus, and Pomoxis nigromaculatus* (Centrarchidae); *Dorosoma cepedianum* and *D. petenense* (Clupeidae); *Moxostoma poecilurum* (Catostomidae); *Cyprinella venusta* and *Notropis* *texanus* (Cyprinidae); and *Morone chrysops* (Moronidae) which accounted for 92% of total catch and 70% of total biomass.

Black bass accounted for 11% of total catch and 22% of total biomass overall (gill net and electrofishing). The percentage of black bass in ECE represented by spotted bass (*Micropterus punctulatus*) increased from 40% to 60% (Figure 14) following nutrient diversion; whereas, spotted bass comprised the majority of black bass throughout the study in DCE (Figure 15).

In ECE, no changes in condition or CPE (gill net or electrofishing) of largemouth bass (*Micropterus salmoides*) were detected ($P > 0.49$). Relative weight decreased ($t_{130} = -4.08$; $P < 0.0001$) from a mean of 91.8 (± 8.1; ± SD) to 83.1 (± 8.6) following oligotrophication. Mean TL increased ($t_{36.5} = 3.88$; $P < 0.004$) from 220 mm (± 137; ± SD) prior to diversion to 318 mm (± 76) following diversion.

In DCE, largemouth bass expressed higher condition (ANCOVA; $t = 2.96$; $P < 0.0056$) during the post for fish collected in the fall (Figure 16), but not for summer caught fish ($P < 0.85$). Mean TL increased ($t_{66} = 2.79$; $P < 0.0069$) from 222 mm (± 103) during the pre to 297 mm (± 117) during the post. No differences ($P > 0.05$) in CPE or Wr were detected.

In ECE, mean CPE of spotted bass collected by electrofishing increased, though statistically similar ($t_5 = 2.51$; $P > 0.054$), from 7 fish/hr (± 2) during the pre to 24 fish/hr (± 2) during the post. No differences ($P > 0.05$) in spotted bass length or condition were detected. However, mean Wr declined ($t_{133} = -4.90$; $P < 0.0001$) from 96.8 (± 9.3) during the pre to 86.3 (± 8.9) during the post.

In DCE, mean CPE of spotted bass collected by gill nets declined (t_5 = -3.70; *P* < 0.014) following nutrient diversion from 18 fish/net during the pre to 4 fish/net during the post. In addition, mean Wr declined (t_{59} = -2.30; *P* < 0.025) from 91.5 ($\pm$ 5.9) during the pre to 87.0 ($\pm$ 8.5) during the post. Mean TL also declined (t_{181} = -3.62; *P* <0.0004) from 280 mm (± 109) during the pre to 212 mm (± 120) during the post.

Bluegill sunfish (*Lepomis machrochirus*) (79%) and redear sunfish (*Lepomis microlophus*) (11%) comprised the majority of sunfish collected. Less dominant species included green sunfish (*Lepomis cyanellus*) (5%), longear sunfish (*Lepomis megalotis*) (3%), warmouth sunfish (*Lepomis gulosus*) (1%), and redbreast sunfish (*Lepomis auritus*) (1%). Differences in CPE among treatments were not evident at either site for any sunfish species regardless of season or gear type, but low sample sizes likely hampered these analyses (*t* <1.61; df range: 1-5; *P* > 0.06).

In ECE, bluegill sunfish were expressed poorer condition (ANCOVA; *t* = -2.21; *P* < 0.028) following nutrient removal for fish collected during the fall (Figure 17), but not for summer caught fish (*P* > 0.90). Relative weight also declined (t_{817} = -14.37; *P* < 0.0001) from a mean of 87.9 (± 9.1) during the pre to 77.9 (± 9.6) during the post. Mean TL also declined (t_{645} = -9.72; *P* < 0.0001) from 121 mm (± 25) to 104 mm (± 31) following nutrient removal and the percentage of fish longer than 89 mm TL decreased from 95% to 67%, respectively.

In DCE, bluegill sunfish were better conditioned (ANCOVA; *t* = 3.92; *P* < 0.0001) following diversion (Figure 18). Mean Wr also increased (t_{284} = 6.00; *P* < 0.0001) from 78.6 (± 5.7) during the pre to 82.7 (± 8.7) during the post. For fish collected in the fall, mean TL increased (t_{442} = 2.22; $P < 0.027$) from 94 mm (± 32) during the pre to 102 mm (± 34) during the post. Redear sunfish expressed higher condition (ANCOVA; $t = 2.45$; $P < 0.0155$) in DCE during the post (Figure 19); however, no differences ($P >$ 0.05) in TL or Wr were detected. In ECE, only three individuals were recorded prior to nutrient diversion preventing statistical comparisons across treatments.

Black crappie (*Pomoxis nigromaculatus*) represented the majority of crappie with only one white crappie (*Pomoxis annularis*) sampled in DCE during fall of 2002. No differences ($P > 0.07$) in CPE or condition were detected at either site. In ECE, mean Wr declined (t_{170} = -3.69; $P < 0.0003$) from 85.4 (± 6.5) during the pre to 78.3 (± 6.9) during the post. Total length remained similar ($P > 0.40$). In DCE, mean TL decreased (t_{150} = -12.37; $P < 0.0001$) from 255 mm (± 23) during the pre to 189 mm (± 51) during the post. No differences ($P > 0.12$) in Wr were detected.

Gizzard shad (*Dorosoma cepedianum*) and threadfin shad (*Dorosoma petenense*) accounted for 18% of total catch and 5% of total biomass (gill net and electrofishing). Threadfin shad comprised 87% of shad abundance, but gizzard shad accounted for 95% of shad biomass. Gizzard shad Wr increased (t_{62} = 2.25; $P < 0.0283$) in DCE following nutrient diversion from a mean of 80.8 (± 13.6) to 87.4 (± 11.3), but remained similar ($P > 0.57$) in ECE between treatments. No differences ($P > 0.45$) in CPE (gill net or electrofishing), condition, or TL were detected among treatments at either site. This was likely caused by low sample sizes during the pre, particularly in ECE where only nine individuals were recorded.

In ECE, mean length of threadfin shad increased (t_{475}= 9.78; $P < 0.0001$) from 56 mm (± 6) during the pre to 63 mm (± 6) during the post. In DCE, mean length of threadfin shad decreased (t_{321}= -6.99; $P < 0.0001$) from 63 mm (± 8) during the pre to 55 mm (± 13) during the post. No differences ($P > 0.05$) in CPE (gill net or electrofishing), condition, or weight were detected.

Blacktail redhorse (*Moxostoma poecilurum*) comprised 90% of suckers with the remaining 10% comprised of spotted suckers (*Minytrema melanops*). Blacktail redhorse condition remained similar ($P > 0.18$) at both sites. Though in ECE, mean total weight decreased (ANCOVA; t = - 5.16; $P < 0.0001$) from 272.7 g during the pre to 251.7 g during the post (Figure 20). In addition, mean TL declined ($t_{45.3}$= -11.12; $P < 0.0001$) from 368 mm (± 27) during the pre to 293 mm (± 59) during the post.

In DCE, blacktail redhorse CPE (gill net) declined (t_4 = -5.58; P< 0.0051) from a mean of 17 fish/net (± 1) during the pre to 6 fish/net (± 1) during the post. Mean TL also declined (t_{248}= -6.80; $P < 0.0001$) from 341 mm (± 43) during the pre to 286 mm (± 51) during the post. Total weight remained similar ($P > 0.42$) between treatments.

Blacktail shiner (*Cyprinella venusta*) (52%) and weed shiner (*Notropis texanus*) (44%) comprised the majority of minnow abundance. Less dominant species included bullhead minnow (*Pimephales vigilax*) (3%) and pugnose minnow (*Opsopoeodus emiliae*) (1%). No differences ($P > 0.05$) in CPE (gill net or electrofishing), condition, or total weight were detected for these species.

In DCE, mean TL of blacktail shiners increased ($t_{48.6}$ = 5.87; $P < 0.0001$) from 59 mm (± 5) during the pre to 78 mm (± 23) during the post. In ECE conversely, mean TL decreased (t_{91} = -2.31; $P < 0.023$) from 89 mm (± 17) during the pre to 76 mm (± 19) during the post. The same trend was detected in weed shiners. Mean TL in DCE increased (t_{80} = 3.40; $P < 0.001$) from 40 mm (± 4) during the pre to 58 mm (± 7) during the post and decreased ($t_{34.9}$ = -5.17; $P < 0.0001$) in ECE from 58 mm (± 2) during the pre to 53 mm (± 5) during the post.

White bass (*Morone chyrsops*) accounted for 88% of *Moronidae* catch (gill net and electrofishing) with hybrid bass (*Morone chrysops x saxatalis*) comprising the rest. No differences ($P > 0.05$) in CPE (gill net or electrofishing), condition, Wr, or TL were detected for either species at either site. Notably however, white bass CPE (gill net) in ECE decreased from 30 fish/net, during fall of 1994, to an average of 6 fish/net during the post. Hybrid bass were only sampled during fall, 2002, but occurred in both embayments. Twelve hybrid bass were collected in ECE compared with only three in DCE.

Channel catfish (*Ictalurus punctatus*) represented 80% of the catfish sampled with flathead catfish (*Pylodictus olivaris*) comprising the remainder. Low sample sizes during the pre treatment prevented pre versus post comparisons among sites (Appendix I,II). Low sample sizes also prevented statistical analysis of mobile logperch (*Percina kathae*), yellow perch (*Perca flavescens*), and common carp (*Cyrpinus carpio*) (Appendix I,II). Mobile logperch were collected at low densities but occurred at both sites and during both treatments. Yellow perch was only collected once, during summer, 2004 in DCE. Common carp was only collected during the post treatment but occurred in both embayments equally.

Benthic Invertebrates from Hester-Dendy Multiplate Samplers

A total of 10,022 benthic invertebrates representing 13 Orders, 21 Families, and 46 taxa were collected from Hester-Dendy multiplate samplers at both sites during the study period. About 77% of the benthic invertebrates sampled consisted of chironomid midges (Diptera), caddisfly larvae (Trichoptera), and aquatic worms (Oligochaeta).

Mean density for samples collected during the fall season in ECE remained similar following diversion (t_4 = -0.19; P > 0.85). However, diversity dramatically declined ($t_{1,272}$ = 18.2; P < 0.0001), as did evenness (Table 6). Following diversion, mean densities of chironomid midges and trichopteran larvae, mainly *Cyrnellus fraternus,* declined 78% and 98%, respectively. Samples collected during the post treamtent were dominated by cladoceran zooplankton, which accounted for 91% of total density. Filter-collectors remained the predominant trophic group following diversion but data collected in 2004 represented five trophic levels, as opposed to only three collected in 1994.

Densities collected during the summer season also remained similar (ANOVA: $F_{2,6}$ = 1.5; P > 0.29) following diversion; although total density in 2002 appeared much higher than in 1994 or 2004 (Table 7). Increased densities of *Glyptotendipes* spp., *Ablabesmyia* spp., and Naidid oligochaetes were responsible for the elevated mean. Most notable overall was a 96% decrease in mean density of *Cyrnellus fraternus* between 1994 and 2004. Species richness, diversity, and evenness increased steadily over the ten-year period. In 1994, only three trophic levels were represented and were dominated by filter-collectors and gather-collectors. All five trophic levels were collected in 2004.

Only two of three multiplate samplers were recovered from DCE during the fall season in 2004. As a result, any disparity between years, such as the declines in species richness, diversity, and evenness, may have been directly attributable to the lost sampling effort and not due to environmental factors (Table 8). Nonetheless, total density significantly increased ($t_{685} = 24.6$; $P < 0.0001$) following diversion, primarily due to a 94% increase in cladoceran zooplankton. Mean density of Chironomidae declined about 82%. Trophic levels collected during both years were dominated by filter-collectors.

In DCE, mean density for data collected during the summer season remained similar (ANOVA: $F_{2,6} = 0.74$; $P > 0.51$); while; species richness, diversity, and evenness increased steadily over the ten year period (Table 9). Diversity values in 2004 were significantly higher ($t_{871} = -5.45$; $P < 0.0001$) than those recorded in 1994. Insects of the Family Ceratopogonidae and Polycentropodidae declined 18% and 82%, respectively, while those belonging to Naididae increased 96%. Three trophic levels were collected in 1994 and were dominated by filter-collectors and shredders. All five trophic levels were represented in 2004 and were dominated by filter-collectors and gather-collectors.

Benthic Invertebrates from Bottom Dredge Samples

A total of 2,357 benthic invertebrates representing 9 Orders, 15 Families, and 30 taxa were collected from bottom dredge samples from both sites during the study. The phantom midge, *Chaoborus* spp. (Chaoboridae), comprised about 79% of benthic invertebrates sampled. Other dominant taxa groups consisted of chironomid midges (Diptera), and aquatic worms (Oligochaeta), which made up an additional 15% of abundance.

In ECE, mean density for data collected during the fall season remained similar (t_4 = 1.56; P < 0.195) following diversion (Table 10). Species richness also remained similar but diversity and evenness significantly increased (t_{360} = -4.13; P < 0.0001) during the post. Mean densities of Chaoboridae, Chironomidae, Copepoda, and Sphaeriidae all decreased following diversion while those of Tubificidae and cladoceran zooplankton increased. Predators, mainly *Chaoborus* spp., were the dominant trophic group in all samples but substantial increases in shredders and gather-collectors were apparent in the 2004 samples.

In ECE, mean density was significantly higher (ANOVA: $F_{2,6}$ = 19.68; P < 0.0023) during 2002 for data collected during the summer season, but remained similar between 1994 and 2004 (Table 11). Data collected in 2002 had the highest densities of most taxa, particularly Chaoboridae and Sphaeriidae. While taxa richness was highest during 2002, diversity and evenness significantly declined (t_{191} = 7.72; P < 0.0001) about 86% between 1994 and 2004. Insects of the Family Chironomidae were most abundant in 1994 but had decreased by 90% in 2004. Predators remained the predominant trophic level following diversion.

In DCE, mean density for data collected during the fall season remained similar (t_4 = -1.18; P < 0.303) following diversion (Table 12). Five fewer taxa were collected during the post but diversity remained similar (t_{213} = 0.50; P < 0.618), as did evenness. Most notable were the increased densities of Chaoboridae and Tubificidae following diversion. Predators remained the predominant trophic group; although gather-collectors and shredders were substantially higher during the post.

In DCE, mean density was higher following diversion for data collected during the summer season (Table 13), though the trend was not statistically significant (ANOVA: $F_{2,6} = 3.51$; $P < 0.303$). Taxa richness was highest during 1994 and decreased by almost half in 2004. Diversity and evenness steadily declined following diversion. Mean densities of Chaoboridae recorded in 2002 (5,541/m^2) and 2004 (5,733/m^2) were twice those recorded in 1994 (2,326/m^2). This was also reflected by the increased dominance of predators during the post (Table 13).

V. DISCUSSION

Nutrient removal resulted in sharp declines in TP and SRP in ECE. Consonant with these declines was a 31% decrease in phytoplankton biomass as well as significant changes in condition and abundance of fish species and density and diversity of macroinvertebrates. However, CHLA values by the end of the study remained nearly twice those recorded in DCE, perhaps indicating the persistence of key nutrients resulting from decades of internal loading and continued non-point source (urban) influence. Meanwhile, elevated hypolimnetic conductivity at the mouth of DCE following diversion suggested the possible encroachment of diverted sewage. However, water quality measurements and phytoplankton biomass remained static following diversion. Biotic communities in DCE displayed mixed results to the potential enrichment.

Elkahatchee Creek Embayment

At E1, TP and SRP declined 77% and 96% following nutrient diversion resulting in a 31% decline in phytoplankton biomass. The influence of TN on CHLA during the pre (78% of the variation in CHLA) and the lack of change in TN levels between treatments suggested that ECE was nitrogen limited during the pre due to high phosphorous loading. This was also supported by the low TN:TP ratio (7.5). Downing and McCauley (1992) reported similarly low ratios associated for: cattle manure seepage (8.9), feedlot runoff (6.4) and sewage (5.3). Following diversion, the TN:TP ratio (15.1) increased to levels more similar to those recorded in DCE (14.6–16.6) and similar to average river water (18.9) (Ebise and Inoue 1991). Moreover, the strong relationship between TN and CHLA decoupled leaving TP as the most significant regressor.

Despite the decrease in phytoplankton biomass following nutrient diversion, CHLA in ECE during the post was still slightly higher than that recorded in DCE. I believe that urban runoff and internal P loading from hypolimnetic sediments were primarily responsible for the elevated CHLA during the post-diversion. Lung and Larson (1995) attributed internal P loading for supplying up to one-third of daily TP loads in Lake Pepin, MS. Edmondson and Lehman (1981) reported over half of the total P income into Lake Washington, WA over a 27-year period was attributed to internal P loading from the sediment. Although sediment P was not sampled, specific conductance values (used as a means of tracking effluent) collected from the bottom were highly correlated with SRP ($R = 0.727$; $P < 0.0001$) during the pre-diversion. Mean conductivity in ECE remained slightly higher than detected at D1 following diversion which suggested that decades of point-source discharge, along with urban runoff, were still affecting the embayment.

Consonant with the decline in CHLA were significant changes in the fish and invertebrate communities. Black bass dominance shifted to favor spotted bass following nutrient removal. Buynak et al. (1989) reported a longitudinal shift in black bass dominance as spotted bass comprised the majority of black bass in the oligotrophic portions of Cave Run Lake, Kentucky. Greene and Maceina (2000) found greater densities of age-0 spotted bass in mesotrophic and oligotrophic reservoirs and greater densities of age-0 largemouth bass in eutrophic reservoirs. Maceina and Bayne (2001) found fewer largemouth bass with lower relative weights and more abundant, yet smaller, spotted bass following oligotrophication in West Point Reservoir, GA.

Spotted bass became more abundant, but less robust following oligotrophication. A study of 10 Alabama reservoirs detected lower Wr in spotted bass sampled from oligo-mesotrophic systems as compared with eutrophic systems (DiCenzo et al. 1995). A later study of 32 Alabama impoundments reported poorer body conditions of spotted bass in oligomesotrophic reservoirs, compared with fish sampled from eutrophic systems (Maceina et al. 1996). Several studies have also documented slower growth of spotted bass in lakes of lower trophic status, though this trend was not detected in our results (Maceina and Bayne 2001; DiCenzo et al. 1995).

Largemouth bass were longer yet less robust following oligotrophication. Maceina and Bayne (2001) reported that crappie expressed higher relative weights and exhibited faster growth in more eutrophic reservoirs. Bayne et al. (1994) reported smaller largemouth bass in the mesotrophic Lake Martin (TSI = 39) compared with the moderately eutrophic Lake Eufaula (TSI = 57), Alabama. However, growth rates (mm TL) in Lake Martin remained similar to those sampled in the highly eutrophic Weiss Lake (TSI = 65) and moderately eutrophic Jones Bluff Lake (TSI = 55).

Black crappie comprised 19% of the fish community by abundance prior to diversion, compared with < 2% following oligotrophication (gill net and electrofishing). Hoxmeier and DeVries (1998) found that crappie contributed a larger proportion to the fish community in oligotrophic reservoirs. In addition, black crappie were less robust following oligotrophication. Maceina et al. (1996) reported that crappie were generally better conditioned and exhibited faster crappie growth in more eutrophic reservoirs. Bayne et al. (1994) also detected faster crappie growth in eutrophic systems.

Bluegill sunfish were significantly shorter and poorer conditioned during the post. A study of four Alabama impoundments found that sunfish were less abundant in a mid-range mesotrophic system (TSI: 39) than in two eutrophic systems (TSI: 55, 57) (Bayne et al. 1994).

Low sample sizes prevented any statistical comparisons of gizzard shad. However, based on previous studies, I would have expected less abundant, yet larger, more robust, and faster growing fish with decreasing productivity (Clayton and Maceina, 2002; Bremigan and Stein 2001; Allen et al. 1999; Hoxmeier and DeVries, 1998; DiCenzo et al. 1996; McQueen et al. 1998; and Bayne et al. 1994). This was similar to the response I observed for threadfin shad, which were longer and had higher average weights following oligotrophication.

Macroinvertebrate community structure shifted after diversion, but densities remained static following oligotrophication in ECE, except for the spike observed in summer, 2002 (Table 7). Dramatic declines in *Cyrnellus fraternus, Glytpotendipes* spp., and *Dictrotendipes* spp. corresponded with significantly higher diversity and evenness in Hester-Dendy summer collections following diversion in ECE. The dominance of several tolerant species is a key signature of degraded systems (Karr and Chu 1999). Improved water quality allows broader partitioning of the resource. The presence of all five trophic levels following diversion as opposed to only three in 1994 was consistent with this aspect of recovery.

Bottom dredge samples from ECE revealed more variable results. Density remained static between treatments, except for the summer 2002 spike (Table 11). Summer diversity steadily declined throughout the study periods and was significantly lower in 2004. This was likely due to the overwhelming dominance (97% of total density) of *Chaoborus spp*. in 2004. The ability of a taxon to grossly out compete all others is usually indicative of a stressed system and is not suggestive of a stable recovery.

Fall collections from ECE revealed that *Chaoborus* spp. and *Coelotanypus* spp. comprised 90% of total density prior to nutrient diversion (see Appendix Table 15). Popp and Hoagland (1995) also reported a 90% composition of these two taxa following increased sediment enrichment in Pawnee Reservoir, NE. Following oligotrophication, diversity and evenness during the fall significantly increased in ECE, which corresponded with: 1) lower densities of *Chaoborus* spp.; 2) a more even distribution of chironomid taxa; 3) increased densities of Tubificidae, and; 3) the addition of a fourth trophic level (shredders) (Table 10). Notably however, the presence of shredders was due solely to the appearance of *Chironomus spp*., which together with Tubificidae was also a pollution tolerant species and also reported at high densities by Popp and Hoagland (1995). Bottom dredge diversity steadily declined during the summer. These results suggest an unstable recovery in the profundal sediments of ECE.

Dennis Creek Embayment

Phytoplankton biomass and TSI in DCE remained similar between treatment periods. Moreover, no changes in key nutrients, such as TP, SRP, TN, or the TN:TP ratio were detected. Specific conductance however increased significantly at both stations in DCE following nutrient diversion. These values were particularly high at D1, located nearest the point of sewage diffusion immediately between the mouths of DCE and ECE (Figure 2).

During periods of chemical stratification, the plume of effluent-enriched water would have entered into DCE via the hypolimnion. I speculate that the substantial increase in conductivity recorded in DCE following effluent diversion, particularly in the mouth, may be evidence of sewage encroachment. However, water quality results provided no evidence of nutrient transfer to the photic zone.

Although trophic state and nutrient levels remained static following diversion, certain fish species displayed the characteristics indicative of eutrophication. For example, as system productivity increases, the black bass dynamic tends to favor largemouth bass (Buynak et al. 1989). Certainly, both species have been shown to exhibit faster growth and better condition in eutrophic systems (DiCenzo et al, 1995; Greene and Maceina, 2000; Allen et al., 1999), but abundance and biomass of largemouth bass can be much higher than for spotted bass (Greene and Maceina, 2000; Buynak et al., 1989). At the end of this study, spotted bass remained the dominant black bass. However, they became less abundant, shorter, and expressed lower relative weights following nutrient diversion; whereas, largemouth bass were longer and had higher condition. These data are inconsistent with existing research with respect to Wr of spotted bass. However, my data suggest that the black bass dynamic may be shifting to favor the production of largemouth bass.

Lepomis spp. also responded in a manner consistent with eutrophication. Bluegill sunfish were longer and more robust following diversion. No differences in abundance were evident, but bluegill sunfish have been shown to increase in numbers at higher trophic levels (McKenna 2001; Bayne et al. 1994). Redear sunfish also expressed higher condition following diversion. While no references were found pertaining to redear condition or growth, a study of 65 Florida lakes revealed increased standing crop with increasing trophic status (Bachmann et al. 1996).

Not all species responded in a manner consistent with nutrient enrichment. Black crappie were significantly shorter following diversion, though no differences in condition or catch were recorded. Shorter and less robust crappie have been associated with lower productivity in several southeastern reservoirs (Maceina et al. 1996; Bayne et al. 1994). However, size difference among treatments was likely more related to variable recruitment than trophic status.

Blacktail shiners and weed shiners were longer following oligotrophication. No studies were found comparing cyprinid metrics across the trophic gradient in southeastern reservoirs. Persson et al. (1991) reported increased size of cyprinids among 13 Swedish lakes as trophic status increased from low to medium productivity. The larger sizes were reported as a size-refuge response to the increased importance of piscivory in medium productive lakes.

Gizzard shad normally respond to eutrophication with increased abundance, slower growth, and lower Wr (Clayton and Maceina, 2002; DiCenzo et al. 1996). I found no differences in abundance, but these fish were significantly more robust following diversion. Threadfin shad were shorter and average weight declined following diversion; a reversal of the trend detected in ECE.

Both shad species are omnivorous filter feeders relying primarily on zooplankton, phytoplankton, and organic detritus (Baker and Schmitz 1971; Cramer and Marzolf 1970; Kutkhun 1957). Also, both species typically show increases in abundance and biomass as trophic status increases in southeastern reservoirs (Allen et al. 2000; DiCenzo et al. 1996; Harman et al. 1995). However, my data suggest that the relative abundance of these shad species to one another may shift along the trophic gradient.

Although no differences in CPE were detected, the percentage of total shad represented by threadfin shad increased 69% during the post treatment. I found no studies that explicitly compared the relative abundance of these shad species to one another across the trophic gradient. However, data taken from Bayne et al. (1994) corroborated this trend and showed the percentage of total shad represented by threadfin shad increased 45% as trophic level increased among four Alabama reservoirs. Indeed, a number of mechanisms might explain such a trend: 1) increased vulnerability of gizzard shad at higher trophic status; 2) capture efficiency in more turbid waters; or; 3) shifts in food and/or habitat availability. If this is a legitimate response, rather than a spurious coincidence, then the relative proportion of threadfin shad could provide a useful and sensitive index for assessing trophic changes in southeastern reservoirs.

Macroinvertebrates inhabiting the littoral zone of DCE did not respond in a manner consistent with nutrient enrichment. Density from Hester-Dendy samplers remained static, richness and evenness increased, and diversity rose by 16% following nutrient diversion. In 1994, Polycentripodidae larvae, mostly *Cyrnellus fraternus*, comprised about 44% of total density. By 2004, density declined to less than 2%. These trends were more indicative of stress recovery and were especially similar to those recorded in ECE. Generally, nutrient enrichment results in greater densities, lower diversity and evenness, and increasing dominance of select taxa (Popp and Hoagland 1995; Welch 1980; Dermott et al. 1977; Peterka 1972; Jónasson 1969; Carr and Hiltunen 1965).

Macroinvertebrates inhabiting the bottom sediments displayed variable responses between seasons that were both consistent and inconsistent with nutrient enrichment. Density appeared higher during both seasons following diversion but the increases were not statistically significant ($P > 0.05$). The ability of ANOVA and t-tests to detect these changes was likely undermined by high variability among replicates. Seasonable variability is often pronounced in lentic macroinvertebrate sampling (Hergenrader and Lessig 1980; Parrish and Wilhm 1978). Fall diversity remained similar among treatments, but structural changes between pre and post displayed characteristics indicative of nutrient enrichment. For example, five taxa present in 1994 were absent in 2004, including the burrowing mayfly (*Hexagenia* spp.). Carr and Hiltunen (1965) reported a 99% decline in *Hexagenia* spp. resulting from organic pollution during a 31-year study of western Lake Erie. Also, densities of Tubificidae and Chaoboridae were dramatically higher in DCE following nutrient diversion. These species are generally associated with organic enrichment during eutrophication (Merritt and Cummins 1996).

Following diversion, summer diversity declined in response to the increasing dominance of Chaoborus spp., which comprised 98% of total density in 2004. However, Tubificidae and Chironomidae densities sharply declined during the post. These taxa are also associated with organic enrichment and would not be expected to decline with increasing eutrophication. Moreover, sharp increases in *Chaoborus* spp. were also recorded in 2004 in ECE.

Summary

Oligotrophication in ECE produced measurable responses within the biotic community. Nutrient reduction resulted in lower algal biomass, a shift in the black bass community, decreased catch and condition of sunfish and crappie, and altered structure and function of littoral macroinvertebrate communities. Despite a 31% decrease following diversion, phytoplankton biomass remained nearly twice that recorded in DCE. This delayed recovery is likely attributed to non-point source (urban) discharge and internal phosphorous loading. Results from other studies suggest that these effects may persist for many years.

Circumstantial evidence suggested that diverted sewage may have infiltrated the DCE. However, despite increased conductivity in DCE, no changes conclusive to nutrient enrichment were found with respect to: nutrients, algal biomass, or macroinvertebrate assemblages. Though, Centrarchid fish (black bass and sunfish) responded in a manner consistent with eutrophication.

Most research assessing faunal changes across the trophic gradient compare systems of disparate trophic status. However, ECE and DCE displayed only mild fluctuations in trophic status between treatments. As provided in this thesis, both pronounced and subtle changes in the nutrient regime can produce a measurable response within the aquatic community. Understanding the nature and magnitude of this response is key for managing the balance between aesthetically clearer water and fish production.

Table 1. Analytical methods used for assessing water quality variables in ECE and DCE in Lake Martin, Alabama.

Variable	Method	Reference
<u>In Situ</u>		
Temperature	thermistor	APHA, 1998
Dissolved oxygen	membrane electrode	APHA, 1998
pH	glass electrode	APHA, 1998
Specific conductance	conductivity cell	APHA, 1998
Visibility	Secchi disk	APHA, 1998
Euphotic zone determination	submarine photometer	Lind, 1985
<u>Laboratory Analyses</u>		
Total suspended solids	vacuum filtration	APHA, 1998
Turbidity	nephelometric	APHA, 1998
Alkalinity	potentiometric titration	APHA, 1998
Total ammonia (NH_3-N)	phenate method	APHA, 1998
Nitrite (NO_2-N)	diazotizing method	APHA, 1998
Nitrate (NO_3-N)	cadmium reduction	APHA, 1998
Total nitrogen	persulfate method	APHA, 1998
Total phosphorous	persulfate method, ascorbic acid	APHA, 1998
Soluble reactive phosphorous	ascorbic acid	APHA, 1998
Total hardness	EDTA titrimetric method	APHA, 1998
Chlorophyll *a*	Spectrophotometric	APHA, 1998

Table 2. Sampling schedule for water quality, phytoplankton, invertebrates, and fish during the 10–year study period in ECE and DCE in Lake Martin, Alabama.

	Year	Water Quality/ Phytoplankton	Invertebrate	Fish
Pre	1994	April – October	August; October	July; October
	1997	May – June	…	…
	1998	June – August	…	…
Post	2002	June – August	July	July; October
	2003	April; June; August	...	July
	2004	April - October	August; November	July; October

Table 3. Summary of Pearson Correlation Coefficients for chlorophyll *a* concentrations in ECE before nutrient removal. The water quality variables that were significantly correlated are reported in the table along with sample size (*n*) and *P* values.

Variable	*r*	*n*	*P* <
pH	0.881	24	0.0001
Total Nitrogen	0.884	14	0.0001
Turbidity	0.770	24	0.0001
Total Suspended Solids	0.770	24	0.0001
Secchi Disk Visibility	-0.859	12	0.0003
Compensation Point	-0.726	22	0.0001
Secchi Disk Visibility	-0.774	24	0.0001

Table 4. Summary of Pearson Correlation Coefficients for chlorophyll *a* concentrations in ECE following nutrient removal. The water quality variables that were significantly correlated are reported in the table along with sample size (*n)* and *P* values.

Variable	*r*	*n*	*P* <
Total Phosphorous	0.488	34	0.0034
Total Suspended Solids	0.526	34	0.0014
Compensation Point	-0.372	34	0.0302
Secchi Disk Visibility	-0.424	34	0.0123

Table 5. Summary of Pearson Correlation Coefficients for chlorophyll *a* concentrations in DCE. The water quality variables that were significantly correlated are reported in the table along with sample size (*n)* and *P* values.

Variable	*r*	*n*	*P* <
Total Nitrogen	-0.401	44	0.0070
Total Ammonia-Nitrogen	-0.403	54	0.0025
Nitrate-Nitrogen	-0.298	54	0.0285

Table 6. Mean values (± SE) of benthic invertebrate assemblage variables from Hester-Dendy multiplate samples taken during the fall season in ECE in Lake Martin, Alabama.

Community Variables	1994	2004
Species Richness	16	13
Shannon Diversity	1.61 (± 0.002)	0.45 (± 0.002)
Evenness	0.581	0.17
Density (# / m^2)		
Total Density	2674 (± 225)	2822 (± 751)
Baetidae	7	…
Caenidae	…	4
Ceratopogonidae	15	…
Chironomidae	896	200
Cladocera	44	2570
Coenagrionidae	7	…
Copepoda	7	…
Hydroptilidae	…	4
Naididae	…	7
Oligochaeta	37	…
Pleuroceridae	…	4
Polycentropodidae	1274	26
Filter-Collector	6000 (± 1155)	2707 (± 774)
Gather-Collector	455 (± 58)	18 (± 3)
Shredder	…	26 (± 15)
Predator	107 (± 10)	…
Piercer	…	11 (0)
Scraper	…	17 (± 5)

Table 7. Mean values (± SE) of benthic invertebrate assemblage variables from Hester-Dendy multiplate samples taken during the summer season in ECE in Lake Martin, Alabama.

Community Variables	1994	2002	2004
Species Richness	12	15	22
Shannon Diversity	1.04 (± 0.001)	1.88 (± 0.0006)	2.17 (± 0.001)
Evenness	0.42	0.69	0.70
Density (# / m^2)			
Total Density	7593 (± 4568)	10633 (± 1173)	3519 (± 1785)
Caenidae	…	30	26
Chironomidae	4267	7433	2111
Cladocera	22	356	748
Coenagrionidae	7	30	7
Copepoda	44	…	15
Heptageniidae	…	…	15
Hydroptilidae	…	30	4
Naididae	74	1156	400
Nematoda	…	207	…
Ostracoda	7	59	44
Polycentropodidae	3140	1304	126
Sphaeriidae	…	…	22
Turbellaria	..	30	…
Filter-Collector	4163 (± 2737)	6352 (± 255)	1881 (± 1199)
Gather-Collector	3052 (± 1779)	1863 (± 425)	570 (± 144)
Shredder	…	…	133 (± 86)
Predator	296 (± 80)	922 (± 695)	237 (± 132)
Piercer	…	89 (0)	11 (0)
Scraper	…	…	15 (2)

Table 8. Mean values (± SE) of benthic invertebrate assemblage variables from Hester-Dendy multiplate samples taken during the fall season in DCE in Lake Martin, Alabama.

Community Variables	1994	2004**
Species Richness	17	7
Shannon Diversity	1.78 (± 0.003)	0.216 (± 0.001)
Evenness	0.630	0.111
Density (# / m^2)		
Total Density	1648 (± 460)	3611 (± 1900)
Caenidae	4	…
Chironomidae	630	112
Cladocera	200	3472
Harpacticoida	7	…
Naididae	59	6
Oligochaeta	7	…
Ostracoda	15	…
Polycentropodidae	…	17
Filter-Collector	1140 (± 268)	3594 (± 946)
Gather-Collector	293 (± 98)	11 (0)
Shredder	100 (± 83)	11 (0)

** Only 2/3 multiplate samplers were recovered from study site

Table 9. Mean values (± SE) of benthic invertebrate assemblage variables from Hester-Dendy multiplate samples taken during the summer season in DCE in Lake Martin, Alabama.

Community Variables	1994	2002	2004
Species Richness	18	19	21
Shannon Diversity	1.86 (± 0.002)	1.99 (± 0.003)	2.21 (± 0.001)
Evenness	0.64	0.69	0.73
Density (# / m^2)			
Total Density	2200 (± 178)	1563 (± 520)	2059 (± 388)
Caenidae	…	…	63
Ceratopogonidae	22	…	4
Chironomidae	1007	352	767
Cladocera	44	78	222
Coenagrionidae	7	7	22
Corduliidae	…	4	…
Copepoda	…	11	11
Heptageniidae	…	4	4
Hydroptilidae	…	59	48
Naididae	22	537	541
Oligochaeta	74	…	…
Ostracoda	22	93	337
Polycentropodidae	963	81	30
Sphaeriidae	…	…	4
Turbellaria	…	311	…
Filter-Collector	1189 (± 151)	311 (± 33)	559 (± 70)
Gather-Collector	622 (± 57)	641 (± 263)	660 (± 85)
Shredder	1561 (-)	333 (± 185)	37 (± 20)
Predator	256 (± 32)	…	44 (± 22)
Piercer	…	59 (± 43)	22 (-)
Scraper	…	11 (-)	52 (± 37)

Table 10. Mean values (± SE) of benthic invertebrate assemblage variables from dredge samples taken during the fall season in ECE in Lake Martin, Alabama.

Community Variables	1994	2004
Species Richness	9	9
Shannon Diversity	1.02 (± 0.004)	1.42 (± 0.006)
Evenness	0.46	0.65
Density (# / m^2)		
Total Density	3585 (± 488)	2593 (± 411)
Chaoboridae	2044	1289
Chironomidae	1274	963
Cladocera	…	44
Copepoda	178	44
Corbiculidae	15	…
Oligochaeta	15	…
Sphaeriidae	59	15
Tubificidae	…	237
Filter-Collector	111 (± 54)	59 (± 15)
Gather-Collector	74 (± 29)	237 (± 115)
Shredder	…	667 (± 245)
Predator	3259 (± 565)	1585 (± 660)

Table 11. Mean values (± SE) of benthic invertebrate assemblage variables from dredge samples taken during the summer season in ECE in Lake Martin, Alabama.

Community Variables	1994	2002	2004
Species Richness	6	8	4
Shannon Diversity	1.04 (± 0.009)	0.42 (± 0.003)	0.15 (± 0.005)
Evenness	0.58	0.20	0.11
Density (# / m^2)			
Total Density	1778 (± 92)	5852 (± 752)	2622 (± 362)
Chaoboridae	1259	5304	2548
Chironomidae	311	15	30
Cladocera	…	44	…
Copepoda	74	15	…
Leptoceridae	…	15	…
Nematoda	…	15	…
Sphaeriidae	133	3111	44
Tubificidae	…	134	…
Filter-Collector	133 (± 51)	356 (± 205)	44 (± 0)
Gather-Collector	…	163 (± 78)	…
Shredder	…	11 (-)	11 (-)
Predator	1526 (± 150)	5304 (± 417)	2563 (± 376)

Table 12. Mean values (± SE) of benthic invertebrate assemblage variables from dredge samples taken during the fall season in DCE in Lake Martin, Alabama.

Community Variables	1994	2004
Species Richness	12	7
Shannon Diversity	1.26 (± 0.015)	1.16 (± 0.012)
Evenness	0.51	0.60
Density (# / m^2)		
Total Density	1318 (± 449)	2074 (± 456)
Chaoboridae	756	1244
Chironomidae	430	370
Cladocera	15	30
Copepoda	15	44
Corbiculidae	15	…
Ephemeridae	15	…
Tubificidae	74	385
Filter-Collector	89 (± 0)	89 (-)
Gather-Collector	133 (± 36)	385 (± 97)
Shredder	67 (± 18)	267 (± 77)
Predator	1096 (± 365)	1348 (± 418)

Table 13. Mean values (± SE) of benthic invertebrate assemblage variables from dredge samples taken during the summer season in DCE in Lake Martin, Alabama.

Community Variables	1994	2002	2004
Species Richness	11	4	6
Shannon Diversity	1.34 (± 0.009)	0.38 (± 0.002)	0.11 (± 0.002)
Evenness	0.58	0.27	0.06
Density (# / m^2)			
Total Density	3111 (± 353)	6133 (± 975)	5852 (± 1143)
Ceratopogonidae	30	…	…
Chaoboridae	2030	5541	5733
Chironomidae	370	474	74
Copepoda	341	400	15
Naididae	119	…	…
Nematoda	30	…	…
Sphaeriidae	15	…	…
Tubificidae	178	118	30
Filter-Collector	11 (-)	…	11 (-)
Gather-Collector	296 (± 29)	518 (± 121)	89 (-)
Shredder	156 (± 54)	…	11 (-)
Predator	2326 (± 602)	5541 (± 1033)	5733 (± 1153)

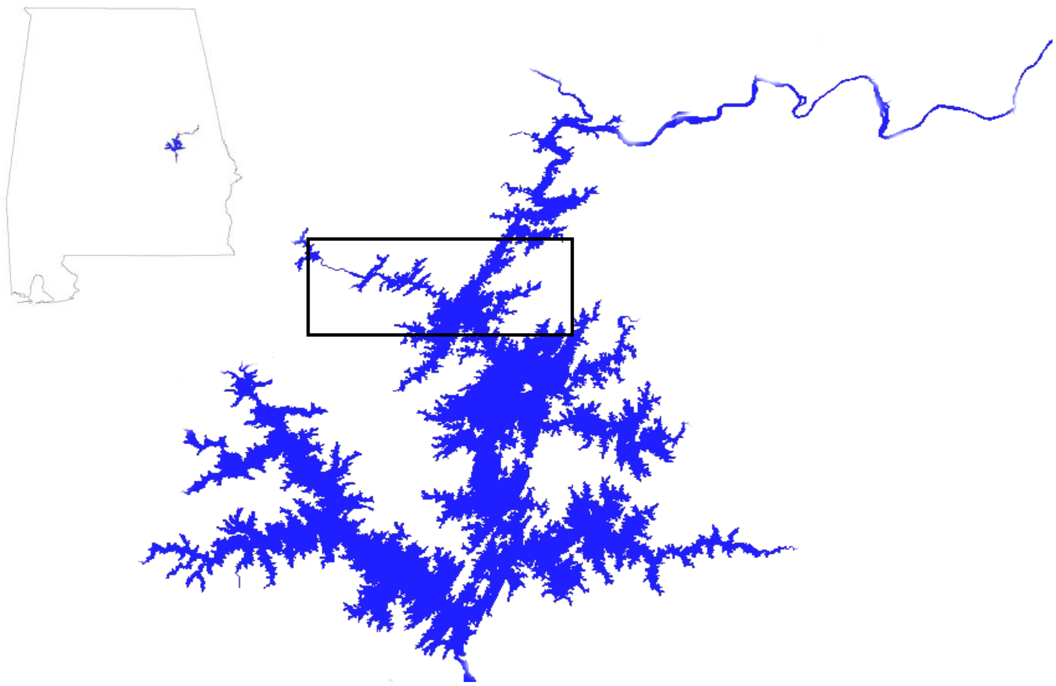

Figure 1. Map of Lake Martin, Alabama and study area.

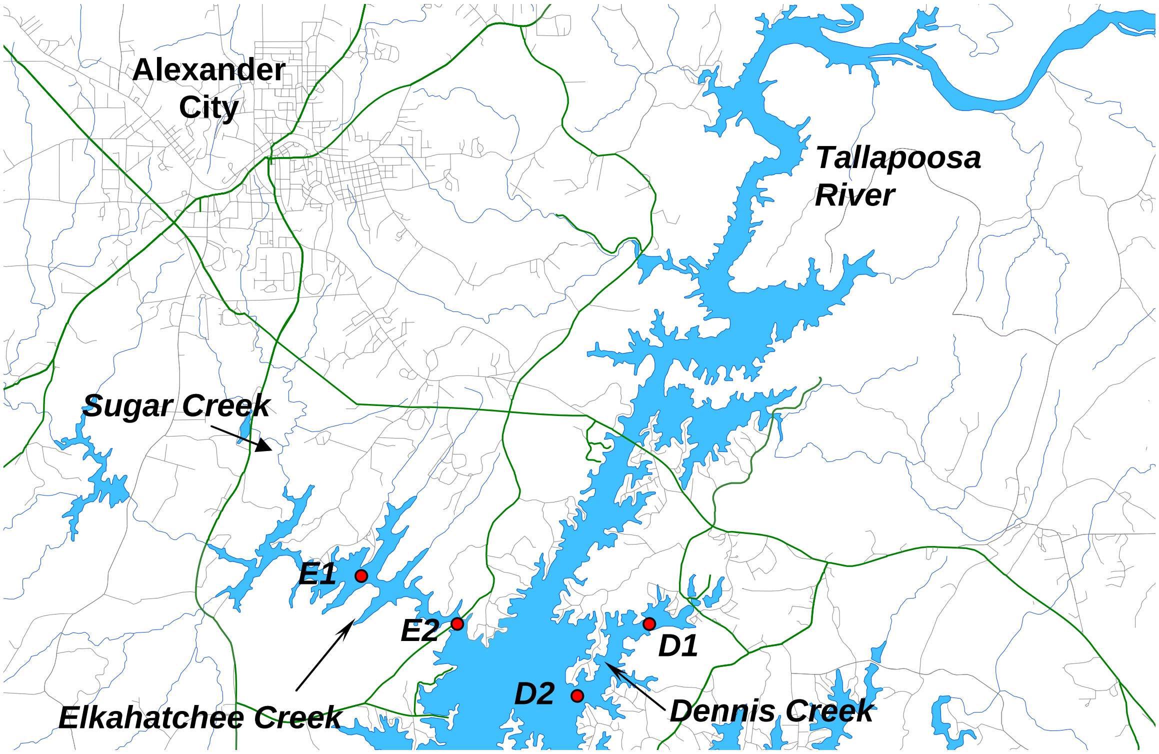

Figure 2. A map of the upstream portion of Lake Martin, Alabama, also containing the study sites, Elkahatchee Cree, Embayment (ECE) and Dennis Creek Embayment (DCE).

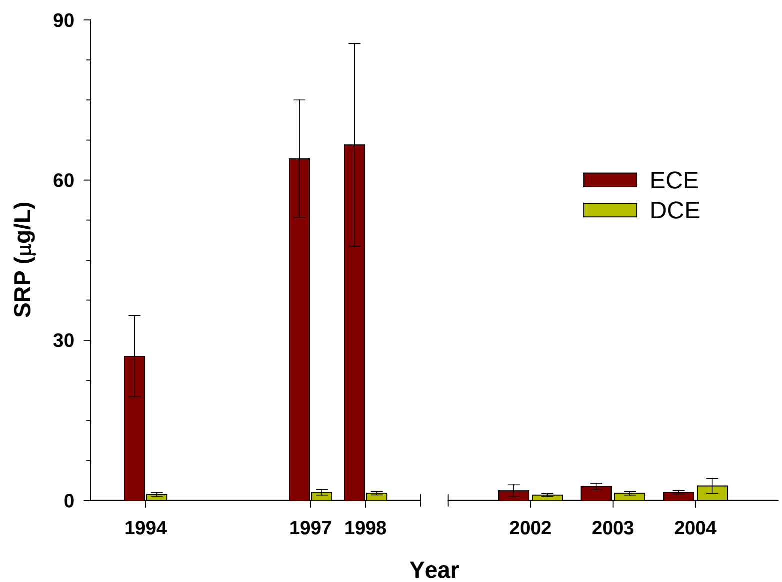

Figure 3. Mean (± SE) growing season soluble reactive phosphorous (SRP) concentrations measured from the photic zone at the mid-embayment sites in Lake Martin at Stations E1 and D1 of ECE and DCE, respectively, from 1994 – 2004.

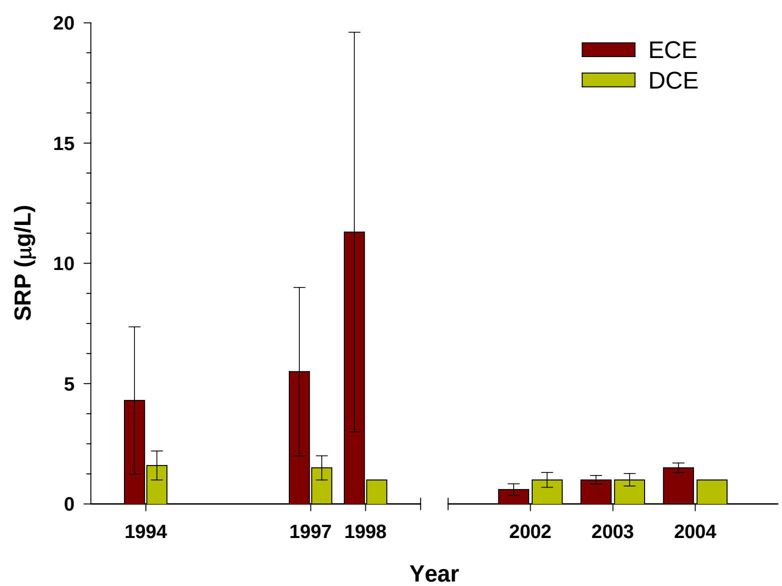

Figure 4. Mean (± SE) growing season soluble reactive phosphorous (SRP) concentrations measured from the photic zone at the downstream sites in Lake Martin at Stations E2 and D2 of ECE and DCE, respectively, from 1994 – 2004.

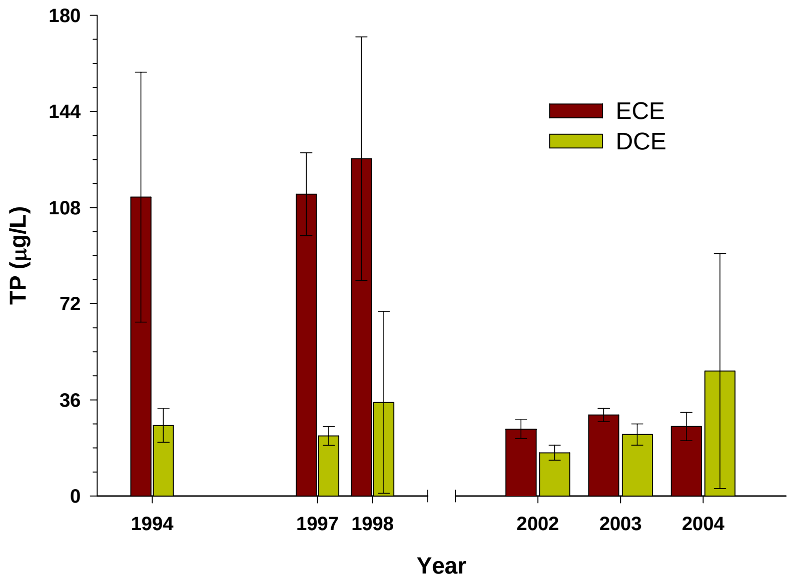

Figure 5. Mean (± SE) growing season total phosphorous (TP) concentrations measured from the photic zone at the mid-embayment sites in Lake Martin at Stations E1 and D1 of ECE and DCE, respectively, from 1994 – 2004.

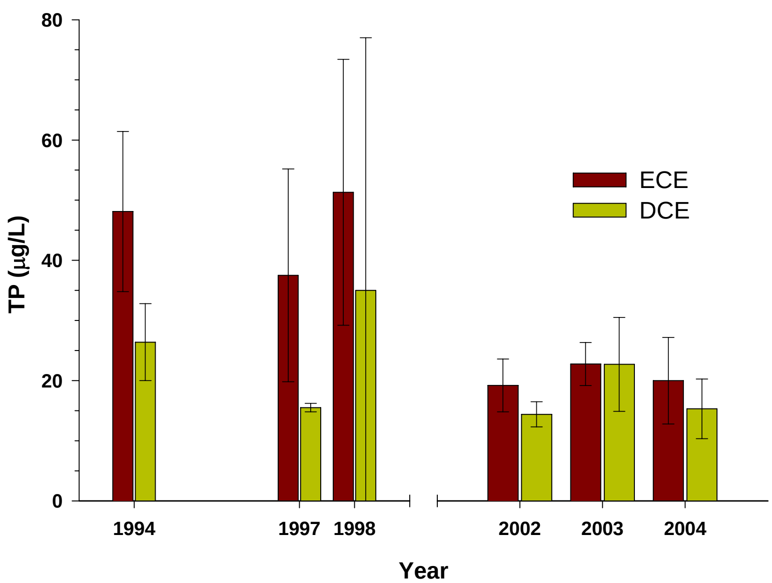

Figure 6. Mean (± SE) growing season total phosphorous (TP) concentrations measured from the photic zone sampled at the downstream sites in Lake Martin at Stations E2 and D2 of ECE and DCE, from 1994 – 2004.

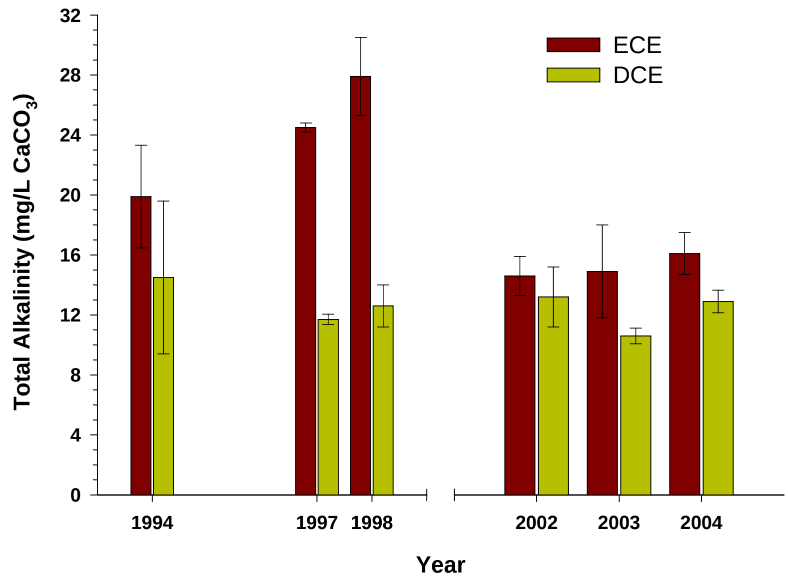

Figure 7. Mean (± SE) growing season total alkalinity measured from the photic zone sampled at the mid-embayment sites in Lake Martin at Stations E1 and D1 of ECE and DCE, respectively, from 1994 – 2004.

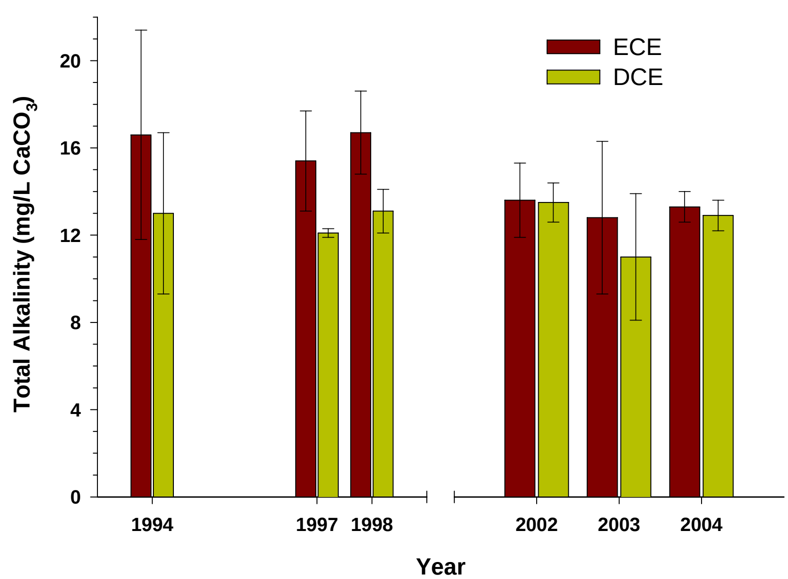

Figure 8. Mean (± SE) growing season total alkalinity measured from the photic zone sampled at the downstream sites in Lake Martin at Stations E2 and D2 of ECE and DCE, respectively, from 1994 – 2004.

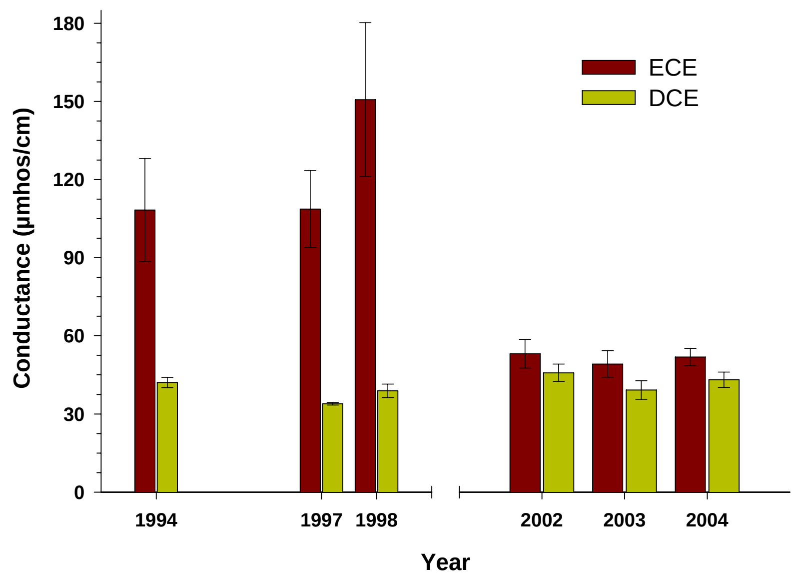

Figure 9. Mean (± SE) growing season specific conductance measured from the photic zone at the mid-embayment sites in Lake Martin at Stations E1 and D1 of ECE and DCE, respectively, from 1994 – 2004.

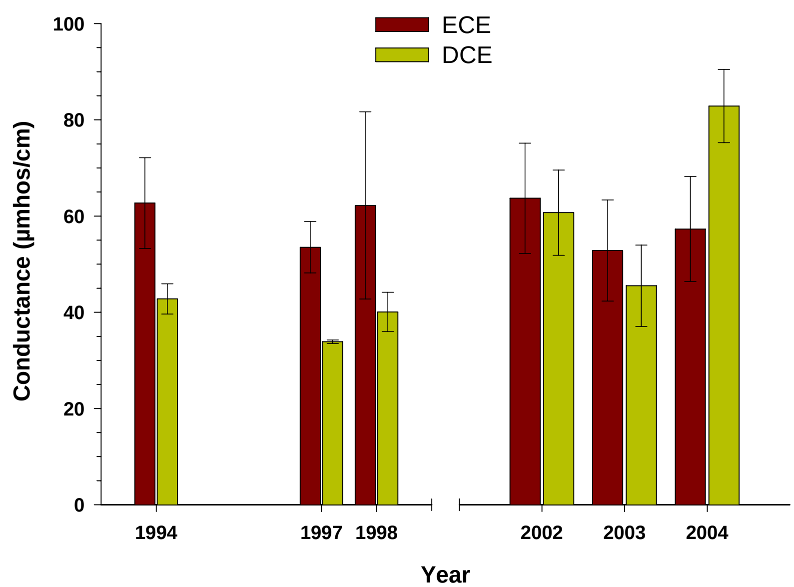

Figure 10. Mean (± SE) growing season specific conductance measured from the photic zone sampled at the downstream sites in Lake Martin at Stations E2 and D2 of ECE and DCE, respectively, from 1994 – 2004.

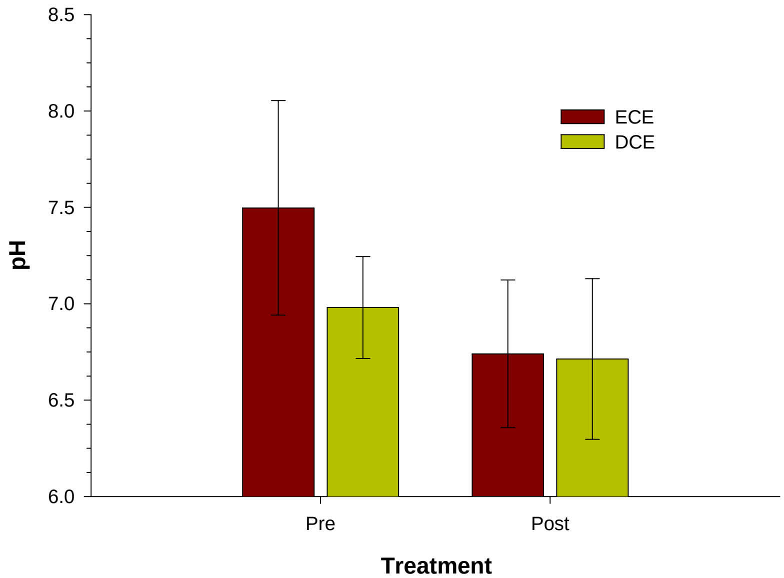

Figure 11. Mean (± SE) growing season pH measured from the photic zone in Lake Martin at Stations E1 and E2 in ECE and Stations D1 and D2 in DCE from 1994–2004.

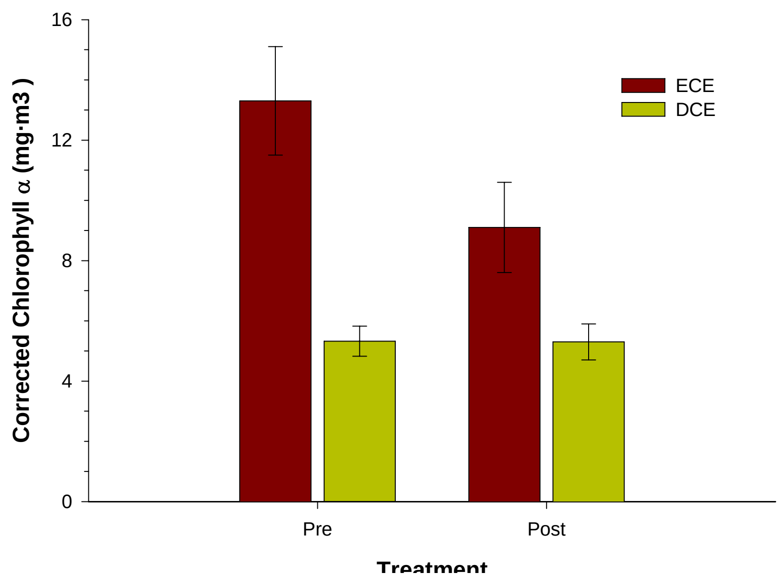

Figure 12. Mean (± SE) growing season corrected chlorophyll *a* values measured from the photic zone in Lake Martin at Stations E1 and E2 in ECE and Stations D1 and D2 in DCE from 1994–2004.

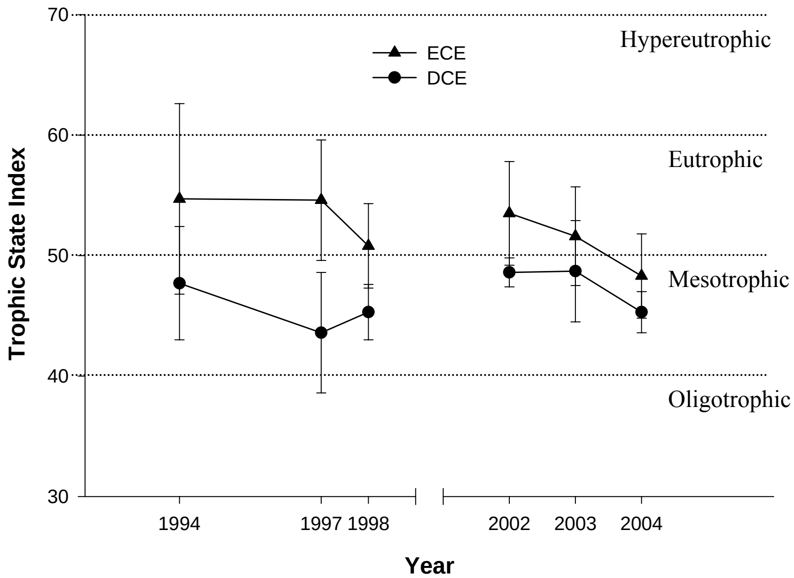

Figure 13. Mean (± SE) growing season TSI values based on corrected chlorophyll *a* measured from the photic zone in Lake Martin at Stations E1 and E2 in ECE and Stations D1 and D2 in DCE from 1994–2004.

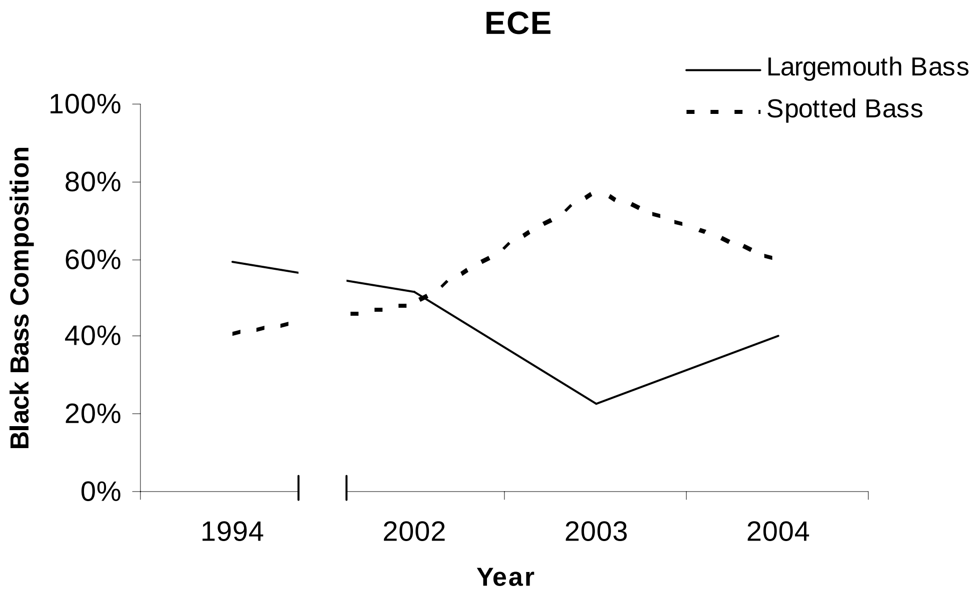

Figure 14. Percent black bass composition represented by spotted and largemouth bass in Lake Martin, Alabama at station E2 in ECE from 1994–2004.

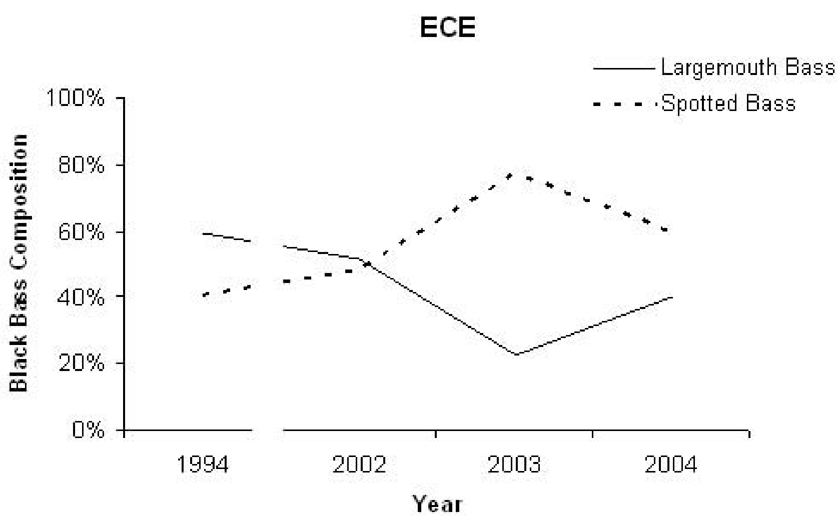

Figure 12. Percent black bass composition represented by spotted and largemouth bass in Lake Martin, Alabama at stations E1 and E2 in Elkahatchee Creek Embayment from 1994 – 2004.

Figure 15. Percent black bass composition represented by spotted and largemouth bass in Lake Martin, Alabama at station D2 in DCE from 1994–2004.

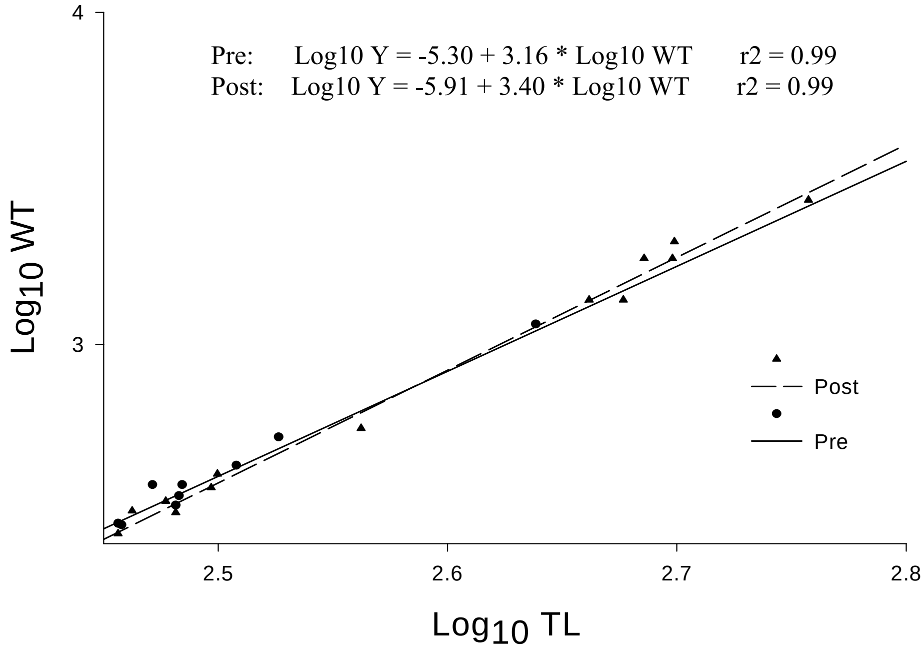

Figure 16. Plot of Weight:Length relations of largemouth bass sampled during the fall in DCE from 1994–2004. Solid and dashed lines represent the regression lines for Pre and Post treatments, respectively.

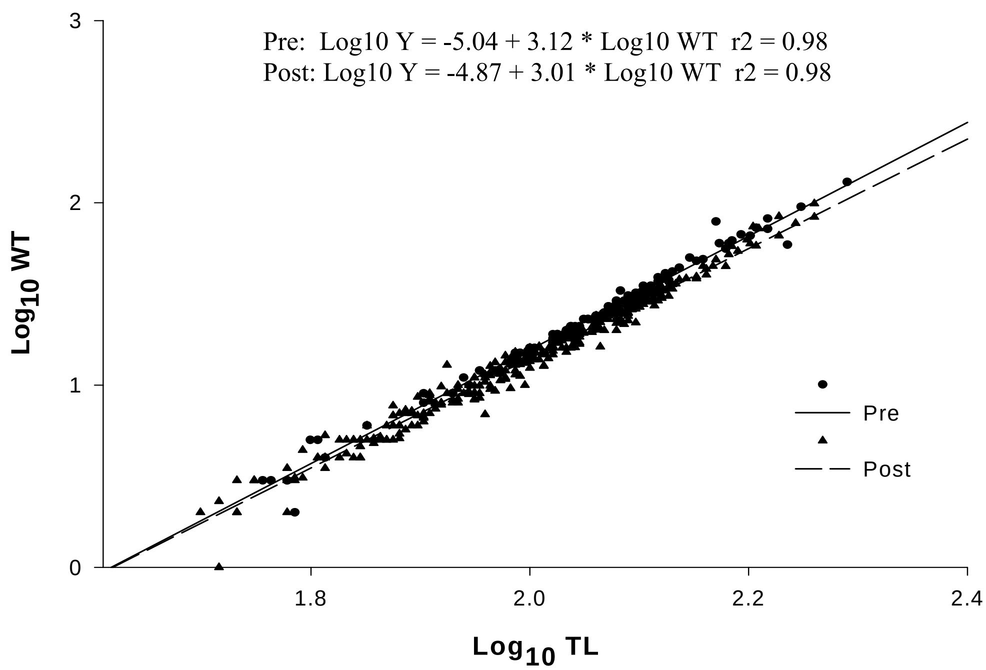

Figure 17. Plot of Weight:Length relations of bluegill sunfish sampled during the fall in ECE from 1994–2004. Solid and dashed lines represent the regression lines for Pre and Post treatments, respectively.

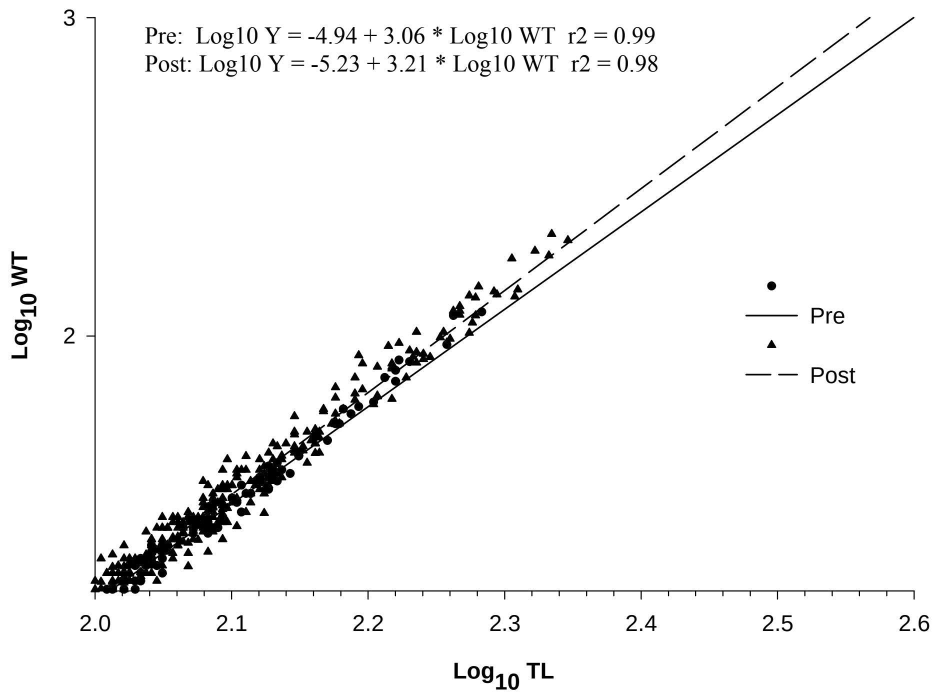

Figure 18. Plot of Weight:Length relations of bluegill sunfish sampled in DCE from 1994–2004. Solid and dashed lines represent the regression lines for Pre and Post treatments, respectively.

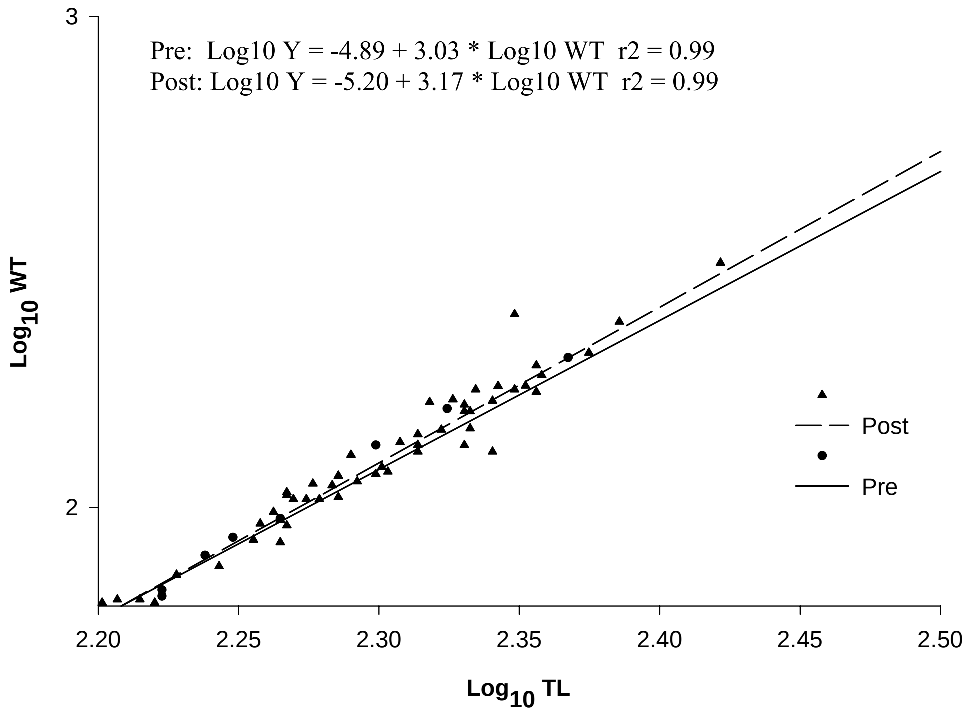

Figure 19. Plot of Weight:Length relations of redear sunfish sampled in DCE from 1994–2004. Solid and dashed lines represent the regression lines for Pre and Post treatments, respectively.

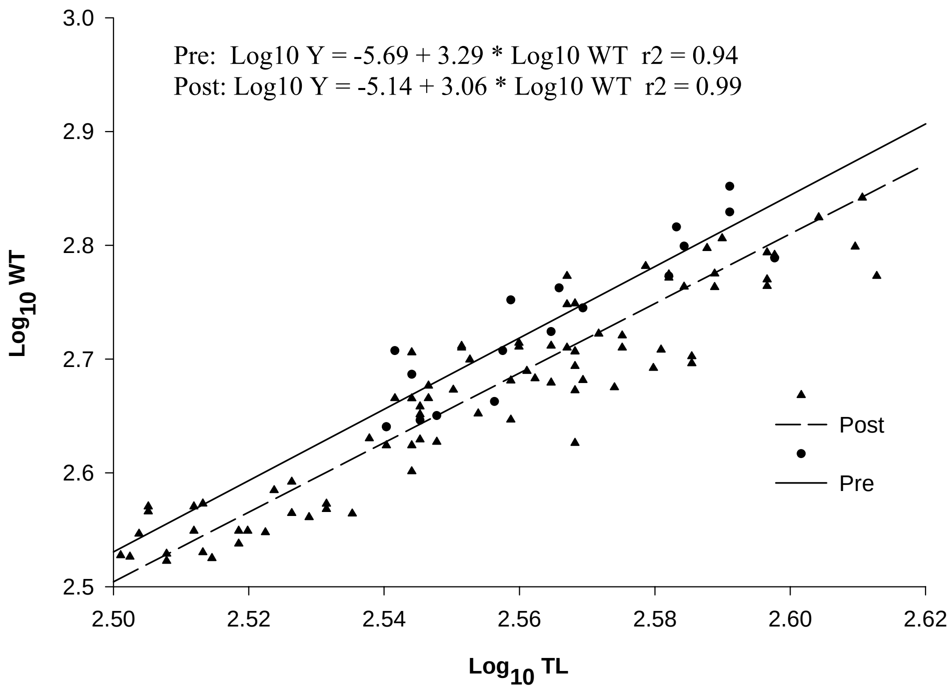

Figure 20. Plot of Weight:Length relations of blacktail redhorse sampled in ECE from 1994–2004. Solid and dashed lines represent the regression lines for Pre and Post treatments, respectively.

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APPENDICES

Appendix I. Summary of fish species and samples sizes collected by electrofishing in ECE and DCE in Lake Martin, Alabama during the sampling period 1994–2004.

Family	Species	Dennis Creek Embayment		Elkahatchee Creek Embayment	
		Pre	Post	Pre	Post
Clupeidae					
	Gizzard Shad (*Dorosoma cepedianum*)	…	17	4	16
	Threadfin Shad (*Dorosoma petenense*)	171	190	95	382
Cyprinidae					
	Blacktail Shiner (*Cyprinella venusta*)	8	77	13	80
	Common Carp (*Cyprinus carpio*)	…	10	…	8
	Weed Shiner (*Notropis texanus*)	2	80	11	58
	Pugnose Minnow (*Opsopoeodus emiliae*)	…	1	…	3
	Bullhead Minnow (*Pimephales vigilax*)	…	3	…	2
Catostomidae					
	Spotted Sucker (*Minytrema melanops*)	…	3	1	…
	Blacktail Redhorse (*Moxostoma poecilurum*)	12	179	16	212
Ictaluridae					
	Channel Catfish (*Ictalurus punctatus*)	…	33	1	11
	Flathead Catfish (*Pylodictis olivaris*)	…	2	2	13
Moronidae					
	White Bass (*Morone chrysops*)	…	…	…	…
	Hybrid Striped Bass (*M. chrysops x saxatilis*)	…	1	…	…
Centrarchidae					
	Redbreast Sunfish (Lepomis auritus)	7	13	1	6
	Green Sunfish (*Lepomis cyanellus*)	5	17	4	73
	Warmouth (*Lepomis gulosus*)	3	3	3	33
	Bluegill (*Lepomis macrochirus*)	166	553	297	707
	Longear Sunfish (*Lepomis megalotis*)	…	1	5	52
	Redear Sunfish (*Lepomis microlophus*)	19	115	3	86
	Spotted Bass (*Micropterus punctulatus*)	19	107	16	136
	Largemouth Bass (*Micropterus salmoides*)	18	35	24	85
	White Crappie (*Pomoxis annularis*)	…	1	…	…
	Black Crappie (*Pomoxis nigromaculatus*)	…	150	68	30
Percidae					
	Mobile Logperch (*Percina kathae*)	4	9	2	5
	Yellow Perch (*Perca flavescens*)	…	1	…	…

Appendix II. Summary of fish species and sample sizes collected by sinking gill nets in ECE and DCE in Lake Martin, Alabama during the sampling period 1994–2004.

Family	Species	Dennis Creek Embayment		Elkahatchee Creek Embayment	
		Pre	Post	Pre	Post
Clupeidae					
	Gizzard Shad (*Dorosoma cepedianum*)	11	36	5	39
	Threadfin Shad (*Dorosoma petenense*)	…	…	…	…
Cyprinidae		…	…	…	…
	Blacktail Shiner (*Cyprinella venusta*)	…	…	…	…
	Common Carp (*Cyprinus carpio*)	…	…	…	1
	Weed Shiner (*Notropis texanus*)	…	…	…	…
	Pugnose Minnow (*Opsopoeodus emiliae*)	…	…	…	…
	Bullhead Minnow (*Pimephales vigilax*)	…	…	…	…
Catostomidae					
	Spotted Sucker (*Minytrema melanops*)	…	8	5	20
	Blacktail Redhorse (*Moxostoma poecilurum*)	34	25	7	40
Ictaluridae					
	Channel Catfish (*Ictalurus punctatus*)	4	10	…	15
	Flathead Catfish (*Pylodictis olivaris*)	1	…	…	2
Moronidae					
	White Bass (*Morone chrysops*)	26	25	30	25
	Hybrid Striped Bass (*M. chrysops x saxatilis*)	…	2	…	12
Centrarchidae					
	Redbreast Sunfish (Lepomis auritus)	…	…	…	…
	Green Sunfish (*Lepomis cyanellus*)	…	…	…	…
	Warmouth (*Lepomis gulosus*)	…	…	…	…
	Bluegill (*Lepomis macrochirus*)	1	…	…	…
	Longear Sunfish (*Lepomis megalotis*)	…	…	…	…
	Redear Sunfish (*Lepomis microlophus*)	1	5	…	5
	Spotted Bass (*Micropterus punctulatus*)	36	21	6	19
	Largemouth Bass (*Micropterus salmoides*)	11	4	8	29
	White Crappie (*Pomoxis annularis*)	…	…	…	…
	Black Crappie (*Pomoxis nigromaculatus*)	43	17	64	10
Percidae					
	Mobile Logperch (*Percina kathae*)	…	…	…	…
	Yellow Perch (*Perca flavescens*)	…	…	…	…

Appendix III. Benthic invertebrates collected from Hester-Dendy multiplate samplers during the summer season in 1994 in the ECE in Lake Martin, Alabama.

Taxon	FFG	Replicates A	B	C	Total	Mean / 0.09 m^2
AQUATIC INSECTS						
Diptera						
Chironomidae						
Ablabesmyia spp.	P	27	14	37	78	289
Dicrotendipes spp.	GC	541	98	88	727	2693
Dicrotendipes tritomus	FC	8	0	0	8	30
Glyptotendipes spp.	FC	162	19	81	262	970
Nanocladius spp.	GC	0	0	7	7	26
Nilothauma spp.	GC	27	0	0	27	100
Tribelos spp.	GC	27	9	7	43	159
Odonata						
Coenagrionidae						
Argia spp.	P	0	0	2	2	7
Trichoptera						
Polycentropodidae						
Cyrnellus fraternus	FC	664	56	72	792	2933
Unidentified larvae	FC	32	16	8	56	207
Trichoptera						
Unidentified pupae		8	0	0	8	30
OTHER AQUATIC INVERTEBRATES						
Cladocera	FC	0	6	0	6	22
Copepoda						
Cyclopoida		8	4	0	12	44
Oligochaeta						
Naididae	GC	0	4	16	20	74
Ostracoda		0	2	0	2	7
Total No. of Organisms		1504	228	318	2050	
Total No. Taxa / Sampler		7	9	8		
Density (No./m^2) / Sampler		16711	2533	3533		7593

Appendix IV. Benthic invertebrates collected from Hester-Dendy multiplate samplers during the fall season in 1994 in the ECE in Lake Martin, Alabama.

Taxon	FFG	Replicate A	B	C	Total	Mean / 0.09 m^2
AQUATIC INSECTS						
Diptera						
Ceratopogonidae	P	0	4	0	4	15
Chironomidae						
Ablabesmyia annulata gp.	P	3	0	0	3	11
Ablabesmyia spp.	P	5	7	8	20	74
Asheum beckae	GC	8	17	0	25	93
Cladopelma spp.	GC	3	0	0	3	11
Dicrotendipes spp.	GC	5	3	6	14	52
Glyptotendipes spp.	FC	33	41	26	100	370
Nilothauma spp.	GC	3	0	0	3	11
Pseudochironomus spp.	GC	3	14	14	31	115
Tribelos spp.	GC	15	14	6	35	130
Unidentified pupae		0	2	2	4	15
Ephemeroptera						
Baetidae						
Centroptilum spp.	GC	2	0	0	2	7
Odonata						
Coenagrionidae						
Argia spp.	P	0		2	2	7
Trichoptera						
Polycentropodidae						
Cyrnellus fraternus	FC	50	48	78	176	652
Polycentropus spp.	FC	0	2	0	2	7
Unidentified larvae	FC	42	34	66	142	526
Trichoptera						
Unidentified larvae		8	2	14	24	89
OTHER AQUATIC INVERTEBRATES						
Cladocera	FC	90	14	16	120	444
Copepoda						
Cyclopoida		2	0	0	2	7
Oligochaeta						
Oligochaeta	GC	2	2	6	10	37
Total No. of Organisms		274	204	244	722	
Total No. Taxa / Sampler		14	11	10		
Density (No./m^2) / Sampler		3044	2267	2711		2674

Appendix V. Benthic invertebrates collected from Hester-Dendy multiplate samplers during the summer season of 1994 in the DCE in Lake Martin, Alabama.

Taxon	FFG	Replicates A	B	C	Total	Mean / 0.09 m^2
AQUATIC INSECTS						
Diptera						
Ceratopogonidae	P	0	4	2	6	22
Chironomidae						
Ablabesmyia spp.	P	17	11	24	52	193
Asheum beckae	GC	14	8	12	34	126
Chironomini	GC	3	0	0	3	11
Dicrotendipes spp.	GC	20	46	33	99	367
Glyptotendipes spp.	FC	20	8	21	49	181
Labrundinia spp.	P	3	0	0	3	11
Nanocladius spp.	GC	3	0	0	3	11
Parachironomus arcuatus gp.	P	3	3	0	6	22
Polypedilum spp.	SH	14	0	0	14	52
Pseudochironomus spp.	GC	3	0	0	3	11
Unidentified pupae		4	0	2	6	22
Odonata						
Coenagrionidae						
Argia spp.	P	0	0	2	2	7
Trichoptera						
Polycentropodidae						
Cyrnellus fraternus	FC	66	32	62	160	593
Unidentified larvae	FC	28	32	40	100	370
Trichoptera						
Unidentified pupae		0	2	4	6	22
Unidentified larvae		0	0	4	4	15

(table continues)

Appendix V. (continued)

		Replicates				Mean
Taxon	FFG	A	B	C	Total	/ 0.09 m^2
OTHER AQUATIC INVERTEBRATES						
Cladocera	FC	4	8	0	12	44
Oligochaeta						
Naididae						
Pristina spp.	GC	6	0	0	6	22
Oligochaeta	GC	4	12	4	20	74
Ostracoda		4	0	2	6	22
Total No. of Organisms		216	166	212	594	
Total No. Taxa / Sampler		15	9	10		
Density (No./m^2) / Sampler		2400	1844	2356		2200

Appendix VI. Benthic invertebrates collected from Hester-Dendy multiplate samplers during the fall season in 1994 in the DCE in Lake Martin, Alabama.

Taxon	FFG	Replicate A	B	C	Total	Mean / 0.09 m^2
AQUATIC INSECTS						
Diptera						
Chironomidae						
Ablabesmyia spp.	P	15	1	0	16	59
Asheum beckae	GC	15	3	4	22	81
Chironomini	GC	0	0	1	1	4
Dicrotendipes spp.	GC	12	10	7	29	107
Glyptotendipes spp.	FC	41	26	21	88	326
Labrundinia spp.	P	3	0	0	3	11
Nilothauma spp.	GC	0	1	0	1	4
Pseudochironomus spp.	GC	0	0	1	1	4
Tribelos spp.	GC	3	0	1	4	15
Unidentified pupae		2	2	1	5	19
Ephemeroptera						
Caenidae						
Caenis spp.	GC	0	1	0	1	4
Odonata						
Coenagrionidae						
Argia spp.	P	6	1	1	8	30
Trichoptera						
Polycentropodidae						
Cyrnellus fraternus	FC	40	37	36	113	419
Unidentified larvae	FC	16	17	20	53	196
Trichoptera						
Unidentified larvae		10	7	5	22	81
OTHER AQUATIC INVERTEBRATES						
Cladocera	FC	54	0	0	54	200
Copepoda	GC	2	0	0	2	7
Oligochaeta						
Naididae						
Pristina spp.	GC	12	2	2	16	59
Oligochaeta	GC	0	0	2	2	7
Ostracoda		0	2	2	4	15
Total No. of Organisms		231	110	104	445	
Total No. Taxa / Sampler		11	10	11		
Density (No./m^2) / Sampler		2567	1222	1156		1648

Appendix VII. Benthic invertebrates collected from Hester-Dendy multiplate samplers during the summer season in 2002 in the ECE in Lake Martin, Alabama.

Taxon	FFG	Replicates A	B	C	Total	Mean / 0.09 m^2
AQUATIC INSECTS						
Diptera						
Chironomidae						
Ablabesmyia spp.	P	16	200	17	233	863
Dicrotendipes simpsoni	FC	114	67	51	232	859
Dicrotendipes spp.	GC	0	100	34	134	496
Glyptotendipes spp.	FC	244	467	324	1035	3833
Nanocladius alternanthera	GC	16	0	0	16	59
Polypedilum beckae		81	167	68	316	1170
Tribelos fuscicorne	GC	16	0	17	33	122
Unidentified pupae		8	0	0	8	30
Ephemeroptera						
Caenidae						
Caenis spp.	GC	0	8	0	8	30
Odonata						
Coenagrionidae						
Argia spp.	P	8	0	0	8	30
Trichoptera						
Hydroptilidae						
Unidentified larvae	PI	8	0	0	8	30
Polycentropodidae						
Cyrnellus fraternus	FC	120	40	192	352	1304
OTHER AQUATIC INVERTEBRATES						
Cladocera	FC	48	16	32	96	356
Nematoda		40	0	16	56	207
Oligochaeta						
Naididae	GC	64	72	176	312	1156
Ostracoda		8	8	0	16	59
Turbellaria	P	0	8	0	8	30
Total No. of Organisms		791	1153	927		
Total No. Taxa / Sampler		13	10	9		
Density (No./m^2) / Sampler		8789	12811	10300		10633

Appendix VIII. Benthic invertebrates collected from Hester-Dendy multiplate samplers during the summer season in 2002 in the DCE in Lake Martin, Alabama.

Taxon	FFG	Replicates A	B	C	Total	Mean / 0.09 m^2
AQUATIC INSECTS						
Diptera						
Chironomidae						
Ablabesmyia annulata gp.	P	0	0	1	1	4
Ablabesmyia spp.	P	2	0	0	2	7
Dicrotendipes simpsoni	FC	2	3	4	9	33
Dicrotendipes spp.	GC	4	5	0	9	33
Glyptotendipes spp.	FC	19	7	4	30	111
Nanocladius alternanthera	GC	2	0	2	4	15
Polypedilum beckae		17	1	8	26	96
Tanytarsus sp. C	FC	2	0	0	2	7
Tribelos fuscicorne	GC	9	3	0	12	44
Ephemeroptera						
Heptageniidae						
Unidentified nymph	SC	1	0	0	1	4
Odonata						
Coenagrionidae						
Argia spp.	P	1	1	0	2	7
Corduliidae						
Unidentified nymph	P	0	1	0	1	4
Trichoptera						
Hydroptilidae						
Ochrotrichia spp.	PI	0	4	0	4	15
Unidentified larvae	PI	1	9	2	12	44
Polycentropodidae						
Cyrnellus fraternus	FC	6	7	9	22	81
OTHER AQUATIC INVERTEBRATES						
Cladocera	FC	5	8	8	21	78
Copepoda	GC	0	3	0	3	11
Nematoda		7	0	0	7	26
Oligochaeta						
Naididae	GC	90	24	31	145	537
Ostracoda		4	21	0	25	93
Turbellaria	P	59	20	5	84	311
Total No. of Organisms		231	117	74	422	
Total No. Taxa / Sampler		16	13	10		
Density (No./m^2) / Sampler		2567	1300	822		1563

Appendix IX. Benthic invertebrates collected from Hester-Dendy multiplate samplers during the summer season in 2004 in the ECE in Lake Martin, Alabama.

Taxon	FFG	Replicates A	B	C	Total	Mean / 0.09 m^2
AQUATIC INSECTS						
Diptera						
Chironomidae						
Ablabesmyia annulata gp.	P	13	1	40	54	200
Ablabesmyia spp.	P	4	0	0	4	15
Chironomus spp.	SH	8	0	27	35	130
Cricotopus/Orthocladius	GC	4	0	0	4	15
Dicrotendipes neomodestus	FC	4	1	0	5	19
Dicrotendipes simpsoni	FC	21	1	54	76	281
Dicrotendipes spp.	GC	4	0	13	17	63
Endochironomus spp.	SH	0	1	0	1	4
Glyptotendipes spp.	FC	25	22	134	181	670
Labrundinia spp.	P	4	0	0	4	15
Polypedilum beckae		29	6	121	156	578
Pseudochironomus spp.	GC	4	1	0	5	19
Tanytarsus sp. P	FC	4	0	0	4	15
Tribelos fuscicorne	GC	0	0	13	13	48
Unidentified pupae		5	2	4	11	41
Ephemeroptera						
Caenidae						
Caenis spp.	GC	1	0	6	7	26
Heptageniidae						
Stenacron spp.	SC	2	1	1	4	15
Odonata						
Coenagrionidae						
Argia spp.	P	0	0	2	2	7
Trichoptera						
Hydroptilidae						
Unidentified larvae	PI	0	1	0	1	4
Polycentropodidae						
Cyrnellus fraternus	FC	10	5	19	34	126

(appendix continues)

Appendix IX. Benthic invertebrates, continued.

Taxon	FFG	Replicates A	B	C	Total	Mean / 0.09 m^2
OTHER AQUATIC INVERTEBRATES						
Cladocera	FC	26	2	174	202	748
Copepoda		0	0	4	4	15
Oligochaeta						
Naididae	GC	22	76	10	108	400
Ostracoda		0	0	12	12	44
Pelecypoda						
Sphaeriidae	FC	5	0	1	6	22
Total No. of Organisms		195	120	635	950	
Total No. Taxa / Sampler		16	12	15		
Density (No./m^2) / Sampler		2167	1333	7056		3519

Appendix X. Benthic invertebrates collected from Hester-Dendy multiplate samplers during the fall season in 2004 in the ECE in Lake Martin, Alabama.

		Replicates				Mean
Taxon	FFG	A	B	C	Total	/0.09 m^2
AQUATIC INSECTS						
Diptera						
Chironomidae						
Chironomus spp.	SH	1	1	4	6	22
Dicrotendipes neomodestus	FC	1	0	1	2	7
Dicrotendipes simpsoni	FC	0	0	2	2	7
Dicrotendipes spp.	GC	0	1	1	2	7
Endochironomus spp.	SH	0	0	1	1	4
Glyptotendipes spp.	FC	11	3	12	26	96
Polypedilum beckae		5	1	8	14	52
Unidentified pupae		1	0	0	1	4
Ephemeroptera						
Caenidae						
Brachycercus spp.	GC	1	0	0	1	4
Heptageniidae						
Stenacron spp.	SC	0	1	1	2	7
Trichoptera						
Hydroptilidae						
Unidentified larvae	PI	1	0	0	1	4
Polycentropodidae						
Cyrnellus fraternus	FC	2	2	3	7	26
OTHER AQUATIC INVERTEBRATES						
Cladocera	FC	347	245	102	694	2570
Gastropoda						
Pleuroceridae						
Elimia spp.	SC	0	0	1	1	4
Oligochaeta						
Naididae	GC	1	0	1	2	7
Total No. of Organisms		371	254	137	762	
Total No. Taxa / Sampler		9	7	11		
Density (No./m^2) / Sampler		4122	2822	1522		2822

Appendix XI. Benthic invertebrates collected from Hester-Dendy multiplate samplers during the summer season in 2004 in the DCE in Lake Martin, Alabama.

		Replicates				Mean
Taxon	FFG	A	B	C	Total	/ 0.09 m^2
AQUATIC INSECTS						
Diptera						
Ceratopogonidae						
Unidentified larvae	P	1	0	0	1	4
Chironomidae						
Ablabesmyia annulata gp.	P	0	1	4	5	19
Chironomus spp.	SH	2	2	0	4	15
Dicrotendipes neomodestus	FC	0	2	4	6	22
Dicrotendipes simpsoni	FC	11	3	4	18	67
Endochironomus spp.	SH	2	2	0	4	15
Glyptotendipes spp.	FC	10	9	39	58	215
Nanocladius (N.) *alternanthera*		8	0	0	8	30
Parachironomus carinatus	GC	0	1	12	13	48
Polypedilum beckae		19	7	54	80	296
Polypedilum spp.	SH	2	0	0	2	7
Tribelos fuscicorne	GC	2	0	0	2	7
Unidentified pupae		1	3	3	7	26
Ephemeroptera						
Caenidae						
Caenis spp.	GC	2	14	1	17	63
Heptageniidae						
Stenacron spp.	SC	0	0	1	1	4
Odonata						
Coenagrionidae						
Argia spp.	P	1	1	4	6	22
Trichoptera						
Hydroptilidae						
Ochrotrichia spp.	SC	11	0	2	13	48
Unidentified larvae	PI	2	0	0	2	7
Polycentropodidae						
Cyrnellus fraternus	FC	2	2	4	8	30

Appendix XI. Benthic invertebrates, continued.

Taxon	FFG	Replicates A	B	C	Total	Mean / 0.09 m^2
OTHER AQUATIC INVERTEBRATES						
Cladocera	FC	16	35	9	60	222
Copepoda		0	0	3	3	11
Oligochaeta						
Naididae	GC	70	33	43	146	541
Ostracoda		45	2	44	91	337
Pelecypoda						
Sphaeriidae	FC	0	0	1	1	4
Total No. of Organisms		207	117	232	556	
Total No. Taxa / Sampler		15	15	16		
Density (No./m^2) / Sampler		2300	1300	2578		2059

Appendix XII. Benthic invertebrates collected from Hester-Dendy multiplate samplers during the fall season in 2004 in the DCE in Lake Martin, Alabama.

		Replicates			Mean
Taxon	FFG	A	B	Total	/ 0.09 m^2
AQUATIC INSECTS					
Diptera					
Chironomidae					
Dicrotendipes neomodestus	FC	10	0	10	56
Endochironomus spp.	SH	0	1	1	6
Glyptotendipes spp.	FC	7	2	9	50
Polypedilum beckae		1	0	1	6
Trichoptera					
Polycentropodidae					
Cyrnellus fraternus	FC	2	1	3	17
OTHER AQUATIC INVERTEBRATES					
Cladocera					
Daphnidae	FC	475	150	625	3472
Oligochaeta					
Naididae	GC	1	0	1	6
Total No. of Organisms		496	154	650	
Total No. Taxa / Sampler		6	4	7	
Density (No./m^2) / Sampler		5511	1711	7222	4815

Appendix XIII. Benthic invertebrates collected from dredge samples during the summer season in 1994 in the ECE in Lake Martin, Alabama.

Taxon	FFG	Replicates A	B	C	Total	Mean / 0.0225 m^2
AQUATIC INSECTS						
Diptera						
Chaoboridae						
Chaoborus spp.	P	26	26	30	82	1215
Chaoborus spp. pupae		1	2	0	3	44
Chironomidae						
Coelotanypus spp.	P	4	3	5	12	178
Parachironomus arcuatus gp.	P	0	1	0	1	15
Tanypus spp.	P	2	0	6	8	119
OTHER AQUATIC INVERTEBRATES						
Copepoda						
Cyclopoida		3	2	0	5	74
Pelecypoda						
Sphaeriidae						
Sphaerium spp.	FC	1	5	3	9	133
Total No. of Organisms		37	39	44	120	
Total No. Taxa / Sampler		5	5	4		
Density (No./m^2) / Sampler		1644	1733	1956		1778

Appendix XIV. Benthic invertebrates collected from dredge samples during the fall season in 1994 in the ECE in Lake Martin, Alabama.

Taxon	FFG	Replicate A	Replicate B	Replicate C	Total	Mean / 0.0225 m^2
AQUATIC INSECTS						
Diptera						
Chaoboridae						
Chaoborus spp.	P	36	58	44	138	2044
Chironomidae						
Cladopelma spp.	GC	0	0	1	1	15
Coelotanypus spp.	P	15	37	29	81	1200
Microchironomus spp.	GC	0	1	0	1	15
Procladius sp.	P	0	0	1	1	15
Unidentified larvae	GC	2	0	0	2	30
OTHER AQUATIC INVERTEBRATES						
Copepoda						
Cyclopoida		8	0	4	12	178
Oligochaeta						
Oligochaeta	GC	1	0	0	1	15
Pelecypoda						
Corbiculidae						
Corbicula fluminea	FC	0	0	1	1	15
Sphaeriidae						
Musculium spp.	FC	0	4	0	4	59
Total No. of Organisms		62	100	80	242	
Total No. Taxa / Sampler		4	4	6		
Density (No./m^2) / Sampler		2756	4444	3556		3585

Appendix XV. Benthic invertebrates collected from dredge samples during the summer season of 1994 in the DCE in Lake Martin, Alabama.

Taxon	FFG	Replicates A	Replicates B	Replicates C	Total	Mean / 0.0225 m^2
AQUATIC INSECTS						
Diptera						
Ceratopogonidae	P	1	1	0	2	30
Chaoboridae						
Chaoborus spp.	P	63	50	24	137	2030
Chironomidae						
Chironomus spp.	SH	2	5	0	7	104
Coelotanypus spp.	P	3	4	0	7	104
Procladius sp.	P	2	2	2	6	89
Tanypus spp.	P	2	3	0	5	74
OTHER AQUATIC INVERTEBRATES						
Copepoda						
Cyclopoida		0	1	22	23	341
Nematoda		1	0	1	2	30
Oligochaeta						
Naididae						
Pristina spp.						
Tubificidae	GC	7	0	1	8	119
Branchiura sowerbyi	GC	1	6	5	12	178
Pelecypoda						
Sphaeriidae						
Sphaerium spp.	FC	0	1	0	1	15
Total No. of Organisms		82	73	55	210	
Total No. Taxa / Sampler		9	9	6		
Density (No./m^2) / Sampler		3644	3244	2444		3111

Appendix XVI. Benthic invertebrates collected from dredge samples during the fall season in 1994 in the DCE in Lake Martin, Alabama.

Taxon	FFG	Replicate A	B	C	Total	Mean / 0.0225 m^2
AQUATIC INSECTS						
Diptera						
Chaoboridae						
Chaoborus spp.	P	10	8	33	51	756
Chironomidae						
Ablabesmyia annulata gp.	P	0	1	0	1	15
Chironomus spp.	SH	0	1	2	3	44
Coelotanypus spp.	P	5	9	7	21	311
Glyptotendipes spp.	FC	0	0	1	1	15
Procladius sp.	P	0	0	1	1	15
Tanytarsus spp.	FC	0	1	0	1	15
Unidentified pupae		0	1	0	1	15
Ephemeroptera						
Ephemeridae						
Hexagenia spp.	GC	0	1	0	1	15
OTHER AQUATIC INVERTEBRATES						
Cladocera	FC	0	0	1	1	15
Copepoda						
Cyclopoida		0	1	0	1	15
Oligochaeta						
Tubificidae						
Branchiura sowerbyi	GC	0	0	2	2	30
Unidentified	GC	0	1	2	3	44
Pelecypoda						
Corbiculidae						
Corbicula fluminea	FC	0	1	0	1	15
Total No. of Organisms		15	25	49	89	
Total No. Taxa / Sampler		2	8	7		
Density (No./m^2) / Sampler		667	1111	2178		1319

Appendix XVII. Benthic invertebrates collected from dredge samples during the summer season in 2002 in the ECE in Lake Martin, Alabama.

		Replicates				Mean
Taxon	FFG	A	B	C	Total	/ 0.0225 m^2
AQUATIC INSECTS						
Diptera						
Chaoboridae						
Chaoborus spp.	P	105	137	116	358	5304
Chironomidae						
Chironomus spp.	SH	1	0	0	1	15
Trichoptera						
Leptoceridae						
Unidentified larvae	GC	1	0	0	1	15
OTHER AQUATIC INVERTEBRATES						
Cladocera	FC	0	2	1	3	44
Copepoda	GC	0	1	0	1	15
Nematoda		1	0	0	1	15
Oligochaeta						
Tubificidae						
Branchiura sowerbyi	GC	0	5	3	8	119
Unidentified	GC	0	1	0	1	15
Pelecypoda						
Sphaeriidae						
Unidentified	FC	2	19	0	21	311
Total No. of Organisms		110	165	120		
Total No. Taxa / Sampler		5	5	3		
Density (No./m^2) / Sampler		4889	7333	5333		5852

Appendix XVIII. Benthic invertebrates collected from dredge samples during the summer season in 2002 in the DCE in Lake Martin, Alabama.

Taxon	FFG	Replicates A	B	C	Total	Mean / $0.0225m^2$
AQUATIC INSECTS						
Diptera						
Chaoboridae						
Chaoborus spp.	P	162	82	130	374	5541
Chironomidae						
Coelotanypus tricolor		0	4	1	5	74
OTHER AQUATIC INVERTEBRATES						
Copepoda	GC	7	7	13	27	400
Oligochaeta						
Tubificidae						
Branchiura sowerbyi	GC	1	1	1	3	44
Unidentified	GC	0	2	3	5	74
Total No. of Organisms		170	96	148	414	
Total No. Taxa / Sampler		3	4	4		
Density (No./m^2) / Sampler		7556	4267	6578		6133

Appendix XIX. Benthic invertebrates collected from dredge samples during the fall season in 2004 in the ECE in Lake Martin, Alabama.

		Replicate				Mean
Taxon	FFG	A	B	C	Total	/ 0.0225 m^2
AQUATIC INSECTS						
Diptera						
Chaoboridae						
Chaoborus spp.	P	59	20	8	87	1289
Chironomidae						
Chironomus spp.	SH	6	25	14	45	667
Coelotanypus spp.	P	1	0	1	2	30
Cryptochironomus spp.	P	0	3	2	5	74
Procladius bellus var. I	P	5	2	6	13	193
OTHER AQUATIC INVERTEBRATES						
Cladocera	FC	1	1	1	3	44
Copepoda		0	3	0	3	44
Oligochaeta						
Tubificidae						
Branchiura sowerbyi	GC	0	5	10	15	222
Unidentified	GC	1	0	0	1	15
Pelecypoda						
Sphaeriidae	FC	1	0	0	1	15
Total No. of Organisms		74	59	42	175	
Total No. Taxa / Sampler		6	7	7		
Density (No./m^2) / Sampler		3289	2622	1867		2593

Appendix XX. Benthic invertebrates collected from dredge samples during the summer season in 2004 in the ECE in Lake Martin, Alabama.

Taxon	FFG	Replicate A	Replicate B	Replicate C	Total	Mean / 0.0225 m^2
AQUATIC INSECTS						
Diptera						
Chaoboridae						
Chaoborus spp.	P	70	60	42	172	2548
Chironomidae						
Chironomus spp.	SH	0	0	1	1	15
Coelotanypus spp.	P	1	0	0	1	15
OTHER AQUATIC INVERTEBRATES						
Pelecypoda						
Sphaeriidae	FC	1	1	1	3	44
Total No. of Organisms		72	61	44	177	
Total No. Taxa / Sampler		3	2	3		
Density (No./m^2) / Sampler		3200	2711	1956		2622

Appendix XXI. Benthic invertebrates collected from dredge samples during the summer season in 2004 in the DCE in Lake Martin, Alabama.

Taxon	FFG	Replicates A	B	C	Total	Mean / 0.0225 m^2
AQUATIC INSECTS						
Diptera						
Chaoboridae						
Chaoborus spp.	P	79	166	142	387	5733
Chironomidae						
Chironomus spp.	SH	0	0	1	1	15
Glyptotendipes spp.	FC	0	1	0	1	15
Nanocladius (N.) *alternanthera*		1	0	0	1	15
Diptera						
Unidentified pupae		1	1	0	2	30
OTHER AQUATIC INVERTEBRATES						
Copepoda		1	0	0	1	15
Oligochaeta						
Tubificidae						
Branchiura sowerbyi	GC	0	0	2	2	30
Total No. of Organisms		82	168	145	395	
Total No. Taxa / Sampler		3	2	3		
Density (No./m^2) / Sampler		3644	7467	6444		5852

Appendix XXII. Benthic invertebrates collected from dredge samples during the fall season in 2004 in the DCE in Lake Martin, Alabama.

		Replicates				Mean
Taxon	FFG	A	B	C	Total	/ 0.0225 m^2
AQUATIC INSECTS						
Diptera						
Chaoboridae						
Chaoborus spp.	P	46	17	21	84	1244
Chironomidae						
Chironomus spp.	SH	9	3	6	18	267
Coelotanypus spp.	P	3	2	1	6	89
Procladius (Holotanypus) sp.	P	0	0	1	1	15
OTHER AQUATIC INVERTEBRATES						
Cladocera						
Daphnidae	FC	2	0	0	2	30
Copepoda	(blank)	0	2	1	3	44
Oligochaeta						
Tubificidae						
Branchiura sowerbyi	GC	6	7	7	20	296
Unidentified	GC	0	0	6	6	89
Total No. of Organisms		66	31	43	140	
Total No. Taxa / Sampler		5	5	6		
Density (No./m^2) / Sampler		2933	1378	1911		2074