THE EFFECTS OF A MODIFIED FLOW REGIME ON FISH ASSEMBLAGES IN A

THIRD-ORDER PIEDMONT STREAM IN LEE COUNTY, AUBURN, ALABAMA

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THESIS ABSTRACT

THE EFFECTS OF A MODIFIED FLOW REGIME ON FISH ASSEMBLAGES IN A THIRD ORDER PIEDMONT STREAM IN LEE COUNTY, AUBURN, ALABAMA

James Edmund Gleason

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I examined the effects of a modified flow regime on fish assemblage condition over a 2-year period at two sites in Chewacla Creek, a third-order Piedmont stream (Lee County, AL). My goal was to assess the biotic condition of fish assemblages in the creek impacted by dewatered conditions that occurred prior to 2003 as a result of drought, no water releases from Lake Ogletree, and quarry mining in the watershed. The two Chewacla sites were compared with a relatively undisturbed reference site, Cane Creek (Cane-1), another Piedmont stream in Lee County. Water quality and fishes were sampled once per year at all three sites during low-flow conditions in the fall of 2004 and 2005. In addition, a physical habitat assessment at each site was conducted in the fall of 2006. The principal method used in this study for comparison of fish assemblages across

sites was the Index of Biotic Integrity (IBI). Differences across sites in water quality, physical habitat quality and fish community health were compared.

Results showed few differences across sites in water quality and physical habitat once continuous flows were reestablished in the study reach in December of 2003. IBI ratings for Cane-1were good to excellent; thus, it compared well with other Piedmont reference streams in terms of fish assemblage condition. In contrast, IBI ratings for the two Chewacla Creek sites (CH-U and CH-D) rated poor to fair, indicating that fish assemblages at both sites were degraded and did not meet fish assemblage conditions present at Cane-1 in either 2004 or 2005. At CH-U, IBI ratings were poor both years suggesting no recovery of the fish assemblage from altered flow conditions that occurred periodically between 1999 and 2001. At CH-D, IBI ratings changed from poor to marginally fair in 2005 suggesting slight recovery of the fish assemblage from periodic dewatering of the stream channel that occurred at the site.

Differences in fish assemblage condition between Cane-1, CH-U, and CH-D were mostly attributable to differences in flow regime. Although water quality and physical habitat influenced the fish assemblages at CH-U and CH-D, a highly modified flow regime resulting from drought in previous years, mining in the watershed, and the presence of Lake Ogletree appeared to be most influential in structuring the fish assemblages at both sites during this study.

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I. INTRODUCTION

Most lotic waters in the world have been impacted by human activities, resulting in modification of their physical, chemical, and biological condition (Allan 1995). Humans have altered natural flow regimes by building dams, channelizing rivers and diverting water for flood control, navigation, irrigation, industrial and municipal water supply, and hydropower. Furthermore, major changes in the surrounding landscape, such as urbanization, agricultural practices, and deforestation have altered natural flow regimes resulting in degraded ecological integrity of lotic ecosystems (Mulholland and Lenat 1992).

Mining is a major anthropogenic factor affecting streams and rivers and its impact on streams and riparian areas in the United States and other regions of the world has been severe (Allan 1995). Within Alabama, mining mineral deposits for crushed stone, sand and gravel in or adjacent to streams is a common practice. For example, in Lee County quarry mining has been practiced intermittently since the early 1900s. Mining activities can have a number of detrimental impacts on streams including degrading water quality from increased sediment load (Kondolf 1994; Waters 1995), increasing stream bed bank erosion (Bull and Scott 1974; Sandecki 1989; Kondolf 1994; Pond 2004), and altering natural flow regimes and instream habitat through flow impediments; each of these activities can reduce overall biological diversity and production (Benke 1990; Lyttle 1993; Waters 1995; Brown et al. 1998).

Chewacla Creek, a third-order Piedmont stream in East Central Alabama, has been the subject of much interest in recent years, in particular the 1.5 mile reach between Lake Ogletree and Chewacla State Park in Lee County. This reach of Chewacla Creek is a regulated stream because of the presence of Lake Ogletree. Portions of the reach downstream of Lake Ogletree, the primary potable water supply for the city of Auburn, have been directly impacted by operation of a crushed rock quarry adjacent to the creek channel. Mining operations at the quarry resulted in sinkholes in reaches of the stream channel near the quarry (United States Fish and Wildlife Service 2003). In addition, periodic drought conditions since 1999 resulted in low lake levels and no water flow over the spillway into the downstream channel (USFWS 2003). Thus, the only water input from Lake Ogletree to the downstream channel during these dry periods was seepage below the base of the dam; however, this input of water was usually insufficient to maintain constant flows in the creek between the lake and the quarry. The combination of sinkholes and no outflow of water from the lake has resulted in dewatered reaches adjacent to the quarry during drought (Rex Griffin, Water and Sewer Director, personal communication). Furthermore, on several occasions between 1999 and 2001 the creekbed was so dry that only small, isolated pools were present (Cliff Webber, AU Fisheries, personal communication).

In 2003 legal action by local citizens resulted in a Chewacla Creek Safe Harbor Agreement between the citizens, the City of Auburn Water Board, Martin Marietta Materials, Inc., the Alabama Department of Conservation and Natural Resources (ADCNR), and the U.S. Fish and Wildlife Service. The agreement provided for establishment of minimum flows of 2.0 million gallons per day (MGD) in the stream just below the dam on Lake Ogtletree. Three requirements were included in the Chewacla Creek Safe Harbor Agreement (2003) to ensure that a flow of 2.0 MGD was maintained in the impacted reach. First, the City of Auburn Water Board (now the Water Resources Management Department) was required to siphon water from the lake at a depth of 1.5 m (5 ft) below the water surface to maintain a minimum flow of 2.0 MGD at the base of Lake Ogletree dam. Second, Martin Marietta was required to monitor and repair sinkholes in the channel near the quarry so that the minimum flow of 2.0 MGD released from the lake was maintained downstream. Third, Martin Marietta could be asked by the Water Board to pump water from the quarry pit to the lake to replenish water siphoned from the lake to maintain the minimum 2.0 MGD flow at the base of Lake Ogletree dam. Pumping water from the quarry pits to the lake rather than directly into Chewacla Creek would allow the settling of any sediment present in the quarry water and a moderation of pH levels. To implement these measures, Martin Marietta and the City of Auburn Water Board were required to monitor flow conditions continuously and conduct annual bioassessments in the impacted reach downstream of Lake Ogletree.

Concern for the survival of 3 threatened species of mussels listed in 1993 by the U.S. Fish and Wildlife Service: the fine-lined pocketbook *(Lampsilis altilis)*, southern clubshell *(Pleurobema decisum*) and ovate clubshell (*Pleurobema perovatum*) prompted a study of the stream and assessment of the mussel assemblage and of fish their host species, the largemouth bass *(Micropterus salmoides*). Females of these species release glochidia which are free-living parasitic larvae that attach to the gills of host fish (Wetzel 2001). Glochidia remain on the gills for several weeks where they metamorphose into juvenile mussels and then drop off the host and settle within the substrate (Wetzel 2001).

The purpose of the Safe Harbor Agreement (2003) was to provide constant flows in the previously dewatered reach of Chewacla Creek in hopes that the fine-lined pocketbook mussel would repopulate all sections of the reach. In addition, migration of largemouth basses into previously dewatered sections would increase the likelihood that southern and ovate clubshell mussels would repopulate the impacted reach.

Since 2003 cooperative agreements have been in place between AU Fisheries and the City of Auburn Water Resources Department to monitor water quality and stream flow and conduct annual bioassessments of the macroinvertebrate and mussel assemblages in the impacted reach. The Safe Harbor Agreement (2003) secured minimum flow and subsidence control requirements in sections of Chewacla Creek to enhance and expand available instream habitat for the three threatened mussel species, but it did not require assessment of the fish assemblage in the reach downstream from Lake Ogletree and adjacent to the quarry. This lack of monitoring of the fish assemblage left a critical gap in the assessment of the aquatic biota in Chewacla Creek as it recovered from the dewatering of the stream channel that occurred between 1999 and 2001 (Webber and Blevins 2000; 2001). In 2004, funding was obtained by AU Fisheries for a survey of the fish assemblages at two sites in Chewacla Creek through a United States Environmental Protection Agency (US EPA) grant for stream restoration. This survey included sampling of the fish fauna at 2 sites in Chewacla Creek, one just downstream of Lake Ogletree and the other further downstream adjacent to the quarry, to examine the impact of an altered flow regime on the resident fish assemblages. In addition, the fish fauna was sampled concurrently in a reference stream during the same 2-year period in Cane Creek, an unregulated third-order Piedmont stream.

This 2-year study allowed me to examine the extent of recovery of the fish assemblages at two sites in Chewacla Creek from an altered flow regime that occurred between 1999 and 2001 after consistent flows were reestablished in the reach downstream of Lake Ogletree. The main objective of this study was to compare fish assemblage condition in a reach recovering from altered flows with one exhibiting a natural flow regime. Specific questions were related to comparisons between Chewacla Creek and reference sites in terms of (1) differences in abundance, species richness and diversity, (2) altered trophic structure and function with an increase in tolerant species, and (3) an increase in dominance of certain species with lower evenness in Chewacla Creek relative to the reference site.

II. LITERATURE REVIEW

Stream velocity, depth, substrate, and instream cover are considered the most important physical habitat characteristics affecting distribution and abundance of fish species (Angermeier 1989). Stream alterations from human activity often disturb physical habitats, changing fish assemblage structure upstream, downstream, and within disturbed areas (Nunnally 1978; Schlosser 1990; Aadland 1993). Managing the integrity of stream water resources requires the characterization and careful analysis of physical, chemical, and biological components. Thus, resource managers often must quantify these changes in stream ecosystems, and distinguish between changes occurring naturally and those caused by anthropogenic impacts.

Frequent, long-term, and continuing anthropogenic impacts are inevitable in Piedmont streams in the southeastern United States because of continued landscape alteration associated with human population growth (Mulholland and Lenat 1992). However, stronger regulatory efforts by state and federal agencies utilizing sound biological principles can potentially offset the effects of physical habitat degradation on instream biological communities. Habitat degradation often results in changes in resident fish assemblages (Karr 1981; Schlosser 1990); thus, careful analysis of fish assemblage characteristics relative to the levels of disturbance can provide benchmarks guiding future habitat improvement programs for Piedmont streams.

Mining Impacts on Aquatic Biota

One of the most highly visible forms of land use change in Alabama is surface mining, in particular crushed rock mining (Alabama Department of Industrial Relations 2003). Crushed rock (aggregate) is one of the most important and highly demanded resources in the United States, having uses in nearly all commercial, industrial, and residential construction (Langer and Glanzman 1993). Aggregates are produced from sand and mineral deposits or from bedrock sources (Poulin et al. 2004). Behind fuel mineral mining, aggregate (non-fuel) mining is the second largest mining industry in the United States (Bull and Scott, 1974; Waters 1995). The amount of aggregate produced annually in the United States increased from a modest 58 million tons mined in 1980 to 2.3 billion tons in 1996 (Bolen 1997; Tepordei 1997). In Alabama, mining for aggregate is currently practiced in 21 counties, including Lee County (United States Geological Survey 2004). About 49,100 metric tons of crushed rock was produced in the state in 2004 with an economic value of about $290 million dolars (USGS 2004).

One of the most common methods of aggregate extraction in the United States is open pit (quarry) mining in the flood plains and terraces of small and medium-sized streams (Roell 1999). The impact of surface mining operations such as quarry mining on the surrounding landscape, stream hydrology and geomorphology, and aquatic communities can be profound (Starnes and Gasper 1996). Surface mining, particularly in the flood plains of streams, can cause temporary elimination of surface vegetation and alteration of surface topography (Starnes and Gasper 1996). Furthermore, mining can alter subsurface geologic structure and subsurface hydrologic regimes. The deep quarry pits that form often result in subsidence features (cracks, fissures, and sinkholes) in stream channels that can dewater reaches and divert flows (Starnes and Gasper 1996). Degraded water quality in aquatic systems impacted by aggregate mining usually occurs as a result of wastewater discharges into streams that increase concentrations of suspended solids as well as sedimentation in pools and riffles (Gammon 1970). Typically, heavy sedimentation and degraded stream channels are common in areas in which rock quarries occur (Roell 1999). Furthermore, streams located near rock quarries often exhibit altered flow regimes and loss of critical habitat (Roell 1999). Such negative effects can result in loss of sensitive fish species and ecological integrity. Changes may occur in the structure and function of fish assemblages through mining-related effects on growth, behavior, migration, and reproduction (Gammon 1970; Ahle and Jobsis 1996; Bell and Payne 1993). In particular, these effects on stream fishes may include reduced feeding efficiency and, thus, reduced growth, respiratory impairment, reduced tolerance to diseases and toxicants, and increased physiological stress (Waters 1995). Additionally, inorganic sediment and alterations in flow regime as a result of mining activities can reduce the reproductive success of certain warmwater fishes, in particular those requiring clean stony sites for spawning (Berkman and Rabeni 1987). Streams near heavily mined areas often show lower benthic macroinvertebrate diversity, abundance, and productivity, which, in turn, reduce food availability for foraging fishes (Waters 1995). The impact of aggregate mining on freshwater fish assemblages and other instream biotic communities in Alabama is not well understood because < 50% of surface mining activity is inspected in the state while mining is in progress (Alabama Department of Industrial Relations 2003).

Relationship between Flow Regime and Stream Fish Assemblage

The effects of altered flow regimes on streams can include changes in physical characteristics such as sediment transport, channel depth and width, current velocity and substrate composition. These physical changes, in turn, can alter both the structure and function of fish assemblages (Ward and Stanford 1983; Bain et al. 1988; Ligon et al. 1995; Poff et al. 1997; Marchetti and Moyle 2001). Drought and other unpredictable environmental disturbances commonly alter many aspects of stream systems, including water chemistry, stream size, water temperature, streambed structure and substrate, and flow (Taylor et al. 1996; Medeiros and Maltchik 2001; Matthew and Matthews 2003). Such environmental variation can dramatically alter the living conditions and aquatic habitats within the water, affecting much of the aquatic fauna inhabiting streams (Keaton et al. 2005).

Several studies have examined how flow variability influences fish community structure and function in streams. The effects of low flows on the distribution of stream fishes have been examined following drought (Larimore et al. 1959; Deacon 1961). Larimore et al. (1959) found that variable flows in Illinois streams as a result of severe drought (created suboptimal habitat conditions) that led to decreased survival of fishes during early life stages or physiological stresses on adults. However, they found that most fish assemblages rapidly reestablished pre-drought populations, noting most species returned to previously dewatered sites. Only 3 species remained absent from sites 3 years after normal flows had returned: the longear sunfish *Lepomis megalotis*, blackstripe topminnow *Fundulus notatus* and hornyhead chub *Nocomis biguttatus* (Larimore et al. 1959). Adult fish reentering a previously dewatered stream reproduced rapidly, resulting in populations dominated by young-of-year individuals.

Deacon (1961) documented the effects of drought on the fishes of the Marais des Cygnes and Neosho rivers, Kansas, noting reductions in some species during the drought and expansions of others in post-drought years. He found that the channel catfish *Ictalurus punctatus*, Neosho madtom *Noturus placidus* and two darter species, the fantail darter *Etheostoma flabellare* and the orangethroat darter *Etheostoma spectabile* rapidly declined in abundance during drought years and did not fully recover to pre-drought numbers. In contrast, Deacon found that that spotted bass *Micropterus punctulatus* and several *Lepomis* species, including the green sunfish *Lepomis cyanellus* increased in numbers during drought years in the two rivers. These habitat generalists quickly repopulated reaches that had been previously dewatered once normal flows had resumed. Species that occupied restricted habitats, such as the Neosho madtom, fantail darter and orangethroat darter, were slow to increase in numbers post-drought. As a result, these species disappeared or became restricted to a single tributary in the stream system (Deacon 1961).

Gorman and Karr (1978) studied the impact of flow modification (as channelization) on fish assemblages in Indiana and Panama streams. There, streams with a more natural flow regime supported fish assemblages with high species diversity and greater seasonal stability than streams with altered flow regimes. The authors concluded that most fishes were habitat specialists and that their presence or absence in a particular stream was dependent on the degree of habitat heterogeneity and flow stability in a stream (Gorman and Karr 1978).

As a test of the resiliency of fish assemblages, Meffe and Sheldon (1990) analyzed fish assemblage structure at 37 sites in South Carolina streams before and one year after defaunation resulting from drought. The authors noted rapid post-defaunation recovery with little change in richness, density, and biomass between the pre- and post-defaunation period. They concluded that assemblages were not randomly structured units, but instead were highly correlated with local habitat structure including depth, width, current velocity, substrate, and type and availability of cover (Meffe and Sheldon 1990).

Poff and Allan (1995) studied the relationship of flow variability on the organization of stream fish assemblages using data from 34 sites in Minnesota and Wisconsin for each of 106 species to test the premise that hydrological factors (water depth, current velocity, food availability, and thermal regime) were significant determinants of assemblage structure. Findings indicated that unstable sites with highly altered flow regimes supported resource generalist species classified as resource generalists where sites with more natural flow regimes were characterized by a higher proportion of specialist species. The authors concluded that hydrological variability induced by climate change could modify stream fish assemblages in the region.

Spence et al. (1999) assessed the health and condition of fish assemblages in several Oklahoma streams during a drought year in 1998. During the summer drought, the authors noted that highly variable flows from local drought resulted in poor habitat quality (dewatered riffles and runs and increased isolated pools). The authors used the Index of Biotic Integrity (IBI) to examine the condition of fish assemblages in streams impacted by the drought. The authors observed high correlation between IBI scores and flow-dependent parameters including pool variability and the presence of rocky runs and riffles. They noted that drought-impacted streams characterized by low flows, high pool variability, and little to no riffle or run habitat had much lower IBI scores than reference streams relatively unimpacted by the summer drought.

Marchetti and Moyle (2001) studied the effects of flow regime on fishes in Putah Creek, a highly regulated stream in central California, as a result of water diversion over the last 40 years. Because of the combination of extreme dry and extreme wet years that were encountered over the 5-year study, the authors were able to test the hypothesis that a more natural flow regime favors native fishes and suppresses exotic fishes because climatic conditions abruptly changed stream flow from conditions characteristic of a dammed stream to conditions similar to a natural flow regime. The authors noted higher numbers of native fishes during wet years with large peak flows in winter and sustained flows during the summer. In contrast, during extremely dry years, exotic species were favored when winter flows were lower and summer flows were intermittent. The authors concluded that higher flows during wet years flushed many exotic species from the stream while simultaneously creating conditions that favored reproduction by native species (Marchetti and Moyle 2001). This study concluded that restoration of natural flow regimes is important for reversing the decline of native fish populations in streams of the western United States. This study illustrated the need for examining the recovery of fish assemblages following restoration of natural flow regimes in artificially dewatered streams, such as in the Chewacla Creek stream reach that comprised my study.

Keaton et al. (2005) studied the effects of drought on fish assemblages in selected streams in the Piedmont ecoregion of South Carolina. Fishes were sampled during drought conditions in 2000 and then again during post-drought conditions in 2003. Between the two time periods the authors observed many more juveniles and young of the year (YOY) in the 2003 (vs. 2000) collections, which suggested that most species exhibited greater reproductive success following the drought. The authors also noted higher variability in common measures of assemblage structure, including abundance, richness, and diversity in collections from 2000 than in 2003. At all sites fish assemblage structure was highly correlated with instream habitat structure, with sites that had greater discharge and deeper water exhibiting more habitat heterogeneity and greater stability in fish community structure (Keaton et al. 2005).

Gelwick et al. (2001) found that instream habitat structure was a good predictor of fish assemblage condition, so habitat instability associated with variation in stream flow will adversely impact the fish assemblage residing within it. For example, low flows during drought can limit habitat resources and fish mobility and kill fishes directly, either by desiccation or predation (Lohr and Fausch 1997). Furthermore, reproduction and juvenile recruitment can be negatively impacted by environmental stress associated with drought (Freeman et al. 1998; Schlosser et al 2000; Keaton et al. 2005).

Altered natural flow regimes and their impact on fish assemblages in streams resulting from impoundment, channelization, urbanization, water diversion, agricultural practices, clearcutting for timber, and construction have been extensively studied. Mining, another highly visible anthropogenic activity, has degraded stream quality in the US and many parts of the world. Research has focused mostly on water quality impacts from mining activities on streams and resident fish communities, such as sedimentation, erosion, altered water chemistry (Gammon 1970; Knight and Newton 1977; Harkins et al. 1980) and decreased stream water pH (Leonard and Orth 1986; Robb 1994; Ahle and Jobsis 1996; Henry et al. 1999; Sams and Beer 2000; Soucek 2001). However, very few studies have been conducted on how mining activities disrupt natural flow regimes in streams and the degree to which this impact aquatic biota (Starnes and Gasper 1996). Thus, my study of how fish assemblages have been impacted by altered stream flows in Chewacla Creek between Lake Ogletree and the quarry was unique because it examined several factors that have contributed to the modified flow regime in the reach: (1) the influence of Lake Ogletree, (2) drought conditions occurring between 1999 and 2001, and (3) sinkhole activity in the stream channel near the quarry as a result of mining.

Biomonitoring

The Water Pollution Control Act was implemented in 1948 to protect aquatic systems in the United States (Dauwalter et al. 2003). Now known as the Clean Water Act after 1972 amendments, this statute mandates state and federal agencies to monitor US surface water conditions and restore and maintain the biological integrity of the nation's water resources (Karr 1991; Shaner 1999). Historically, monitoring of US surface waters focused primarily on point-source effluents and setting criteria levels for chemical contaminants (Bowen 1996; Shaner 1999). For many years, it was believed that monitoring point-source discharges was sufficient to protect water resources and, thus, provided water regulatory agencies with a means to satisfy the directives of the Clean Water Act. Thus, assessment of water resources in the US historically emphasized non-biological variables, specifically physical and chemical water quality, because it was presumed that improvements in the physical and chemical water quality would restore biological integrity (Karr 1993; Shaner 1999).

Implementation of effluent regulatory programs for point-source discharges helped to maintain adequate water quality, but this approach was insufficient to protect a variety of aquatic resources, particularly fishes, from non-point source impacts (Karr 1991). In addition, habitat alteration, flow regime modification, and changes in the energy base for stream biota are detrimental impacts not detected by point-source effluent monitoring programs.

A need existed for a more comprehensive approach to monitoring water quality in response to the continued decline in biological integrity of aquatic resources. This prompted some regulatory agencies to integrate a biological approach, or biomonitoring, into their water quality monitoring programs (Karr 1991; Shaner 1999). Karr (1999) defined biomonitoring as the evaluation of the health of a biological system to assess degradation to aquatic ecosystems from a multitude of human impacts. Biomonitoring was based on the direct observation of aquatic communities, for which traditional physical and chemical water quality monitoring programs generally have proven to be unreliable alternatives (Shaner 1999, 2001).

To address the directives of the Clean Water Act, biological assessment methods have incorporated a variety of taxonomic groups including algae, benthic macroinvertebrates, and fishes, all of which reflect water quality through the structure, composition, and functional relationships of their communities (Karr 1981; Fausch et al. 1984; Angermeier and Karr 1986; Saylor and Scott 1987; Hughes 1990; Shields et al. 1995; Yoder and Rankin 1998; Roth et al. 1998; Barbour et al. 1999). Fishes are especially useful in assessing environmental impacts resulting from anthropogenic activities because they occupy multiple trophic levels in aquatic communities, are relatively long-lived, are sensitive to a variety of stressors, and are easily sampled (Karr et al. 1986; Helms et al. 2005). Furthermore, fishes are an important, visible component of any aquatic ecosystem and, as such, the condition of the fish assemblage can be easily communicated to the public.

Biomonitoring Tools

Index of Biotic Integrity (IBI). Monitoring and managing the integrity of water resources can be a cumbersome and time-consuming process, often lacking a suitable framework for the analysis and interpretation of biological data (Weber 1981). A practical method of assessing stream quality using fish assemblages is the index of biotic integrity, first introduced by Karr (IBI, 1981). The IBI is widely used to rapidly, efficiently, and directly assess the quality of water resources based on the condition of the fish community (O'Neil et al. 2006).

The IBI originally developed by Karr (1981) consisted of 12 metrics that assessed 3 attributes of the fish assemblage: species richness and composition, trophic composition and dynamics, and fish abundance and condition. Each of the 12 metrics was scored by comparing its value to expected values determined from regional reference sites (Karr et al. 1984). A reference site was considered a stream of similar size in the same ecoregion that remained relatively undisturbed by human impacts (Shaner 1999). An ecoregion is a large area of land and water that contains a geographically distinct assemblage of natural communities and is defined primarily by similar landforms, climate and vegetation (Omernick 1987).

The metrics were scored based on whether they approximated (5), deviated somewhat (3), or deviated strongly (1) from the values measured from the reference site (Karr et al. 1986; Shaner 1999, 2001), and metric scores were summed to generate a total IBI score. Based on the total IBI score, a stream was assigned into an integrity class (excellent, good, fair, poor, very poor, and no fish), which allowed for easy interpretation of the condition of the instream fish assemblages (Karr et al. 1984; Shaner 1999). Table 1 lists the original metrics suggested by Karr et al. (1984) for streams in the midwestern US.

The IBI offers a broad range of application when added to any water quality monitoring program. It can be used to evaluate current conditions at a site and identify local sources of disturbance, detect temporal trends at a site sampled repeatedly, and to compare fish assemblages across multiple sites (Karr et al. 1986; O'Neil et al. 2006). In addition to evaluating fish assemblages, IBIs have been developed for other aquatic organisms including macroinvertebrates (Kerans and Karr 1994; Resh et al. 1995; Barbour et al. 1999; Maxted et al. 2000; Klemm et al. 2003) and periphyton (Barbour et al. 1999; Hill et al. 2000). Past studies have shown the IBI to be an effective tool in identifying many types of environmental degradation in aquatic systems, including: (1) wastewater treatment effluents (Karr et al. 1985; Hughes and Gammon 1987), (2) flow modification (Bowen et al., 1996), (3) urbanization and riparian zone disturbance (Steedman 1988; Schleiger 2000, Helms et al. 2005), (4) agricultural and construction practices (Karr et al. 1987; Crumby et al. 1990; Rabeni and Smale 1995; Frenzel and Swanson 1996), sedimentation impacts (Walters et al. 2001), and (6) acid mine drainage (Leonard and Orth 1986; Ahle and Jobsis 1996).

Since Karr (1981) originally proposed the IBI, improvements in IBI procedures have been made. Guidelines for identifying reference sites (Hughes 1995), improvements in metric selection (Scott and Hall 1997, Hughes et al. 1998; McCormick et al. 2001; O'Neil et al. 2001, 2006), detection of metric redundancy (Bowen et al. 1996;

Angermeier et al. 2000; Dauwalter et al. 2003; Brambett et al. 2005) and modification of the original scoring criteria and development of new scoring methods (Fausch et al. 1984; Minns et al. 1994; Hughes et al. 1998; Smogor and Angermeier 1999; Dauwalter and Jackson 2004; Bramblett et al. 2005) all have enhanced the performance of the IBI.

The IBI was originally developed for streams in the midwestern United States (Fausch et al. 1984; Karr et al. 1984), but has been shown to be highly adaptable to different geographic regions and aquatic ecosystems (Fore et al. 1994). For example, an IBI has been developed in the United States for lake ecosystems (Minns et al. 1994), modified rivers (Bowen et al. 1996), tailwaters (Scott 1999), coldwater streams (Leonard and Orth 1986; Lyons et al. 1995; Mundahl and Simon 1999), prairie streams (Lydy et al. 2000; Shearer and Berry 2002), reservoirs (Scott 1992; McDonough and Hickman 1999), and wetlands (Danielson 1998; Wilcox et al. 2002; Teels et al. 2004). Outside of the US, the IBI has been successfully utilized for biological assessment of aquatic systems worldwide (Steedman 1988; Oberdoff and Hughes 1992; Lyons et al. 1995; Hugueny et al. 1996; Hughes and Oberdoff 1999; Karr and Chu 1999). Miller et al. (1988) found that the IBI was applicable to a broad range of stream sizes with varying levels of habitat and water quality degradation. The authors also noted that the IBI was more responsive to disturbance after modification of the original Karr (1981) metrics to account for regional differences.

When developing an IBI for a particular ecoregion, it must be adjusted to local ecological conditions (Paller et al. 1996). This typically requires selection and (or) modification of the original Karr (1981) metrics to accurately discriminate between disturbed and undisturbed sites in the region in which the IBI will be used. However, the basic index structure developed by Karr (1981) was maintained to maximize metric sensitivity to a range of impacts (Paller et al. 1996). Metrics were included within each of the basic categories in keeping with Karr's original index: species richness and composition, trophic composition, abundance, condition and local indicator species (Paller et al. 1996).

Karr's original metrics (1981) were developed for midwestern streams, many modifications have been necessary in streams in different US regions (Miller et al. 1988). Five of Karr's original metrics have required frequent modification in different published versions of the IBI, including the number of darter species, the number of native sunfish species, the proportion of individuals as insectivorous cyprinids, the proportion of individuals as green sunfish and the proportion of individuals as hybrids (Hughes 1990). Where darter species were depauperate or rare, the numbers of benthic insectivore cyprinids and sculpin species have been used as substitutes in streams in New England and on the West Coast, respectively (Hughes 1990). In the Southeast, madtoms and sculpins have been added to this metric (Hughes 1990; Paller et al. 1996; Schleiger 2000; Walters et al. 2001; Compton et al. 2003). Schleiger (2000), in particular, noted the need to include benthic species in the IBI because those species are often disproportionately affected by degradation of aquatic systems. Thus, Schleiger (2000) modified the number of darter species metric to include the number of madtom and sculpin species in lower Piedmont and upper Coastal Plain streams in Georgia.

The number of sunfish species metric has been replaced by the number of minnow species, or water-column species in the Northeast and on the West Coast, respectively (Hughes 1990). This modification was necessary because sunfish species were introduced or depauperate in both regions (Hughes 1990). In the Southeast, the number of sunfish species metric required modification to include native sunfish species only, and excluded non-native species and hybrids such as the redbreast sunfish *Lepomis auritus* because of their increase in both the Tennessee and Alabama drainages (Georgia Department of Natural Resources 2005). Onorato et al. (1998) suggested that the original number of sunfish species metric be substituted with the number of centrarchid species because of the increase in the species richness of the centrarchid family in the southeastern United States. Hlass et al. (1998) and O'Neil et al. (2006) substituted the number of native minnow species metric for the original number of sunfish species because minnows exhibit a wider range of pollution tolerances, habitat preferences, and trophic and reproductive guilds and, as such, have greater value as an ecological indicator of water quality than sunfish. O'Neil and Shepard (1998) suggested that the original number of sunfish species should be substituted or eliminated entirely in IBI studies in the southeastern United States because the metric did not reliably distinguish between moderately and severely impacted streams. They found that *Lepomis* species often dominate in moderately degraded streams in Alabama and as such the original number of sunfish species metric showed low sensitivity to disturbance (O'Neil and Shepard 1998).

The proportion of insectivorous cyprinids metric was originally suggested by Karr et al. (1984) because these species are a dominant trophic group in streams in the Midwest. However that metric has been replaced by "all insectivores", "specialized invertebrate feeders", or "all invertivores" in other geographical regions. Ahle and Jobsis (1996), Hughes and Oberdoff (1999), O'Neil and Shepard (2000), and O'Neil et al. (2006) substituted the proportion of insectivorous cyprinids metric with the proportion as invertivores metric because it was more inclusive and ecologically accurate than percent insectivorous cyprinids for assessing the degree to which the invertebrate community was degraded in streams. Compton et al. (2003) expanded the original percent insectivorous cyprinids metric to include all insectivorous individuals because of the high number of darter and madtom species classified trophically as insectivores.

The original proportion of individuals as green sunfish metric has been replaced frequently, particularly in southeastern streams (Hughes 1990). The most commonly substituted metric for percent green sunfish has been the proportion of tolerant species, which was viewed as more inclusive than green sunfish alone (GA DNR 2005, O'Neil et al. 2006). Crumby et al. (1990), Shields et al. (1995), Ahle and Jobsis (1996), Shaner (1999), and Compton et al. (2003) replaced the percent green sunfish metric with a percent tolerant species metric because of the scarcity of green sunfish in streams. Several studies conducted by the Geological Survey of Alabama (GSA) have substituted percent green sunfish with percent *Lepomis* species as several of these *Lepomis* spp. have become dominant in disturbed habitats in Alabama streams (Shepard et al. 1997; O'Neil and Shepard 2000). IBI studies conducted by Schleiger (2000) and Bowen et al. (1996) eliminated the percent green sunfish metric entirely because these fishes composed < 1% of the total fish fauna collected in sampled streams. The Tennessee Valley Authority (TVA) has replaced the percent green sunfish metric with a percent tolerant species metric in its IBI assessments in the Tennessee River basin (Saylor and Scott 1987; Saylor et al. 1988; Wade and Stalcup 1987; Saylor 1989; Saylor and Ahlstedt 1990; Scott 1992).

The original proportion of individuals as hybrids metric has been problematic in other regions outside the Midwest and has been regularly substituted or eliminated entirely in IBI studies (Hughes 1990). Researchers have found this metric to be of limited use because of difficulty in identification of hybrid individuals and the lack of a consistent relationship between hybridization and environmental degradation (Shields et al. 1995; Schleiger 2000). An exotic species metric or reproductive guild metric has been substituted regularly for the percent hybrids metric in southeastern streams as recommended by the Ohio EPA (1987). Schleiger (2000) and O'Neil et al. (2006) substituted the percent hybrids metric with a reproductive guild metric because (1) few hybrids were found in the study streams, and (2) reproductive guild metrics were extremely sensitive to even moderate stream degradation and, as such, were deemed more reliable metrics than the percent hybrids metric. A list of metrics previously used in published IBI studies is presented in Table 2. These metrics were considered as potential metrics for this study from which a group of twelve metrics were selected based on evaluation of fish assemblages representative of southeastern Piedmont streams.

IBI Studies of Streams in the southeastern United States

A number of regional IBIs have been successfully developed for warmwater aquatic systems in the southeastern United States. In Alabama, the IBI has been used by the TVA in the Tennessee River basin since 1986 (Saylor and Scott 1987; Saylor et al. 1988; Wade and Stalcup 1987; Saylor 1989; Saylor and Ahlstedt 1990; Scott 1992). The

Bowen et al. (1996) developed an IBI for the Tallapoosa River system to evaluate the sensitivity of the IBI to fish community changes resulting from streamflow alteration. The authors determined that the IBI successfully discriminated between unimpacted sites and sites undergoing severe flow fluctuations as a result of dam impoundment. The authors recommended that the IBI be used in long-term monitoring programs in regulated southeastern river systems.

The IBI has been applied by Onorato et al. (1998) and Davenport et al. (2005) to fish assemblages in the upper Cahaba River system. To assess the ecological health of the upper Cahaba, Onorato et al. (1998) conducted biomonitoring at 13 sites between 1995 and 1997. IBI scores were found to be lower at sites located downstream, and within the metro Birmingham area, which were most heavily influenced by urbanization. The authors concluded that these changes were related to point- and non-point sources of pollution from extensive urban development. Davenport et al. (2005) surveyed fishes at 15 sites on the upper Cahaba and used a modified version of the Shepard et al. (1997) IBI for the Cahaba River. The authors obtained similar IBI results to those of Onorato et al. (1998) as IBI scores were lowest in samples from the middle reaches of the river located in the most heavily urbanized areas in Birmingham. Sediment accumulation was found as the principal source of environmental degradation in those reaches where IBI scores were lowest (Shepard et al. 1997).

An extensive watershed study by the Geological Survey of Alabama (GSA) of the Tallapoosa and Coosa river systems incorporating the IBI was conducted between 2003 and 2005 (O'Neil et al. 2006). IBI metrics and scoring criteria were calibrated to conditions in the Tallapoosa and Coosa River systems and then applied to a watershed study in Terrapin Creek (Coosa River drainage) to demonstrate the newly calibrated IBI and evaluate biological conditions in the stream. The authors showed that the IBI fits well into the framework of biomonitoring because of its integrative nature, ease of calculation, and broad regional applicability. However, they concluded that a better understanding of the relationship between fish assemblages and classification factors such as drainages, ecoregions, and physiography needed to be developed to guide future IBI assessments. Finally, they concluded once IBI regions have been delineated, they should be thoroughly sampled across the gradient of human disturbance and stream sizes, and IBI metrics should be calibrated to account for both natural faunal variation and type and intensity within each IBI region.

Regional IBIs also have been developed for other southeastern states. Crumby et al. (1990) applied the IBI to fish collected from Roaring River in north-central Tennessee to determine IBI sensitivity to known disturbances. They found the IBI to be useful in assessing habitat and water quality disturbances in Roaring River and that IBI scores were consistently lower at sites heavily impacted by sedimentation.

An IBI study conducted by Shields et al. (1995) was used to quantify environmental quality in 27 watersheds bordering the Mississippi River alluvial plain in northwestern Mississippi, and to test the IBI for relating fish community characteristics to habitat degradation. The authors concluded that the IBI was effective for assessing biotic integrity over a large area with widespread degradation of physical habitats. The authors cautioned that future IBI studies must take into consideration the need for suitable reference data for setting metric scoring criteria, variability in IBI scores at different sites over time, and variability in IBI scores at sites with small sample sizes.

Ahle and Jobsis (1996) developed an IBI to assess the impact of effluents from gold mine operations in nine Piedmont streams of South Carolina. It was determined that the IBI successfully differentiated between streams receiving mining effluents from those affected by only minor anthropogenic impacts. The authors concluded that the IBI was an excellent tool for monitoring the fish assemblages of South Carolina Piedmont streams.

Paller et al. (1996) developed an IBI to assess the health of fish assemblages in streams in the Sand Hills ecoregion on the upper coastal plain of South Carolina. Metrics were selected for the IBI based on their ability to effectively discriminate disturbed and undisturbed sites in the Sand Hills ecoregion. One unique aspect of this study was the development of a procedure based on species-area curves to remove the potentially confounding effects of unequal stream size and sampling effort from selected species richness metrics. A modified IBI was developed that was minimally affected by stream size and sampling effort and accurately discriminated between disturbed and undisturbed sites in South Carolina coastal plain streams (Paller et al. 1996).

Schleiger (2000) conducted an intensive IBI study to detect land-use effects on fish communities in > 500 Piedmont streams in west-central Georgia. It was determined that the IBI effectively identified streams in low-density urban areas adversely impacted by nonpoint and point-source runoff. Schleiger (2000) concluded that the IBI was useful in differentiating stream quality between sites and that it was a valuable tool in long-term biological monitoring of Piedmont streams.

Paller (2002) developed an IBI for assessing fish assemblages sampled over a 2- to 10-year period in 17 undisturbed and disturbed streams in the Savannah River Site (SRS), a Department of Energy reservation located on the upper coastal plain of South Carolina. An IBI was used to compare temporal fish assemblage variability between the disturbed and undisturbed sites. IBI scores were significantly lower in disturbed (vs. undisturbed) streams and showed greater variation over time in disturbed streams. Paller

(2002) also found that assemblages exhibited greater temporal variability in fish community structure in disturbed streams. In contrast, assemblages were more stable and persistent in undisturbed sites.

Regulatory agencies in other southeastern states that have developed an IBI for inclusion in water resources biomonitoring include the Alabama Department of Environmental Management (ADEM), the Kentucky Department for Environmental Protection, the North Carolina Department of Environment, Health, and Natural Resources, and the South Carolina Department of Health and Environmental Control (Shaner 1999). In addition, the Fisheries section of the Georgia Department of Natural Resources has developed a standardized IBI for wadeable streams in the Piedmont ecoregion of Georgia (GA DNR 2005).

Although the IBI has been applied extensively to studies relating many different types of anthropogenic disturbances to the health of fish assemblages in aquatic ecosystems, few studies used the IBI to assess how mining in a stream watershed impacts stream fish assemblages (Leonard and Orth 1986; Ahle and Jobsis 1996). Although Spence et al (1999) used an IBI to monitor fish assemblage health in streams with variable flow regimes during a drought year, no studies were found that used the IBI to examine changes in the resident fish community in a stream following an extended period in which the flow regime was altered. Thus, the IBI is an underutilized tool for assessing how an altered flow regime, resulting from anthropogenic activities in the watershed such as mining, can affect stream fish assemblages. As an effective biomonitoring tool, IBIs can provide biological data that can be used to identify perturbations in the watershed and measure effects based on changes to fish assemblages (Karr 1981; Moyle 1994; Stewart and Loar 1994).

Index of Well-Being (Iwb). Another approach to assessing fish assemblage data is the Index of Well-Being (Iwb). It was originally developed by Gammon (1976) to evaluate large riverine fish assemblages (Yoder and Smith 1999). It incorporates measures of abundance, biomass, and the Shannon diversity index (H') into a single index value for assessing fish assemblage health (Gammon 1976; Ohio EPA 1987; Yoder and Smith 1999; Schleiger 2000). The Iwb was based on 4 measures of species diversity, abundance, and biomass, and was an attempt to produce an integrated evaluation of these fish assemblage attributes (Yoder and Smith 1999). Individual performances of abundance, biomass, and the Shannon index as indicators of fish assemblage quality have historically proven disappointing; however, when combined in the Iwb, these individual attributes were better predictors of fish assemblage condition (Yoder and Smith 1999). However, the Iwb does not incorporate the more sensitive aspects of community function such as trophic and reproductive guilds. A high Iwb value is indicative of a healthy fish assemblage characterized by high diversity and abundance of fishes.

The Ohio EPA (1987) developed a modified Iwb that was different from the original index developed by Gammon (1976). Any species designated as highly tolerant, exotic, and hybrid were eliminated from the numbers and biomass components of the Iwb. This eliminated the positive bias produced by increased abundance of tolerant, hybrid, and introduced species at degraded sites, but retained their influence on the Shannon diversity indices (Yoder and Smith 1999; GA DNR 2005).

Relationship between the IBI and the Iwb

The Iwb is used as a complementary evaluation to the IBI by the Ohio EPA and GA DNR for monitoring environmental quality in streams and rivers. It reflects overall production in terms of fish abundance and biomass and usually precedes the IBI in indicating the initial stages of fish assemblage recovery following a disturbance (Yoder and Smith 1999). The Iwb is very responsive to severe disturbances that result in reduced abundance, biomass, and species richness but does not incorporate the more sensitive attributes of the fish community such as the functional guilds of the IBI (Ohio EPA 1987). The IBI incorporates attributes of the fish assemblage such as the trophic and reproductive guilds, so it tends to respond more slowly than the Iwb in terms of exhibiting fish assemblage recovery (Yoder and Smith 1999). Both indices, when used together, can provide a more accurate assessment of fish assemblage response to anthropogenic disturbance (Ohio EPA 1987; Yoder and Smith 1999; Schleiger 2000; GA DNR 2005). Use of the Iwb requires the collection of biomass data, which is not required in current versions of the IBI (Yoder and Smith 1999; Schleiger 2000). Because both indices have their strengths and weaknesses, the Ohio EPA and GA DNR recommended both be used in bioassessments of stream fish assemblages.

III. MATERIALS AND METHODS

Physiography and Geology

For this study, three third-order stream sites located in the Tallapoosa River Drainage, East Central Alabama, USA, were sampled in fall 2004 and 2005. All three streams were in the Southern Outer Piedmont subecoregion above the Fall Line in Lee County. The Southern Outer Piedmont is a Level 4 subecoregion of the Piedmont ecoregion, portions of which are in both East Central Alabama and West Central Georgia (Omernick 1987; Griffith et al. 2001). The Piedmont ecoregion comprises a transitional area between the mountainous Appalachians to the northwest and the relatively flat coastal plain to the southeast (Omernick 1987; Griffith et al. 2001). The topography is characterized by rolling hills, broad and narrow ridges, and deeply narrow valleys (Mulholland and Lenat 1992). Piedmont soils are fine-textured, acidic, and nutrient-poor (Mulholland and Lenat 1992; Boyd 2000; Webber 2002). Major forest cover types in the Piedmont ecoregion are loblolly-short leaf pine and oak-pine (Turner 1987; Walser and Bart 1999). Piedmont streams have low to moderate gradients with cobble, gravel and sandy substrates, although many streams now contain large quantities of sand and silt as a result of sediment deposition from human activities (Webber 2006). Stream elevations range from about 50 to 600 m above sea level (Mulholland and Lenat 1992).

Almost all Piedmont streams have been dramatically affected by historical land use (Mulholland and Lenat 1992). In the past 50 years landscape alterations resulting from agriculture, silviculture, and watershed urbanization in heavily populated areas have altered stream hydrology, water quality, and physical habitat; these alterations have adversely affected biological communities (Mullholland and Lenat 1992; Helms et al. 2005). East Central Alabama and the areas surrounding the cities of Opelika and Auburn are notable examples of this trend of increased urbanization (Webber 2002).

Description of Study Sites

Chewacla Creek in the study area is a third-order reach in Lee County, Alabama. The stream basin encompasses a total drainage area of about 383 km^2 (148 mi^2). As part of the Tallapoosa River drainage, Chewacla Creek's headwaters originate south of Opelika. The stream flows through pasture, commercial, and private developments before emptying into Lake Ogletree, a 1.2 km^2 (300 acres) lake that is the primary drinking water supply for the city of Auburn (Figure 1). Flow into Chewacla Creek downstream of Lake Ogletree dam is regulated by water levels in the lake. When the lake is at full pool, water flows over the spillway into the channel downstream. When the lake is low during dry periods of the year, little or no flow goes over the spillway. During the latter periods only seepage below the dam and local springs supply water to the downstream channel.

Downstream from Lake Ogletree, Chewacla Creek flows through several geological formations including the Manchester Schist, the Chewacla Marble, and the Hollis Quartzite (USFWS 2003). Extensive mining of the Chewacla Marble first began in the early 1900s (Prouty 1916). The Chewacla Marble is a narrow strip of highly crystalline dolomite that originates ~ 8 km (5 miles) southeast of Opelika (Prouty 1916).

The belt in which the dolomite occurs is ~ 0.5 km (a third of a mile) wide and extends ~ 4 km (2.5 miles) in a northeast and southwest direction (Prouty 1916). The primary aggregate mined in the Chewacla Marble has been limestone. The chief quarry located along Chewacla Creek, now owned and operated by the Martin Marietta Company, was first opened in 1961 for the purpose of supplying building materials to the Alabama Department of Transportation (DOT) to construct Interstate 85 (Scott Weinhold, Martin Marietta Company, personal communication). The Martin Marietta Company purchased the quarry in 1995 from Drevo Industries and has operated the stone-crushing facilities at the quarry since 1995 (Scott Weinhold, Martin Marietta Company, personal communication).

The major tributaries into Chewacla Creek downstream of Lake Ogletree are Moores Mill Creek and, further downstream, Town and Parkerson Mill creeks. Moores Mill Creek was an ADEM 303 (d)-listed stream, impaired because of pollution from excessive sediment (Webber 2006). Moores Mill Creek flows into Chewacla Lake in Chewacla State park. Downstream from Chewacla Lake, Moores Mill Creek joins Chewacla Creek to form a fourth-order reach that flows south to Opintlocco Creek in Macon County, Alabama, where they join to form Uphapee Creek. Major tributaries into Chewacla Creek upstream of Lake Ogletree include Nash and Robinson creeks.

Two sampling sites were chosen in the reach of Chewacla Creek between Lake Ogletree and the quarry for the collection of fish samples and analysis of hydrological and chemical variables. Both sampling stations were on the Chewacla Creek main stem. The reach has in the past exhibited a highly altered flow regime during dry periods as a result of no water flowing over the lake spillway and sinkholes in the channel in the vicinity of the quarry further downstream. The upstream site (CH-U) was ~ 300 m downstream from the Lake Ogletree dam. The downstream site (CH-D) near the quarry was ~ 100 m downstream of the “pretty hole”, a deep pool providing refuge for fishes during periods when this reach was completely dewatered. Location and stream characteristics for CH-U and CH-D are presented in Table 3.

The reference stream for this study was a third-order reach of Cane Creek (Figure 2). The Cane Creek site (Cane-1) was in a relatively undisturbed area of the Ropes Creek watershed. Ropes Creek is a tributary to Sougahatchee Creek, also in the Piedmont ecoregion in Lee County, Alabama. The sampling site was located just upstream of the bridge on Lee County Road 217. Collection of biological and chemical data was conducted concurrently with CH-U and CH-D in 2004 and 2005; however, no hydrological data were collected in Cane-1 both years. Location and stream characteristics for Cane-1 are also presented in Table 3.

Physical Habitat Assessment

A detailed habitat assessment was conducted at each site according to methods described in the US EPA Rapid Bioassessment Protocols (Barbour et al. 1999). Habitat in each reach was scored on the basis of instream variables including substrate and available cover, channel morphology, bank features, and streamside (riparian) vegetation. A total of 10 metrics developed by Barbour et al. (1999) were used in the habitat assessment for each study site (Table 4). Each metric was scored in the field based on observation and best professional judgment. Scoring criteria used to visually estimate physical habitat quality at sites in Chewacla and Cane creeks are presented in Table 5. The assessment used to evaluate physical habitat quality at all sites is presented in Table 6.

Characteristics of Flow Regime in Chewacla Creek

As part of the Chewacla Creek Safe Harbor Agreement (USFWS 2003), continuous monitoring of stream flow was conducted at 4 stations downstream of Lake Ogletree (Figure 3). Water level was measured at each station using Solinst Model 3001 Levellogger gages (pressure transducers) (Webber 2006). These data were available to characterize the flow regime in the study reaches downstream of Lake Ogletree. Discharge was measured in the field at least monthly using the velocity-area method as described in Rantz et al. (1982), Wahl et al. (1995), and Gore (1996). A Marsh-McBirney Flo-Mate Model 2000 portable flowmeter was used to measure the velocity. A Microsoft Excel program was developed to calculate discharge from the velocities recorded.

Flow measurements taken in the field were used to calculate stage/discharge regressions for each station (Webber 2006). Using the regression model, discharge (MGD) was calculated from stage level readings for each station. Hydrographs showing mean daily discharge (MGD) were prepared from these data for both CH-U and CH-D.

Water Chemistry Analysis

Field and laboratory measurements of physical and chemical variables were conducted annually at Cane-1, CH-U, and CH-D during the study. Dissolved oxygen and water temperature were measured in situ using a Yellow Springs Instrument (YSI) Model

51B Dissolved Oxygen meter. Specific conductance was measured in situ using a YSI Model 30 Conductivity meter. Water samples were collected in 2-L Nalgene bottles at each site for analysis in an AU Fisheries laboratory. Nine water quality variables were measured: turbidity, total alkalinity, total suspended solids (TSS), total hardness, total nitrogen (TN), total phosphorous ($T\text{-}PO_4$), orthophosphate ($O\text{-}PO_4$), total ammonium ($NH_3\text{-}N$) and nitrate nitrogen ($NO_3\text{-}N$). All water quality analyses conducted in the laboratory followed methods recommended by the American Public Health Association (APHA 1998). Turbidity was measured using the nephelometric method with a Model 18900 Ratio Turbidimeter. Total alkalinity was measured by titrating a water sample with a standard 0.1 N hydrochloric acid (HCl) solution to a pH of 4.5. Total hardness was measured by titrating a water sample using an EDTA titrant and Erichrome Black T dye as a color indicator. Total nitrogen (TN) and total phosphorous (TP) were measured using the persulfate digestion method. Orthophosphate ($O\text{-}PO_4$) was measured using an ascorbic acid test. Total ammonium ($N\text{-}NH_3$) was measured using the automated phenate method. Nitrate nitrogen ($N\text{-}NO_3$) was measured using the cadmium reduction method.

Fish Sampling Methodology

Fishes were collected annually over a 2-year period at both impacted and the reference sites using Rapid Bioasssessment Protocols (RBP) recommended by the US EPA (Barbour et al. 1999). Sampling locations were selected to include riffle, run, and pool habitat. Sampling all major habitat types was important to obtain a representative sample of fishes illustrating assemblage attributes (such as abundance, richness, and trophic composition as these attributes can change along a habitat gradient (Schlosser

1982). Electroshocking was the primary method of fish collection. Equal sampling effort was used at each site to ensure the capture of the most representative sample possible at each site and so that direct comparisons of richness would be possible without species-area bias.

Block nets using 30-foot seines were set at the upstream and downstream ends of each reach to minimize fish escaping during sampling. Two separate electroshocking runs were conducted in each 100 meter reach. Pulsed DC current was used to shock fishes at each site. The first run utilized a small 10-foot jonboat with a portable generator and booms with attached electrodes hanging from the end of each boom. The boom electrodes were the anodes, and the boat hull was the cathode. Starting at the most downstream end of the reach, a three-person crew was utilized with one person pushing the jonboat upstream and two other persons positioned on each side of the boat to net stunned fish. A maximum current of 4 to 5 amps was applied by the generator which sufficiently stunned fish for netting. The second run involved the use of a Coffelt Mark-10 portable backpack electroshocker powered by a Honda EX350 generator. The backpack shocker consisted of a separate anode connected to a pole held by the operator and a trailing rattail cathode. Starting at the most downstream end, a two-person crew was utilized with one person carrying the shocker and a second person netting stunned fish. A maximum current of 4 to 5 amps was applied to stunned fish. The use of the backpack electroshocker was important because it allowed access to potential fish habitat difficult for the jonboat to access such as rootwads, undercut banks, and shallow riffle areas. Both runs utilized a bank-to-bank sweeping technique to ensure maximum coverage of all available fish habitat (Barbour et al. 1999). Captured fishes were transported in buckets to an aerated livewell located on the stream bank.

One minor logistical problem encountered during sampling was the natural low baseline conductivity in Cane Creek, which is typically ~ 50 µS (Webber and Blevins 2001; Webber 2002, 2006). Salt was used to increase the conductivity of the water during electroshocking. Fifty pound bags of salt were positioned in a riffle area at the most upstream end of the reach and then opened to allow the salt to disperse downstream and dissolve (Vincent 1960 and 1971; Hindman et al. 1990). This procedure was not necessary at the impacted sites because conductivity was typically greater than 100 µS at each site (Webber 2006), which was sufficient for generating the 4 to 5 amps needed without the addition of salt (Webber 2006). As a result, at no time during sampling was conductivity considered a limiting factor in electrofishing success at any of the sites.

Fishes were identified on the stream bank using appropriate reference materials (Mettee et al. 1996; Boschung and Mayden 2004). Individuals were weighed, measured for total length, inspected for lesions, tumors, and other abnormalities, and then returned alive to the stream. Individuals that could not be identified in the field were preserved in 10% formalin and returned to the laboratory at Auburn University for identification. All individuals captured were identified to species

Designation of Guilds and Tolerance Classes

Fish species were assigned to one of five feeding guilds (FG) based on published descriptions of their feeding ecology: herbivores (H), insectivores (INS), omnivores (O), top carnivores (TC), and invertivores (INV) (Mettee et al. 1996; Boschung and Mayden

2004; O'Neil et al. 2006). Herbivores consume plant material, either living or dead. Insectivores consume insects or insect nymphs almost exclusively. Omnivores consume both plant and animal matter. Invertivores are more general feeders consuming insects, crustaceans, mollusks, and other invertebrates. Top carnivores are opportunistic predators that consume a wide range of terrestrial and aquatic animal species; many consume fish as their major food source (O'Neil et al. 2006).

Fish species were assigned to one of four tolerance classes (TC): pioneer (P), intolerant (INT), intermediate (MI), and tolerant (TOL) (Ohio EPA 1987; Mettee et al. 1996; Goldstein and Simon 1999; Smogor and Angermeier 1999; Schleiger 2000; Tabit and Johnson 2002; Boschung and Mayden 2004; O'Neil et al. 2006). Schleiger (2000) defined pioneer species as those that will rapidly invade a previously disturbed habitat, and their presence in a site indicates temporal or unstable habitats. Examples of pioneer species include *Campostoma*, *Nocomis*, and *Lepomis* species (Schleiger 2000; Tabit and Johnson 2002). Intolerant species are classified as those whose ranges or abundances have decreased, presumably because of anthropogenic disturbance, and tolerant species as those known to be least detrimentally affected by anthropogenic disturbances to stream watersheds; many of these species historical ranges or local abundances have increased with increases in anthropogenic disturbance (Ohio EPA 1987; Smogor and Angermeier 1999). Ohio EPA (1987) defined intermediate species as those that are commonly observed and strongly associated with healthy fish assemblages, but are occasionally observed in slight to moderately degraded streams. Assignment of tolerance classes can be highly subjective because a number of fish species are sensitive to some types of disturbance and tolerant of others (Leonard and Orth 1986; Ebert and Filipek 1988; Hlass et al. 1998).

Fish species were assigned to one of two breeding guilds based on published descriptions of spawning behavior: lithophilic spawners and non-lithophilic spawners (Mettee et al. 1996; Boschung and Mayden 2004; O'Neil et al. 2006). Lithophilic spawners obligately use clean mineral substrates (i.e., sand, gravel, cobble) in which to deposit their eggs. Non-lithophilic spawners deposit their eggs on the remaining types of substrate or spawning surfaces such as aquatic vegetation, coarse organic matter, and other features not related to lithophilic substrates (O'Neil et al. 2006).

Characterization of Fish Assemblages Using an Index of Biotic Integrity (IBI)

Selection and modification of IBI metrics. Fish data collected during this study were analyzed using the Index of Biotic Integrity (IBI) (Karr, 1981). Previous IBI studies conducted in the southeastern United States were thoroughly researched to assist in the selection of IBI metrics for this study. Metrics were grouped into three categories as originally suggested by Karr et al. (1986): (1) species richness and composition, (2) trophic and reproductive guilds and (3) fish abundance and condition. Modifications of Karr's (1981) original metrics were made to reflect the fish fauna typically found in southeastern Piedmont streams. Each metric was scored and individual scores were summed to calculate a total IBI score for each site. A separate IBI score was calculated for each site for each year sampled (2004 and 2005). Thus, a total of six IBI scores were calculated for the three study sites. Each IBI score was then equated to a particular stream condition as originally developed by Karr (1981): Excellent, Good, Fair, Poor, and Very Poor.

Defining the reference condition. Many IBI studies used the concept of the reference condition to establish a network of reference streams that exhibited the most natural conditions against which biotic integrity can be measured (Compton et al. 2003). The reference condition refers to the range of measurable elements (i.e., chemistry, habitat and biology) that are found in natural environments. Thus, stream location, physical and chemical attributes of the water and surrounding watershed, and the resident biological communities are criteria often used for selection of regional reference sites and the establishment of the reference condition (Simon and Lyons 1995; ADEM 1999; Compton et al. 2003). Natural, undisturbed land cover is almost nonexistent in the southeastern Piedmont, so regional reference sites should not be considered pristine or undisturbed. Rather, regional reference sites should be defined as representing the best possible conditions given anthropogenic impacts that have occurred historically (O'Neil et al. 2006).

Typically, multiple reference sites must be sampled to obtain a reasonable estimate of the reference condition for a specific region (Shaner 1999). Since only one stream (Cane-1) was selected and sampled as a reference site, it was necessary to determine if it sufficiently met the reference condition established for Piedmont streams in other IBI studies. Analysis of fish data from museum collection records (Auburn University Museum of Natural History, unpublished data; University of Alabama Ichthyological Collection, unpublished data; Tulane University Museum of Natural

History, unpublished data, University of Georgia Museum of Natural History, unpublished data), previous ichthyological studies (Gilbert 1969; Dahlberg and Scott 1971; Webber 1971; Pierson et al. 1986), and ADEM reference streams provided reference data to compare with data from Cane-1. The ADEM reference sites were Cornhouse Creek (Randolph County Road 821), Hatchet Creek (Tyler Ford), Hurricane Creek (upstream of Randolph County Road 26), Emuckfaw Creek (3 miles west of Daviston, AL on unnamed Tallapoosa County road) and Weogufka Creek (County Road 41 near Stewartville, AL) (D. Miller, unpublished data)

Description of Core Metrics. The method of Karr et al. (1984) was used in applying the IBI to data sets from Chewacla and Cane creek sites with modifications intended to make the IBI more sensitive to regional conditions found in Piedmont streams. Twelve core metrics were chosen from the pool of 33 (Table 2) candidate metrics used commonly in previous IBI studies. These metrics were selected because they were most representative of fish assemblages found in Piedmont streams. The twelve core metrics selected for this study appear in Table 7.

Metrics 1-4 evaluated species richness and composition at each site. A description of each metric in this category is given below.

1. Total number of native fish species. This metric was a count of all the native fish species in the sample. The number of native fish species supported by similar streams in a given region generally decreases with environmental degradation (Karr et al. 1986). Exotic and hybrid species were excluded in the calculation of this metric as

recommended by the Ohio EPA (1987) because their presence does not provide an accurate assessment of long-term biotic integrity.

2. Total number of native cyprinid species. Fish species belonging to the family Cyprinidae are highly ubiquitous in streams in Alabama with a broad range of tolerances, habitat preferences, and trophic and reproductive guilds (Shepard et al. 1995; Mettee et al. 1996; O'Neil and Shepard 2000; Boschung and Mayden 2004). Many cyprinid species are specialized feeders, and high richness and abundance in a sample is indicative of a diverse aquatic macroinvertebrate community and a healthy trophic structure of a stream fish assemblage (Niemela et al. 1999). Cyprinids are found in a variety of habitats over a diverse array of substrates, thus providing a measure of the quality of instream cover and bottom substrates (O'Neil and Shepard 1998; GA DNR 2005). Many cyprinid species are sensitive to environmental disturbance and thus have value as ecological indicators (Fausch and Schrader 1987; Hlass et al. 1998).

3. Total number of native round-bodied sucker species. Sucker diversity is high in Alabama streams (23 species) (Boschung and Mayden 2004). Most sucker species are sensitive to environmental degradation (O'Neil and Shepard 1998; Boschung and Mayden 2004). Schleiger (2000) found sucker species in Georgia Piedmont streams to be sensitive to habitat alteration, siltation, and changes in water quality. Sucker species are found in all types of habitats and typically have long life spans and, thus, can provide long-term assessments of past and current environmental conditions (O'Neil and Shepard 1998).

4. Number of intolerant species. This metric included species intolerant of several types of environmental disturbance including siltation, low dissolved oxygen levels, flow alterations or chemical contamination (O'Neil and Shepard 2000). Intolerant species are typically the first to disappear after some form of disturbance to a stream (Ohio EPA 1987). Examples include minnow (*Hybopsis* and *Cyprinella*), darter (*Etheostoma* and *Percina*), madtom (*Noturus*), sunfish (*Ambloplites* and *Micropterus*) and sculpin (*Cottus*) species (Tabit and Johnson 2002; Boschung and Mayden 2004; O'Neil et al. 2006).

Metrics 5-10 measure species composition and trophic dynamics at each site. These metrics assess the quality of the energy base and the flow of energy through the stream community; they also provided a measure of the availability of suitable spawning habitat in each stream (Karr et al. 1986). A description of each metric in this category is given below.

5. Proportion of individuals as tolerant species. This metric has been commonly used as a replacement for Karr's (1984) original metric of proportion as green sunfish (Ohio EPA 1987; Hughes and Oberdorff 1999). In degraded streams, more pollution tolerant species typically dominate at the expense of less pollution tolerant species (Karr et al. 1986). Thus, diversity of the fish assemblage can decrease without a loss in species richness because of an increase in relative abundance of one or more pollution tolerant-species (GA DNR 2005). Examples include certain minnow (*Campostoma*, *Luxilus* and *Nocomis*), sucker (*Minytrema*), catfish (*Ameiurus*), sunfish (*Lepomis*), and darter

(*Percina*) species (Mettee et al. 1996; Boschung and Mayden 2004; GA DNR 2005; O'Neil et al. 2006).

6. Proportion of individuals as top carnivores. Viable and healthy populations of top carnivore species typically indicate a healthy, trophically diverse community (Karr et al. 1986; Hlass et al. 1998). Hughes and Oberdorff (1999) noted that top carnivores can be affected by long-term environmental impacts because they are typically long-lived. Only species that feed primarily on fish and larger invertebrates as adults were included (O'Neil and Shepard 2000; O'Neil et al. 2006). Examples include *Cottus*, *Lepomis*, *and Micropterus* species (O'Neil et al. 2006).

7. Proportion of individuals as *Lepomis* species. This metric represented the proportion of individuals in a sample that were *Lepomis* species. Non-native species and *Lepomis* hybrids were included in this metric. *Lepomis* species are highly responsive to increased disturbance with relative proportions increasing dramatically at disturbed sites (GA DNR 2005). An assemblage dominated by *Lepomis* species may be indicative of a reduction in macroinvertebrate diversity and a lack of suitable spawning habitat for broadcast spawners (GA DNR 2005). *Lepomis* species were dominant in moderately disturbed Alabama streams, sometimes exceeding 50% of the sample (O'Neil and Shepard (1998). Also, the proportion of *Lepomis* species differed significantly between disturbed and undisturbed sample sites in South Carolina coastal plain streams (Paller et al. 1996).

8. Proportion of individuals as benthic invertivores. This metric represents the proportion of darters, madtoms, and sculpins collected in a sample. Because of their

specificity for feeding and reproducing in benthic habitats, benthic invertivores species assess the availability of suitable benthic habitat at a site (Ohio EPA 1987). Many IBI assessments strictly used darter species as a measure of environmental degradation in streams; however, because darters, madtoms, and sculpins are naturally scarce in streams, are benthic species, and exhibit similar feeding and spawning requirements these three groups were combined into a single metric (Paller et al. 1996; Schleiger 2000).

9. Proportion of individuals as non-lithophilic spawners. Non-lithophilic spawners included those species that use a variety of spawning substrates including snags, aquatic vegetation, and coarse organic matter, or spawn in open water (O'Neil et al. 2006). Non-lithophilic spawners include *Cyprinella*, *Ameirus, Noturus*, *Etheostoma*, and *Micropterus* species (Boschung and Mayden 2004; O'Neil et al. 2006). O'Neil et al. (2006) found that the proportion of non-lithophilic spawners was more strongly related to disturbance and discriminated more effectively between least and most-disturbed sites in their Terrapin Creek watershed study than did simple lithophils. Non-lithophilic spawners showed a broader response to disturbance than simple lithophils by incorporating not only the effects of substrate siltation but also more complex disturbances such as changes in flow regime and loss of large woody debris that can adversely affect instream fish habitat (O'Neil et al. 2006).

10. Proportion of individuals as omnivores and herbivores. This metric evaluated the shift in trophic composition of the fish assemblage in degraded streams (Karr et al. 1986). The abundance of omnivores often increases at degraded sites as specialist feeders emigrate or are eliminated because of limited food resources (Hlass et al. 1998).

Examples of generalists include *Nocomis* and *Ameiurus* species (Smogor and Angermeier 1999; O'Neil et al. 2006). This metric also includes those species that feed primarily as herbivores, such as *Campostoma* species, whose increase abundance in a sample is associated with elevated nutrient levels (O'Neil and Shepard 1998).

Metrics 11 and 12 evaluated the abundance and condition of the fish assemblage. A description of each metric in this category is given below.

11. Number of individuals per 100 meters. This metric represented the total number of fish collected over a designated 100 meter stream reach (Ohio EPA 1987; Barbour et al. 1999). Abundance was one of the original Karr metrics (1986) and has been widely employed in IBI applications (Hughes and Oberdorff 1999). Fish abundance is generally assumed to decrease with habitat disturbance (Ohio EPA 1987); however, in some disturbed streams it can lead to higher proportions of tolerant species. Therefore, the scoring criteria were adjusted to avoid rewarding a possible degraded site with a very high catch rate as a result of increased numbers of tolerant individuals (Ohio EPA 1987; Simon 1991). Sites with catch rates <100 individuals or > 400 individuals received a score of 0. Sites with catch rates within an optimum range of 200 to 300 individuals received a score of 100. Sites where catch rates deviated moderately, either above or below the optimum range, received a score of 50.

12. Proportion of individuals with external anomalies plus hybrids. The original metric proposed by Karr et al. (1986) represented the proportion of individuals with deformities, eroded fins, lesions, and tumors (DELTs). The occurrence of unhealthy individuals in a fish assemblage often reflects the health and condition of the assemblage

as whole (O'Neil et al. 2006). However, occurrence of large numbers of diseased individuals is rare except in the most highly degraded streams (Karr et al. 1986). Similarly, hybridization between species is usually indicative of highly disturbed streams (Karr et al. 1986). However, occurrence of hybrid individuals is rare except in the most degraded streams (O'Neil et al. 2006). Most streams exhibit only low to moderate numbers of diseased and hybridized fishes, so I combined the 2 into a single metric to better distinguish between least and most disturbed sites.

Development of Expected Values for IBI Metrics. Expected values are defined as those metric values that would be expected for fish assemblages typically observed in relatively undisturbed Piedmont streams (O'Neil et al. 2006). Fish data from the reference site and from the 5 ADEM Piedmont reference streams as well as IBIs developed for Piedmont streams (Bowen et al. 1996; Schleiger 2000; O'Neil et al. 2006) were used to establish expected values (EV) (Table 7). Often, richness can vary with upstream watershed area (drainage size) with larger streams typically supporting a higher number of species (Fausch et al. 1984; Karr et al. 1996; Osborne et al. 1992; Bowen et al. 1996). Therefore, expected values (EV) were adjusted to account for variation in drainage size in Piedmont streams using the method outlined by O'Neil et al. (2006) (Table 7). Relative abundance is often not influenced by the size of the watershed (GA DNR 2005), so relative abundance metrics were not adjusted to account for differences in watershed area (Table 7).

Development of Scoring Criteria for IBI. The original method of scoring metrics using a trisection of the scoring range has been well-documented (Karr et al. 1986; Ohio

EPA 1987; Fore et al. 1996; Barbour et al. 1999; O'Neil et al. 2006). However, Minns et al. (1994) and Hughes et al. (1998) standardized metrics as percentages of an expected metric value. A similar metric scoring system was developed for this study. Raw metric scores were standardized to a scale of 0-100 by dividing each raw metric score by an expected value for that metric as determined from reference data (Cane Creek and ADEM data) and different studies in Piedmont streams using the IBI (Bowen et al. 1996; Schleiger 2000; O'Neil et al. 2006) and multiplying by 100 (Hughes et al. 1998, Barbour et al. 1999; Compton et al. 2003; Bramblett et al. 2005). A continuous scale was used because it was shown to be more responsive to environmental impacts than categorical scoring (5, 3, 1) used by Karr et al. (1984) (Hughes et al. 1998; McCormick et al. 2001; Compton et al. 2003). Metrics from sites that performed well and scored >100 were set at 100. Metrics from sites that performed poorly and had negative values were set at zero. The equations for calculating standardized metric scores were:

Standardized metric score = (raw metric score/expected metric value) x 100

for positive-response metrics and

Standardized metric score = 100 – (raw metric score/expected metric value) x 100

for negative-response metrics. Positive-response metrics are defined as metrics whose scores are expected to decrease with degradation, and negative-response metrics are those whose scores are expected to increase with degradation (Compton et al. 2003). The final IBI score, on a 0-100 point scale, was the average of the standardized metric scores. Streams were placed into one of six integrity classes based on IBI scores (adapted from Minns et al. 1994 and Dauwalter et al. 2003). The integrity classes for evaluating stream condition were: 0 (no fish), > 0 - < 20 (very poor), 20 - < 40 (poor), 40 - < 60 (fair), 60 - < 80 (good) and > 80 (excellent).

Characterization of Fish Assemblages Using an Index of Well-Being (Iwb)

The Index of Well-Being (Iwb) was calculated for each site for both 2004 and 2005. The Iwb was expressed as:

$$Iwb = 0.5 \ln (N / 100\ m) + 0.5 \ln (B / 100\ m) + H' (no.) + H' (wt.)$$

where

$N / 100\ m$ = number of individuals per 100 m

$B / 100\ m$ = total biomass for all species per 100 m

$H' (no.)$ = Shannon diversity index based on numbers of fish

$H' (wt.)$ = Shannon diversity index based on biomass of fish

In the calculation of the Iwb, any highly tolerant, hybrid, and introduced species were excluded from the relative abundance components of the Iwb, but retained in the calculations for H' as recommended by the Ohio EPA (1987) and GA DNR (2005). Each Iwb score was then equated to a particular stream condition: Excellent, Good, Fair, Poor, and Very Poor. Scoring criteria for the Iwb used were the same as that used by the GA DNR (2005) and Ohio EPA (1987) in their stream bioassessments (Table 8).

Data Analysis

Index of Biotic Integrity. For statistical purposes, the distribution of the 12 metric scores for each site were categorized as k independent groups where $k = 3$. A Kruskall-

Wallis (*K-W*) test was then used to compare the groups of metric scores across sites to determine if differences in median metric scores existed. To determine where group differences occurred, a Steel-Dwass (*S-D*) nonparametric post-test was used. The *S-D* test is the nonparametric equivalent of the Tukey-Kramer HSD test and is used to assess pairwise differences between groups (Zar 1996). Raw metric scores for each individual metric were evaluated using a, an *F*-test to determine if median metric values across sites were statistically equivalent or distinct (US EPA 2006). Differences for all statistical tests were considered significant at $p \leq 0.05$.

Measures of Assemblage Structure and Function. Descriptive measures related to community structure included total abundance (N), species richness (S), species diversity (H'), and evenness (E). These measures were compiled for each site and sampling period. Total abundance (N) was the total number of individuals present in a sample. Species richness (S) was the total number of species present in a sample.

To measure the diversity of the fish assemblages at each site, the Shannon diversity equation (H') was selected. This index weighs species exactly by their relative proportions in a sample, without favoring rare or common species (Peet 1974; Jost 2006). This equation is given as:

$H' = - \sum (n_i/N) \log_e(n_i / N)$ (Shannon and Weaver 1949).

Wilhm and Dorris (1968) interpreted Shannon diversity values as follows: (1) little stress / relatively unimpacted (≥ 3.0), (2) moderate stress / moderately impacted (1.0 – 2.9) and (3) stressful / impact is evident (< 1.0).

Evenness, the distribution of individuals among species in a sample, was calculated using the following equation:

Evenness (E) = $e^{H'}/S$ (Sheldon 1969)

Evenness values range from 0 to 1 and approach 0 as a single species becomes more dominant in the community (Brower and Zar 1984; Ludwig and Reynolds 1988). Although evenness indices can be used to assess the condition of the fish assemblage, comparisons across sites and years must be interpreted with caution because most evenness indices are highly sensitive to the number of species in a sample (Alatalo 1981; Zar 1984; Ludwig and Reynolds 1988).

A chi-squared (X^2) test for association (involving *r x c* contingency tables) was used to examine across-site differences in trophic structure using feeding guilds and tolerance class for each sampling period (Sokal and Rohlf 1987; Tabit and Johnson 2002; Keaton et al. 2005). Trophic structure of the resident fish assemblages at each site was determined based on the percentage of the number of captured individuals represented by each feeding guild, respectively relative to the total number of individuals in the sample. The relative abundance of fishes in each tolerance class was calculated as the number of captured individuals in each tolerance class divided by the total number of individuals in the sample. For the chi-square tests, differences were considered significant at $p \leq 0.05$.

To test for normality of fish abundance data, histograms showing the distribution of abundance data were plotted. If the data was non-normal, a Wilcoxon signed-rank test was used to examine between-year differences in trophic structure (feeding guilds) and relative abundance in each tolerance class (Tabit and Johnson 2002; Keaton et al. 2005).

The Wilcoxon test is useful for between-year comparisons of fish community data when the same localities are sampled over time (fixed-station sampling) resulting in correlated, non-independent samples (McCall 1986; Maceina et al. 1994; Sokal and Rohlf 1987). For the Wilcoxon test, differences were considered significant at $p \leq 0.05$. All statistical analyses were conducted with JMP Version 4.04 Statistical Software (SAS Institute, Inc.) (Sall et al. 2001).

IV. RESULTS AND DISCUSSION

Physical Habitat Assessment

Scores of the visual assessment of physical habitat at each site are presented in Table 9. In terms of habitat quality, the upstream and downstream impacted sites (CH-U and CH-D) were comparable to the reference site (Cane-1), each scoring 191 out of a possible 200 points. Cane-1 scored 195 out of a possible 200 points. The main differences between Cane-1 and CH-U and CH-D were related to the amount of sediment present in the stream channel and the frequency of large, well-developed pools. Because of local clearcutting for timber on the Cane Creek watershed, more sediment deposition was evident at Cane-1, particularly in the deeper pool areas, relative to CH-U and CH-D. Because of the presence of Lake Ogletree, there was little sediment deposition at either CH-U or CH-D. This was because lake water flowing over the spillway or through the siphon (drawing water at about 5 feet below the surface) was typically low in sediments as heavier sediment particles entering the lake with stream water were settling to the lake bottom.

In addition, there were a series of deep pools at Cane-1 that were noticeably absent at both CH-U and CH-D. At CH-U and CH-D, pools were present in each reach but were smaller and shallower than those at Cane-1. In addition, there were only portions of large, deep pools located at the lower ends of both CH-U and CH-D. However, large pools were located upstream and downstream of both sites. At CH-U, there was a long pool upstream of a small ford that crosses the creek at the upper end of the site as well as several large pools located just downstream of the site. At CH-D, there was a very large pool located upstream (known locally as the "pretty hole") as well as a long series of pools located just downstream of the site. Although deep pools were mostly absent within each site, the presence of local pools both upstream and downstream of the sites likely served as refuge and cover for fishes, particularly during low flows. Schlosser (1982) found that stable stream reaches were characterized by large, deep pools containing older age class fishes and larger pool species, particularly catostomids and centrarchids.

In comparing physical habitat between the two impacted sites minor differences were evident. In terms of available fish habitat, there were no undercut bank areas present at CH-D as was observed at CH-U. In addition, there were no large, deep pools present at CH-D needed by older age class fishes and larger pool species (Schlosser 1982). Emergent vegetation including American water willow *Justicia Americana* and river weed *Podostemon ceratophyllum* was evident at both sites. Instream cover in the form of woody debris was more prevalent at CH-U than at CH-D. Both sites were observed to have nearly identical substrate characteristics. Riffles were composed of gravel, medium-sized cobble, and small boulders at both sites. There was much more exposed bedrock in the run areas at CH-D than at CH-U, although medium-sized cobble and small boulders were present similar to that observed in the runs at CH-U. Although habitat scores were the same for both sites, the lack of undercut bank habitat at CH-D was the most noticeable difference between the two sites.

Flow Regime between Lake Ogletree and Martin Marietta Quarry

The hydrographs indicated very similar flow conditions in the channel at both sites (Figure 4). At CH-U, discharge approached 2.0 MGD only during August and September 2004, May 2005 and again in March 2006. CH-D exhibited a similar flow pattern with lowest flows in August and September 2004 and May 2005. Both sites exhibited highest discharge values during the spring 2005 and 2006 as a result of several large spring rain events. During the study period from August 2004 to April 2006, water levels never dropped below 2.0 MGD at either site. Thus, there was consistent water flow in the stream channel such that fish migration was not impeded at any time during the study.

However, both hydrographs, in particular for CH-U, exhibited modified flow conditions common in streams downstream of an impoundment (Poff et al. 1997). These changes in flow have the potential for detrimental effects on stream ecosystems (Poff et al. 1997; Richter and Thomas 2007). Between August 2004 and April 2006, there were a high number of low flow pulses but few high flow pulses (Figure 4). In addition to low flow pulses, periodic high flow pulses are necessary to maintain natural seasonal variability in stream flows to support natural biota (Poff et al. 1997; Richter and Thomas 2007). During rainfall events, water levels in Lake Ogletree were regulating downstream flows into the stream. This dampening of flow amplitude by the lake resulted in reduced high flow pulses and higher base flow variability, as seen on the hydrographs for CH-U and CH-D (Figure 4). Although measures were taken to stabilize base flows downstream of the lake (USFWS 2003); periodic high flows necessary for maintaining natural flow variability were lacking in the study reach (Richter and Thomas 2007). Unfortunately, because no flow data was collected at Cane-1, comparisons in mean daily discharge

(MGD) between flow-regulated sites (CH-U, CH-D) and an unregulated site (Cane-1) could not be made.

Water Chemistry

Water chemistry data for sampling dates in 2004 and 2005 are presented in Table 10. Water chemistry in Chewacla Creek was influenced by both point and nonpoint sources (Webber and Blevins 2001; Webber 2002, 2006). However, downstream of Lake Ogletree water chemistry was influenced primarily by the dynamics of water flow through the lake and by discharge from an Opelika limestone quarry into a tributary flowing into Lake Ogletree (Webber 2002, 2006).

No differences in water chemistry were found at Cane-1, CH-U, and CH-D that indicated potential problems for the local fish fauna. Because of the presence of Lake Ogletree, lake water flowing over the spillway or through the siphon was typically low in nutrients and sediments. This was probably because plankton communities in the lake utilized much of N and P washed into the lake from the upstream watershed. In addition, heavier sediments entering with stream water settled to the bottom of the lake so that soil particles leaving the lake were greatly reduced in quantity (Webber and Blevins 2001; Webber 2002; Webber 2006).

Water temperatures ranged from a low of 12.5^{o}C at Cane-1to a high of 15.5^{o}C at CH-U just downstream of Lake Ogletree (Table 10). The flow of warmer surface water (epilimnion) from the lake over the spillway into the stream in addition to periodic siphoning of warmer, epilimnetic water from the lake into the stream probably caused the higher temperatures at CH-U. Dissolved oxygen (DO) ranged from 7.50 mg/L to 10.8 mg/L across sites (Table 8). DO was highest at Cane-1 in both 2004 and 2005, with lowest DO measurements recorded at CH-U both years. The variation in DO levels across sites was related to water temperature, which explains why Cane-1 had the highest DO and lowest temperature measurements (Horne and Goldman 1994). The highest water temperature and lowest DO measurements occurred at CH-U suggesting that Lake Ogletree greatly influenced hydrological and chemical properties at the site. All sites in 2004 and 2005 had sufficient oxygen at the time of sampling to meet the needs of the resident fish communities.

The pH of Piedmont streams is typically slightly acidic based on the nature of soils (Boyd 2000). The values for pH ranged from 6.74 to 7.12 across sites (Table 10). In 2004 and 2005, Cane-1 was slightly acidic while CH-U and CH-D were slightly alkaline. The pH at all sites during the time of sampling was within the acceptable range of 6.5 – 9.0 for supporting healthy fish communities in 2004 and 2005 (Boyd 2000).

N and P levels were low at all 3 sites in 2004 and 2005 (Table 10). Nitrate (NO_3-N), ammonium (NH_3-N), orthophosphate (O-PO_4) and total phosphorous (T-PO_4) concentrations at the two Chewacla Creek sites were similar to those measured at the Cane Creek site in both 2004 and 2005. NO_3-N levels ranged from 0.012 mg/L to 0.09 mg/L across sites in 2004 and 2005. NH_3-N levels ranged from 0.006 mg/L to 0.095 mg/L across sites in 2004 and 2005. T-PO_4 levels ranged from 0.006 mg/L to 0.02 mg/L across sites in 2004 and 2005. Total phosphorous concentrations at all sites were within the acceptable range of 0.01 to 0.05 mg/L for natural waters at all sites in 2004 and 2005 (Wetzel 2001). O-PO_4 levels were a fraction of the T-PO_4 levels at all sites

Turbidity was variable across sites with a high reading observed at Cane-1 in 2004 (Table 10). However, turbidity levels at CH-U and CH-D were low both years. In

2005, turbidity measured in Cane-1 was close to the low turbidity measurements observed at CH-U and CH-D. The high turbidity but low TSS measured in 2004 at Cane-1 was difficult to explain, suggesting that a rain event prior to sampling had occurred on the watershed upstream of the sampling site (Webber 2006). The low turbidity and TSS levels at CH-U and CH-D both years reflected the influence of Lake Ogletree. Heavier solids settled in the lake and were not discharged downstream.

Alkalinity, conductivity, and hardness were higher at CH-U and CH-D than at Cane-1 both years (Table 10). In fact, total alkalinity and hardness levels at CH-U and CH-D were twice as high as those at Cane-1 both years. Conductivity levels were more than 20 times higher at CH-U and CH-D than at Cane-1 in 2004 and almost twice as high in 2005. The elevated alkalinity, conductivity and hardness measurements may have resulted from discharge from an Opelika limestone quarry into a tributary of Chewacla Creek above Lake Ogletree (Webber and Blevins 2001; Webber 2002, 2006). Water from limestone quarries is relatively high in alkalinity, often exceeding 100 mg/L $CaCO_3$ (Webber 2006). However, these measurements were not high enough to have deleterious effects on the resident fish assemblages at either site. The alkalinity and hardness measurements observed at Cane-1 were typical of relatively undisturbed Piedmont streams (Boyd 2000). The conductivity measurement at Cane-1 was unusually low (5.0 µS) in 2004, suggesting that a rain event occurred prior to sampling in the watershed above the sampling site. Webber (2002, 2006) reported that the natural baseline conductivity at Cane-1 is ~ 50 µS. The conductivity measurement at Cane-1 (63.0 µS) in 2005, although slightly elevated, was closer to conductivity levels typically observed at the site

Fish Data

A total of 1,647 fish representing 7 families and 35 species were collected over the 2-year study from the three sites (Table 11). Cyprinidae (carps and minnows), Centrarchidae (sunfish), Catostomidae (suckers), and Percidae (perches) were the dominant families in all samples, composing 97.3% of the total number of fish. The remaining 2.7% belonged to Ictaluridae (freshwater catfishes), Fundulidae (topminnows and killifishes), and Cottidae (sculpins). Of the total fish captured, 45.8% were centrarchids, 30.2% were cyprinids, 15.1% were catostomids, and 6.2% were percids.

A comprehensive list of fish species collected at all sites with associated tolerance classes, feeding guilds, and breeding guilds is presented in Table 12. Nine species were common to all sites, including the largescale stoneroller *Campostoma oligolepis*, blacktail shiner *Cyprinella venusta*, Alabama hogsucker *Hypentelium etowanum*, speckled madtom *Noturus leptacanthus*, green sunfish, bluegill sunfish *Lepomis macrochirus*, longear sunfish, spotted bass, and speckled darter E*theostoma stigmaeum* (Table 12). Six species, including the weed shiner *Notropis texanus*, spotted sucker *Minytrema melanops*, channel catfish, shadow bass *Ambloplites ariommus*, redear sunfish *Lepomis microlophus*, and largemouth bass were found only at CH-U. The banded sculpin *Cottus carolinae* was found only at CH-D. Six species were unique to Cane-1, including the Tallapoosa shiner *Cyprinella gibbsi*, lined chub *Hybopsis lineapunctata*, striped shiner *Luxilus crysocephalus*, bluehead chub *Nocomis leptocephalus*, rough shiner *Notropis baileyi*, and silverjaw minnow *Notropis buccatus*.

Index of Biotic Integrity (IBI) and Index of Well-Being (Iwb)

Assessment of Cane Creek as a Reference Site. Cane Creek (Cane-1) was chosen as a reference site for this study, based on (1) a high proportion of its watershed in forest (>70%) and correspondingly low proportions in urban/suburban (<1%) and agriculture (< 6%) areas (Webber 2002), (2) a diverse macroinvertebrate assemblage (Bayne et al. 2004), and (3) historically good water quality and physical condition (Webber and Blevins 2001, Webber 2002, 2006). However, because it was the only reference site used for this study, it was necessary to compare fish data collected at the site with data collected from other established Piedmont reference sites. For comparison with Cane-1, ADEM data from designated Piedmont reference sites including Hatchet Creek (HATC-3), Weogufka Creek (WGFC-1), Hurricane Creek (HCR-1), Emuckfaw Creek (EMKT-14) and Cornhouse Creek (CRHR-9) were used (D. Miller, ADEM, unpublished data).

Comparing data sets between Cane-1 and the 5 ADEM reference site, it appeared that Cane-1compared favorably with the ADEM sites in terms of fish species richness and diversity. Cane-1 exhibited high cyprinid richness and diversity similar to data from the ADEM sites. Furthermore, samples from Cane-1 contained high numbers of centrarchid species with relatively low abundance per species, similar to the ADEM samples, indicating an equitable distribution of individuals among species and low dominance, typical of relatively undisturbed streams (GA DNR 2005). Schlosser (1982) noted that balanced, healthy assemblages in streams typically exhibit high cyprinid and centrarchid diversity.

One area where the Cane-1 did not compare as favorably with the ADEM sites was the total number of darter species collected. At Cane-1 only 3 darter species, the speckled darter, the Tallapoosa darter and the blackside darter were collected whereas at the ADEM sites, typically ≥ 5darter species were found, including the blackbanded darter, greenbreast darter E*theostoma jordani*, speckled darter, rock darter *Etheostoma rupestre*, lipstick darter *Etheostoma chuckwachattee*, Tallapoosa darter, muscadine darter *Percina sp.*, and the bronze darter *Percina palmaris*. Despite fewer darter species present, the data from Cane-1 compared reasonably well with ADEM reference data such that the site provided reasonable reference data for measuring environmental degradation at CH-U and CH-D.

Metric Selection for IBI. Reference data from Cane Creek and ADEM sites in addition to previous applications of the IBI was used to select the 12 core metrics for this study. Percent tolerant species, % *Lepomis*, and % benthic invertivores were included as metrics because they were commonly used in other IBI studies and added trophic composition information to the IBI. Commonly used madtom, darter, and sculpin metrics were combined into a single metric, % benthic invertivores, for 2 reasons. First, the scarcity of these fishes in samples at all sites necessitated their combination into a single metric. Second, as benthic fishes they often respond to environmental disturbances similarly, allowing them to be grouped together (GA DNR 2005). Percent DELTs and % hybrids, 2 metrics used frequently in IBI studies, were combined into a single metric because of their scarcity in samples at all sites.

Two commonly used metrics in IBI studies, % insectivorous cyprinids and number of native sunfish species were not included because they correlate with the number of native cyprinid and % *Lepomis* metrics, respectively (O'Neil and Shepard

1998; ADEM 1999; GA DNR 2005),. It was important to select metrics that were not redundant and presented unique information to the IBI index (Dauwalter et al. 2003; Bramblett et al. 2005).

Finally, % non-lithophilic spawners was chosen as a trophic composition metric over the more commonly used % lithophilic spawners metrics. O'Neil et al. (2006) found that the % non-lithophilic spawners metric better discriminated most from least disturbed sites in streams in both small and large watersheds in the Tallapoosa and Coosa river drainages than did % lithophilic spawners. CH-U and CH-D were not heavily impacted by sedimentation based on low TSS and turbidity levels at the 2 sites both years (Table 10); this metric was not very useful in discriminating across sites. Rather, the % non-lithophilic spawners metric was more suitable for my IBI because fishes in this breeding guild exhibit a more diverse array of spawning behaviors as well a greater variety of spawning substrates than lithophilic spawners (Boschung and Mayden 2004). Thus, the % non-lithophilic spawners metric can detect more broad disturbances across sites such as changes in flow regime that reduce habitat diversity resulting in loss of instream fish cover (O'Neil et al. 2006).

Comparison of IBI and Iwb Scores Across Sites. IBI metrics, raw metric scores and IBI scores for Cane-1, CH-U, and CH-D for 2004 and 2005 are presented in Table 13. In addition, IBI scores with associated integrity classes for each site are presented in Table 14. IBI scores ranged from 29.5 to 83.3 across sites in 2004 and from 35.6 to 69.3 across sites in 2005 (Tables 13 and 14). Iwb scores ranged from 6.0 to 13.5 across sites in 2004 and from 6.4 to 12.1 across sites in 2005 (Table 14).

At Cane-1, IBI scores were 83.3 and 69.3 in 2004 and 2005, respectively, placing the site in the "Excellent" integrity class in 2004 and the "Good" integrity class in 2005. Moreover, Iwb scores were 13.5 and 12.1, respectively placing the site in the "Excellent" class both years. The fish assemblage at Cane-1 was characterized by high cyprinid diversity, relatively low abundance of tolerant fishes, presence of intolerant species and high habitat diversity. Shields et al. (1995) found sites with high biotic integrity were associated with greater habitat diversity, particularly at sites with deep pools and abundant woody debris. The high sunfish diversity in Cane-1 was apparently indicative of the presence of deep pools with abundant instream cover. Also, the presence of several intolerant species, lower numbers of tolerant fishes, and no fishes with DELTs or hybrids in Cane-1 suggested minimal site impacts in comparison with CH-U and CH-D. The tolerant species collected in Cane-1 both years (metric score = 40.8 and 0, respectively) included the largescale stoneroller, bluehead chub, striped shiner, longear sunfish, redbreast sunfish, bluegill sunfish and green sunfish (Table 12). However, because these species were observed in relatively low numbers in 2004, their presence most likely represented natural abundance that occurs in relatively undisturbed Piedmont streams rather than a response to some perturbation.

In 2005, IBI and Iwb scores at Cane-1 were lower than in 2004. The decrease in scores of both indices was related to increases in relative proportions of tolerant species, % *Lepomis* species (metric score = 63.3 and 0, respectively), and % omnivores and herbivores (metric score = 25.5 and 29.5, respectively) and decreases in % non-lithophilic spawners (metric score = 83.0 and 60.1, respectively) and number of individuals per 100 m (metric score = 100 and 50.0, respectively). These increases in relative proportions of tolerant fishes, omnivores and herbivores, and *Lepomis* species can indicate a disturbance to a stream; furthermore, degradation of stream habitats often results in lower abundances of fishes in streams (Kovalak 1981; Karr et al. 1986; Schleiger 2000). Schleiger (2000) found that tolerant fishes increased in Georgia Piedmont and Coastal Plain streams with degraded water quality. However, water quality in Cane-1 was characteristic of relatively undisturbed Piedmont streams both years (Boyd 2000) (Table 10). Degraded water quality and physical habitat in a stream often results in a decrease in diversity of the macroinvertebrate food base for fishes (GA DNR 2005). However, Webber (2006) reported high EPT (Ephemeroptera, Plecoptera, Tricoptera) richness (23 taxa) collected at Cane-1 in 2005, suggesting that a diverse macroinvertebrate assemblage was present at the site during this study. Furthermore, there were increases in relative proportions of intolerant fishes and benthic invertivores at Cane-1 in 2005; high proportions of intolerant and benthic species are often characteristic of healthy fish assemblages in streams (Karr et al. 1986; Ahle and Jobsis 1996). Based on water quality, macroinvertebrate, and fish data from the site, it appeared that the lower IBI and Iwb scores were probably related to natural year-to-year variation in fish assemblage structure and function at Cane-1. Although logging did occur in the Cane Creek watershed during this study, the effects on the fish assemblage were not measured. Thus, the impact of logging on the fish assemblage at Cane-1 was unknown.

The high IBI and Iwb scores observed at Cane-1, particularly in 2004, typically occur at "high-quality" sites and illustrated ecosystem stability based on other studies (Karr et al. 1987; Ahle and Jobsis 1996). However, IBI and individuals metric scores were lower in 2005, suggesting either high natural variation in fish assemblage structure and function, or the presence of some disturbance, possibly related to logging in the watershed. The fish assemblage in Cane Creek was characteristic of an unmodified stream system having a natural flow regime with a relatively high number of intolerant species, relatively low abundance of tolerant species, high cyprinid richness, and low numbers of lentic species such as *Lepomis* spp., spotted bass, and largemouth bass. In addition, the redeye bass was present at Cane-1. The redeye bass is considered a relatively intolerant species (Schleiger 2000; Boschung and Mayden 2004), and is typically found in higher numbers than either the spotted bass or the largemouth bass in free-flowing streams (Boschung and Mayden 2004). In contrast, the redeye bass was absent in stream reaches at CH-U and CH-D, both of which have exhibited historically modified flow regimes.

At CH-U, IBI scores were 37.0 and 35.6 in 2004 and 2005, respectively, placing the site in the "Poor" integrity class both years. Moreover, Iwb scores were 9.4 and 7.8, respectively, placing the site in the "Good" class in 2004 and the "Fair" class in 2005. Thus, IBI and Iwb scores showed that CH-U had low biotic integrity when compared with Cane-1 (Tables 13 and 14), even though the physical habitat score at CH-U was similar to that at Cane-1 (Table 9). In comparison with Cane-1, higher abundances of tolerant *Lepomis* species, in particular, bluegill and longear sunfish (metric score = 0 for 2004 and 2005), coupled with no intolerant fishes (metric score = 0 for 2004 and 2005) contributed most to the low IBI scores at CH-D (Table 13). High abundances of *Lepomis and Micropterus* species were present at CH-U compared to Cane-1 and CH-D, probably because of the presence of large pools both upstream and downstream of CH-U. These local pools possibly served as sources of fish colonists to the site, particularly for lentic species like the bluegill and longear sunfish and the largemouth bass. In addition, these pools may have offered refuge for these species during the periodic dewatering of the channel that occurred prior to 2003 such that local extinction was minimized. Once consistent flows returned to the reach, fishes in these pools might have been able to repopulate the site quickly. Other studies have found similar occurrences in which the proximity of pools to stream sites highlighted the role of pools as sources of immigration and refuge for fishes during severe disturbance events such as drought (Meador and Matthews 1992; Hoyt et al. 2001; Medeiros and Maltchick 2001).

In contrast, there were few flow-dependent minnow and darter species collected at CH-U except for the blacktail shiner, and two tolerant species, the blackbanded darter and the largescale stoneroller. Because there was no pre-drought fish data available, it was possible that minnows and darters were naturally scarce at CH-U before dewatering of the study reach occurred. However, the relative absence of these fishes was still surprising considering there was gravel/cobble riffle habitat present at CH-U during the study. This suggested that during the periodic dewatering that occurred prior to 2003, minnow and darter species that perhaps were present at the site disappeared, either desiccating in the dry riffles or becoming trapped in isolated pools and consumed by larger predatory fishes or terrestrial predators. Furthermore, once consistent flows were present in 2003, flow-dependent species appeared to colonize more slowly at CH-U than lentic species such as *Lepomis* spp., spotted and largemouth basses, and the largescale stoneroller.

At CH-D, IBI scores were 29.5 and 43.5 in 2004 and 2005, respectively, placing the site in the “Poor” integrity class in 2005 and the “Fair” integrity class in 2005.

Moreover, Iwb scores were 6.0 and 6.4, respectively, placing the site in the "Poor" class both years. Thus, IBI and Iwb scores indicated that CH-D exhibited low biotic integrity compared with Cane-1 (Tables 13 and 14). Higher IBI scores in 2005 indicated some improvement in the condition of the fish assemblage at CH-D, compared to Cane-1. Higher sucker richness (metric score = 66.7), an increase in the relative proportion of benthic invertivores (metric score = 100) and decreases in relative proportions of omnivores and herbivores (metric score = 76.4) and diseased fishes/hybrids (metric score = 100) contributed most to the improved IBI scores in 2005.

There were several possible reasons for the low IBI and Iwb scores at CH-D. First, the complete absence both years of intolerant species contributed to lower IBI scores at the site. In terms of available habitat, CH-D exhibited no deep pools and little instream cover, especially undercut banks. Uniform shallow habitat, as was observed at CH-D, often results in simple fish assemblages usually dominated by 1 or 2 opportunistic species (Schlosser 1987; Ahle and Jobsis 1996). This pattern was evident at CH-D, as 2 tolerant *Lepomis* species, the bluegill and longear sunfish dominated the assemblage. Like CH-U, there were few minnows and darters present at CH-D either year, with the exception of the largescale stoneroller and blackbanded darter. Riffle habitat at CH-D was less abundant that at CH-U, so the low richness of minnow and darter species might have represented natural scarcity at the site rather than slow recovery from dewatered conditions. Finally, the dewatering of the creek channel at CH-D caused by periodic drought that occurred from 1999 to 2001 and sinkhole development may have affected fish biotic integrity resulting in low IBI scores, compared with Cane-1. An unstable flow environment can disrupt fish assemblage balance resulting in dominance by a single family or trophic group (Gorman and Karr 1987; Ahle and Jobsis 1996).

Unlike Cane-1, CH-U and CH-D had lower abundances of intolerant species, higher abundances of tolerant species in particular the bluegill sunfish, longear sunfish, blackbanded darter, and largescale stoneroller, and lower numbers of flow-dependent minnows and darters. In addition, there were higher numbers of lentic species such as the yellow bullhead, channel catfish, largemouth bass, spotted sucker, black redhorse, redbreasted sunfish, redspotted sunfish, largescale stoneroller, and blacktail shiner, which are more common in impounded streams (Mettee et al. 1996; Boschung and Mayden 2004). The relative absence of flow-dependent fishes such as darters and minnows is characteristic of streams with highly altered flow regimes impacted by drought, impoundment or other disturbance (Mammoliti 2002). Samples collected at all 3 sites indicated that fish assemblage structure at CH-U and CH-D was dramatically different from that at Cane-1, a stream exhibiting a natural flow regime. The fish assemblages at both CH-U and CH-D have been slow to recover to that present in a healthy, unmodified Piedmont stream such as Cane Creek, even though consistent flows have been present in the reach between Lake Ogletree and the quarry since 2003. This suggested that Lake Ogletree has influenced fish assemblage structure and function at CH-U and CH-D and that both sites support fish assemblages characteristic of regulated streams (Mammoliti 2002).

Statistical Assessment of Metrics. In 2004, the *K-W* test showed that group medians were significantly different among the 3 sites (*K-W*, $X^2 = 13.43$, df =2, $p = 0.0012$). The *S-D* test showed that significant pairwise differences existed between the

Cane-1and CH-U ($p \leq 0.01$) and Cane-1 and CH-D ($p \leq 0.01$) but not between CH-U and CH-D ($p > 0.05$) in 2004. In 2005, group medians were not significantly different among the 3 sites ($X^2 = 3.54$, df = 2, $p = 0.170$).

The *K-W* and *S-D* tests showed that in 2004, the groups of metric scores among the 3 sites were different. Based on the *K-W* and *S-D* tests for 2005, it appeared that the groups of metric scores among the 3 sites were more similar. This occurred because of a decline in the number of cyprinids as well as an increase in the number of tolerant and *Lepomis* species at Cane-1, suggesting that the fish assemblage was being impacted, perhaps from increased sediment input to the stream as a result of logging in the upper watershed. Not surprisingly, *K-W* and *S-D* tests showed that fish assemblages at CH-U and CH-D were similar in terms of trophic structure. The main differences in the fish assemblages between CH-U and CH-D were in terms of total abundance and species richness. Samples from CH-U contained higher species richness and overall abundance of fishes than at CH-D suggesting that proximity to Lake Ogletree, and differences in physical habitat contributed to differences in the fish assemblages between the sites.

F-test results for raw metric scores showed that 5 of the 12 metrics effectively discriminated among the 3 study sites (Table 15), including 3 species richness and composition metrics (number of native fish species, number of native cyprinid species, number of intolerant species) and 2 trophic composition metrics (% top carnivores, % benthic invertivores). Two additional metrics (number of native round-bodied sucker species, % non-lithophilic spawners) also gave good discrimination among the 3 sites; however, the differences in group means were close but not statistically significant ($F_{2,3}$ =

8.96, $p = 0.054$ and $F_{2,3} = 0.081$, $p = 0.081$). Two trophic composition metrics and both fish abundance and condition metrics discriminated poorly among the 3 sites ($p > 0.05$). Based on *F*-test results, it was evident that species richness and composition metrics (number of native fish species, number of native cyprinid species, number of intolerant species) contributed the most to the high variation in IBI scores among the 3 sites. At CH-U and CH-D, the numbers of cyprinid species were low compared to Cane-1. Also, the relative abundance of *Lepomis* species, particularly, bluegill and longear sunfish, was much higher compared to the reference site.

These above differences were likely related to the modification of the natural flow regime in the study reach in Chewacla Creek compared with the natural flow regime in Cane Creek. Based on the data from this study between 1999 and 2001, the fish assemblages at CH-U and CH-D had very few riffle-dwelling cyprinids and high abundances of *Lepomis* species occupying the small, isolated pools that were present at both sites during the dewatering of the creek channel. Larimore et al. (1959) and Deacon (1961) also found similar distributions of stream fishes with low numbers of cyprinid species and higher numbers of more lentic fishes including *Lepomis* species in study streams following drought and in regulated streams. As consistent flows have been reestablished in my study reach since 2003, habitat generalists such as bluegill, longear and other *Lepomis* species capable of movement from permanent pools quickly repopulated both sites. Species that occupied restricted habitats, such as cyprinids, have been slow to increase in numbers at both CH-U and CH-D now that consistent flows are present in the creek channel between Lake Ogletree and the quarry.

The IBI and Iwb are useful tools for measuring biotic integrity in streams because they summarize a large amount of data into a single index value (Hughes 1990). The Iwb may have overestimated biotic integrity at CH-U, which was characterized by high abundance and species richness, but also high dominance, low evenness, and low diversity, because of the Iwb's high dependence on abundance and biomass data (Table 16). In contrast, IBI scores rated biotic integrity lower than Iwb scores at CH-U both years, suggesting that the IBI better reflected the high dominance and overall low trophic diversity at the site. At CH-D, low Iwb scores both years reflected the low abundance and biomass at the site; however, the IBI rated biotic integrity higher than the Iwb in 2005 suggesting moderate improvement in the fish assemblage at the site. The IBI may have rated biotic integrity at CH-D higher than the Iwb because it was more sensitive to the trophic composition of the fish assemblage.

The IBI offered one major advantage over the Iwb in measuring biotic integrity at each site. Although the IBI measured both abundance and species richness, it also incorporated functional aspects of the fish assemblage including breeding guilds, feeding guilds, and tolerance classes, which the Iwb did not. As a result, the IBI was perhaps more sensitive to fish assemblage response to disturbance than the Iwb and probably offered a truer reflection of fish assemblage condition at each site. However, the Iwb incorporated biomass, an important component of the fish assemblage. Because these indices incorporated different aspects of the fish assemblages, their combined use appeared to accurately assess fish assemblage condition across sites and years.

Analysis of Fish Assemblage Structure Across Sites and Years

Total abundance (N) varied considerably across sites and years, ranging from 114 to 537 captured individuals (Table 16). Species richness (S) varied across sites and years, ranging from 13 to 21 captured species in samples (Table 16). At Cane-1, total abundance decreased considerably from 327 to 114 captured individuals between 2004 and 2005 whereas species richness was the same both years (19 species). It was unclear exactly why there was such a large decline in total numbers of fishes collected at Cane-1 in 2005. Sampling methodology was the same both years, and there was no difference in collecting efficiency. The data from Cane-1 suggested that the large between-year difference in abundance represented natural variation in the fish assemblage rather than a response to disturbance.

At the upstream site (CH-U), total abundance and species richness decreased from 537 to 332 captured individuals, and 21 to 17 species, respectively, at CH-U. The high abundance at CH-U, particularly in 2004, resulted from high numbers of fishes commonly found in regulated streams downstream of an impoundment including the largescale stoneroller, blacktail shiner, spotted sucker, black redhorse, bluegill sunfish, and longear sunfish (Table 12). The presence of these species in relatively high numbers suggested that Lake Ogletree was influencing the fish assemblage at CH-U. In 2005, the decline in total abundance and species richness at CH-U most likely represented natural variation in the fish assemblage rather than a response to disturbance.

At the downstream site (CH-D), total abundance decreased from 186 to 151 captured individuals, whereas richness increased slightly from 12 to 13 species in 2005. The low abundance and richness at CH-D perhaps was indicative of an unhealthy fish assemblage and reflected the site's low habitat diversity, particularly the lack of undercut bank and large, deep pools. Furthermore, abundance and richness changed little from 2004 to 2005, suggesting that overall recovery of the fish assemblage from the dewatering of the channel has been slow, even though consistent flows were present in the study reach both years.

Shannon diversity (H′) values were relatively low across sites and years and there was little difference in species diversity across sites (Table 16). At Cane-1, H′ increased slightly from 2.35 to 2.54 between 2004 and 2005. This increase occurred because of the decline in total abundance in 2005 but richness remained the same as in 2004. Relatively low H′ values for Cane-1 was surprising given the high IBI and Iwb scores both years of the study. There were no known impacts that could account for the relatively low H′ values unless continued logging in the Cane Creek watershed impacted the fish assemblage, which seemed unlikely. Furthermore, because of the stable flow regime present at Cane-1, there was high habitat heterogeneity and abundant fish habitat in the form of glide riffles, deep pools, undercut bank areas and instream woody debris. In addition, Webber (2006) found a highly diverse macroinvertebrate assemblage at the site, suggesting that Cane-1was relatively undisturbed even with logging occurring in the watershed.

There was a decline in H′ at CH-U between 2004 and 2005, from 2.38 to 1.98, respectively, indicating a decline in fish assemblage at the site. This decrease was most likely related to the decline in both abundance and richness in 2005. At CH-D, there was no change in H′ from 2004 to 2005 indicating slow recovery of the fish assemblage from altered flow conditions that occurred periodically from 1999 to 2001. However, because of the lack of pre-drought fish data for CH-U, there was also the possibility that the low H′ values were indicative of a naturally sparse fish assemblage at the site.

Based on the interpretation of Shannon index (H′) values by Wihlm and Dorris (1968), fish assemblages at all 3 sites were moderately stressed or impacted. This contradicted the IBI and Iwb results, which showed that both CH-U and CH-D exhibited highly stressed fish assemblages compared with Cane-1. However, the Shannon diversity index did not incorporate functional attributes of the fish community such as trophic structure and tolerance class and, perhaps, was a less reliable indicator of fish assemblage condition across sites.

There was relatively little difference in evenness (E) values among sites (Table 16). Surprisingly, E values were relatively low across sites, particularly at Cane-1 in 2004. Pielou (1966) noted that evenness values < 0.6 are typical for stressed biotic communities. This indicated that the fish assemblage at Cane-1 was moderately stressed in 2004 but less stressed in 2005 which contradicted IBI and Iwb scores at the site. In 2004, E values at CH-D indicated that the fish assemblage was only slightly stressed; however, IBI and Iwb scores for the site were indicative of a highly stressed fish assemblage. Also, E values were relatively high at CH-D which was contradictory to the very low H′ values (1.97 and 2.05, respectively). It was unclear exactly why CH-D exhibited low diversity but relatively high evenness both years. In 2005, evenness was highest at Cane-1 because abundance was lower but richness was the same as in 2004, suggesting a more equitable distribution of species present at the site in 2005.

There was no clear explanation for the relatively low H′ and E values both years and the low abundance in 2005 at Cane-1. Despite these low values, IBI and Iwb scores clearly showed that Cane-1 had the healthiest and most balanced fish assemblage of the three sites both years. There had been no alterations to the natural flow regime, so fish assemblage structure at Cane-1 had been largely determined by instream physical habitat. The greater habitat heterogeneity and diverse macroinvertebrate assemblage at Cane-1, compared with CH-U and CH-D, likely explained the healthier and more balanced fish assemblage present at Cane-1. Studies by Schlosser (1982, 1987, 1990), Capone and Kushlan (1991) and Hoyt et al. (2001) all reported that healthy fish assemblages were often found in stream reaches with greater diversity of habitat including deep pools, swift-flowing gravel/cobble riffles, and available instream cover in the form of woody debris, such as that present at Cane-1.

Fish assemblage structure at CH-U appeared to have been largely structured by the presence of Lake Ogletree, and appeared typical of fish assemblages located downstream of an impoundment (Mammoliti 2002). In terms of overall abundance and richness, the fish assemblage at CH-U was dominated by species found in regulated streams, in particular the blacktail shiner, spotted sucker, bluegill sunfish, longear sunfish, black redhorse and largescale stoneroller. With the exception of large, deep pools, habitat diversity at CH-U was comparable to that in Cane Creek. However, during the periodic drought that occurred from 1999 to 2001, there was no water flowing throughout most of the upstream reach and the channel was almost completely dry with the exception of small, isolated pools (Cliff Webber, AU Fisheries, personal communication). It was very likely that during dewatered conditions in the channel, species that favor lentic waters survived in the large pools both upstream and downstream of the study reach and then repopulated the site once flows were reestablished. In contrast, minnows and darters, fishes that require swift-flowing water, disappeared and have not repopulated the site as quickly despite the presence of gravel/cobble riffle habitat.

CH-D was in a section of Chewacla Creek where numerous sinkholes were present, and, thus, during dry periods this reach was often completely dewatered. Also, habitat diversity at the site was low, lacking deep pools and undercut banks. Once consistent flows were reestablished in 2003, fish assemblage data for CH-D suggested that the assemblage had not recovered to that expected for an undisturbed stream, such as Cane Creek. Schlosser (1982) noted that streams with low habitat diversity, particularly those lacking deep pools often have fish assemblages with low abundance and richness. Other studies related richness and diversity to availability of pool habitat (Evans and Noble 1979; Meffe and Sheldon 1988; Ebert et al. 1991; Shields and Hoover 1991; Shields et al. 1994). Deep pools are important because they serve as refuge for many fish species during drought and as sources of dispersal migration when normal flows return to a stream (Medeiros and Maltchik 2001).

Analysis of Fish Assemblage Function Across Sites

Feeding Guilds. There were significant differences in feeding guild composition among the 3 sites in 2004 ($X^2 = 266.99$, df = 6, $p \leq 0.001$) and 2005 ($X^2 = 186.10$, df =8, $p \leq 0.001$) (Figure 6). However, between-year differences were not significant at any site (Wilcoxon test, $p < 0.001$, Figure 6). At Cane-1, invertivores and insectivores were the dominant feeding guilds both years whereas invertivores were the dominant feeding guild both years at CH-U and CH-D.

Cane-1 was dominated by insectivorous cyprinids in 2004 with invertivores, omnivores, herbivores and top carnivores in relatively even proportions (Figure 6). Insectivorous cyprinids are trophic specialists who feed at all levels of the water column, so high cyprinid richness often indicates a diverse aquatic macroinvertebrate assemblage is present in a stream, and, thus, a healthy fish assemblage (Niemela et al. 1999; GA DNR 2005). Webber (2006) reported high aquatic macroinvertebrate diversity at Cane-1 with total EPT (Ephemeroptera, Plecoptera, Tricoptera) taxa of 23 species in 2006 whereas EPT richness was 13, 15, and 16 taxa, respectively, in sites located in the Chewacla Creek study reach. This might explain the higher trophic complexity of the fish assemblage at Cane-1, as fishes there were exploiting a more diverse macroinvertebrate food base, compared to CH-U and CH-D. In addition, top carnivores were found in higher proportions at Cane-1 than at either CH-U or CH-D. An abundance of top carnivores (> 5%) can be indicative a healthy and trophically diverse fish community (Karr et al. 1986; GA DNR 2005). However, an overabundance of top carnivores (> 20%) can be indicative of stream degradation (GA DNR 2005). The relative proportion of top carnivores was not > 15% at any site in either 2004 or 2005. Their presence in higher proportions at Cane-1 possibly reflected the more abundant pool habitat at the site, compared to CH-U and CH-D. Schleiger (2000) reported higher numbers of top carnivores in lower Piedmont streams with abundant pool habitat.

In 2005, there was a shift in feeding-guild dominance from insectivores to invertivores at Cane-1. It was unclear what may have contributed to this trophic shift at Cane-1 in 2005. Often, shifts in trophic structure from highly specialized feeders such as insectivores to more generalist feeders are caused by a decrease in the diversity of the food base in a stream (Niemela et al. 1999; GA DNR 2005). However, it appeared that a relatively diverse macroinvertebrate assemblage was present at Cane-1, as a high number of EPT taxa were observed in 2005 (Webber 2006). Although logging did occur in the Cane Creek watershed during the study, water quality, macroinvertebrate and fish data from Cane-1 suggested minimal impacts to the stream. Thus, it appears that the trophic shift that occurred at Cane-1 in 2005 represented natural year-to-year variation rather than a response of the fish assemblage to disturbance.

At CH-U and CH-D, trophic structure was characterized by dominance by a single feeding guild, invertivores. Insectivores, omnivores, and herbivores were observed in much lower proportions relative to invertivores at both sites. Typically, a high proportion of invertivores in a sample is indicative of a healthy fish community (Niemela and Feist 2002). However, the high abundance of invertivores in samples collected at CH-U and CH-D both years was the result of the dominance of two *Lepomis* species, bluegill and longear sunfish, which are habitat generalists (Boschung and Mayden 2004). An aquatic assemblage dominated by *Lepomis* species can be indicative of poor diversity of the macroinvertebrate food base (O'Neil and Shepard 1998; GA DNR 2005), as Webber (2006) reported for sites in Chewacla Creek between Lake Ogletree and the quarry. As food resources become less reliable in degraded stream habitats, habitat generalists often become the dominant members of the fish assemblage because they are capable of exploiting a less diverse food base (Karr et al. 1986; GA DNR 2005). Thus, the high abundance of invertivores relative to other feeding guilds at CH-U and CH-D suggested degraded fish assemblages were present both years, compared to the fish assemblage at Cane-1.

Lepomis species are highly resilient to stream disturbance and can thrive in degraded stream environments (Detenbeck et al. 1992). Disturbed stream sites often exhibit simple community structure with one or several species in a single family dominating the fish assemblage at the expense of species in other families (Schlosser 1987; Ahle and Jobsis 1996). Bluegill and longear sunfish were probably more abundant relative to other species at CH-U and CH-D because there was no predation pressure on them from piscivorous *Micropterus* species such as the spotted and largemouth bass. There were higher numbers of piscivorous predators such as the spotted and largemouth bass at CH-U because of the close proximity of Lake Ogletree; however, their numbers relative to *Lepomis* sunfish were so small they likely had little influence on populations of the smaller-sized *Lepomis* sunfish. The absence of large piscivorous fishes at the downstream site was likely related to lack of suitable habitat, particularly deep pools, and undercut banks, and woody debris to serve as cover for these ambush predators (Boschung and Mayden 2004).

Tolerance Classes. The relative abundance of tolerance classes was statistically different across sites in 2004 ($X^2 = 279.32$, df = 6, $p \leq 0.001$) and 2005 ($X^2 = 117.08$, df = 6, $p \leq 0.001$) (Figure 7). However, between-year differences did not differ significantly at any site (Wilcoxon test, $p < 0.05$, Figure 7). During both years of this study samples were dominated by sensitive and intolerant species. Species designated as intolerant observed at Cane-1 included the lined chub, Tallapoosa shiner, redeye bass and the blackside and Tallapoosa darters. Six tolerant species including the striped shiner, largescale stoneroller, green sunfish, longear sunfish, bluegill sunfish and yellow bullhead were observed at Cane-1 both years. However, numbers of these species most likely represented natural abundance in the fish assemblage rather than a response to some disturbance in the watershed. The higher numbers of intolerant and moderately intolerant fishes and lower numbers of tolerant fishes, compared to CH-U and CH-D, suggested a healthier fish assemblage at Cane-1 both years.

At CH-U, tolerant species were numerically dominant both years, and moderately intolerant species were the second most abundant tolerance class. The relative abundance of intolerant species at CH-U was low relative to other tolerance classes both years. The high proportions of tolerant species such as the largescale stoneroller, spotted sucker, yellow bullhead, bluegill sunfish, longear sunfish and blackbanded darter suggested that these species quickly repopulated the site once normal flows were reestablished in the creek channel. In contrast, even with normal flows, moderately intolerant and intolerant minnow and darter species appear to be colonizing more slowly at CH-U. There was no change in numbers of tolerant and pioneer fishes, or in number of moderately intolerant and intolerant fishes at CH-U from 2004 to 2005. This result suggested little to no recovery of the fish assemblage from dewatered conditions at the site, as IBI and Iwb scores both years indicate. Despite the presence of consistent flows in the channel since 2003, it appeared that Lake Ogletree influenced fish assemblage structure and function at CH-U more than at CH-D.

At CH-D, tolerant species including the largescale stoneroller, bluegill sunfish, longear sunfish and blackbanded darter were numerically dominant both years. Pioneer species such as the largescale stoneroller, and blackbanded darter were the second most abundant tolerance class in 2004, with moderately intolerant species the second most abundant tolerance class in 2005. There were no intolerant species collected at CH-D in either 2004 or 2005. Tolerant and pioneer species were present in fewer numbers in 2004, suggesting that fish assemblage condition had improved slightly during the two-year study. Some recovery of the fish assemblage from dewatered conditions had taken place, but perhaps not to the level present at Cane-1. This assertion was supported by the IBI and Iwb scores for CH-U, showing improvement in fish assemblage condition in 2005.

Summary. Results from my study showed that fish assemblages at two sites in Chewacla Creek between Lake Ogletree and the quarry (CH-U and CH-D) were highly stressed in comparison with the fish assemblage at the reference site in Cane Creek (Cane-1). Several natural and anthropogenic factors have greatly influenced fish assemblage structure and function at CH-U and CH-D, including: (1) drought that occurred periodically between 1999 and 2001, (2) sinkhole development in the channel as a result of quarry mining in the watershed, and (3) the presence of Lake Ogletree. Both CH-U and CH-D were dominated by tolerant species commonly found in disturbed streams, in particular two *Lepomis* species, the bluegill and longear sunfish. These species quickly colonized both sites and became dominant once consistent flows were reestablished in 2003. In contrast, flow-sensitive minnow and darter species were slow to colonize both sites once consistent flows were reestablished.

The fish assemblage at Cane-1 was characteristic of a relatively undisturbed Piedmont stream with a stable flow regime, good instream habitat diversity and a relatively undisturbed watershed. In terms of fish assemblage condition, the site compared well with other Piedmont reference streams and thus served as a high quality site against which fish assemblages at more disturbed sites could be measured.

In comparison with Cane-1, both CH-U and CH-D have exhibited slow recovery from dewatered conditions in the study reach. Despite the presence of consistent flows in the stream channel since 2003, fish assemblages at both sites were similar to those found in highly modified or degraded streams. Higher IBI and Iwb scores at CH-D in 2005 suggested that the fish assemblage had recovered slightly from dewatered conditions, but not to that present at Cane-1. There was no recovery from dewatered conditions at CH-U, based on similar IBI and Iwb scores in both years. Although fish assemblage structure at CH-U was characterized by high abundance and species richness, a large proportion of the total abundance was tolerant species such as longear and bluegill sunfish and other lentic fishes commonly found in regulated streams. Despite the presence of consistent flows at CH-U, the fish assemblage continues to be influenced by the presence of Lake Ogletree.

Table 1. Index of biotic integrity (IBI) metrics originally developed by Karr et al. (1984) to assess stream fish assemblages in the midwestern United States.

Category	Metric
Species richness and composition	Total number of fish species
	Number and identity of darter species
	Number and identity of sunfish species
	Number and identity of sucker species
	Number and identity of intolerant species
	Proportion of individuals as green sunfish
Trophic composition	Proportion of individuals as omnivores
	Proportion of individuals as insectivorous cyprinids
	Proportion of individuals as piscivores
Fish abundance and condition	Number of individuals in sample
	Proportion of individuals as hybrids
	Proportion of individuals with disease, tumors, fin damage, and skeletal anomalies

Table 2. Candidate IBI metrics evaluated for use in Chewacla and Cane creeks, Lee County, Alabama. *DELTS includes fishes with diseases, erosions, lesions, and tumors Barbour et al. (1999).

Category	Metric
Species richness and composition	Number of native fish species
	Number of darter species
	Number of native sunfish species
	Number of native round-bodied sucker species
	Number of intolerant species
	Number of native centrarchid species
	Number of native minnow species
	Number of native benthic insectivore species
	Number of benthic invertivores
	Number of *Lepomis* species
	Number of benthic fluvial specialists
	Number of native families
Trophic composition	Proportion of individuals as green sunfish
	Proportion of individuals as omnivores
	Proportion of individuals as insectivorous cyprinids
	Proportion of individuals as piscivores
	Proportion of individuals as green sunfish + yellow bullheads
	Proportion of individuals as stonerollers
	Proportion of individuals as invertivores
	Proportion of individuals as benthic invertivores
	Proportion of individuals as omnivores + herbivores
	Proportion of individuals as lithophilic spawners
	Proportion of individuals as non-lithophilic spawners
	Proportion of individuals as *Lepomis* species
	Proportion of individuals as bluegill sunfish
	Proportion of individuals as pioneer species
	Proportion of individuals as benthic fluvial specialists
	Ratio of generalist feeders to specialist feeders
Fish abundance and condition	Number of individuals per 100 meters
	Proportion of individuals as hybrids
	Proportion of individuals with DELTs*
	Proportion of individuals with DELTs + hybrids
	Proportion of individuals with DELTS + hybrids + exotics
	Biomass (g) per 100 meters

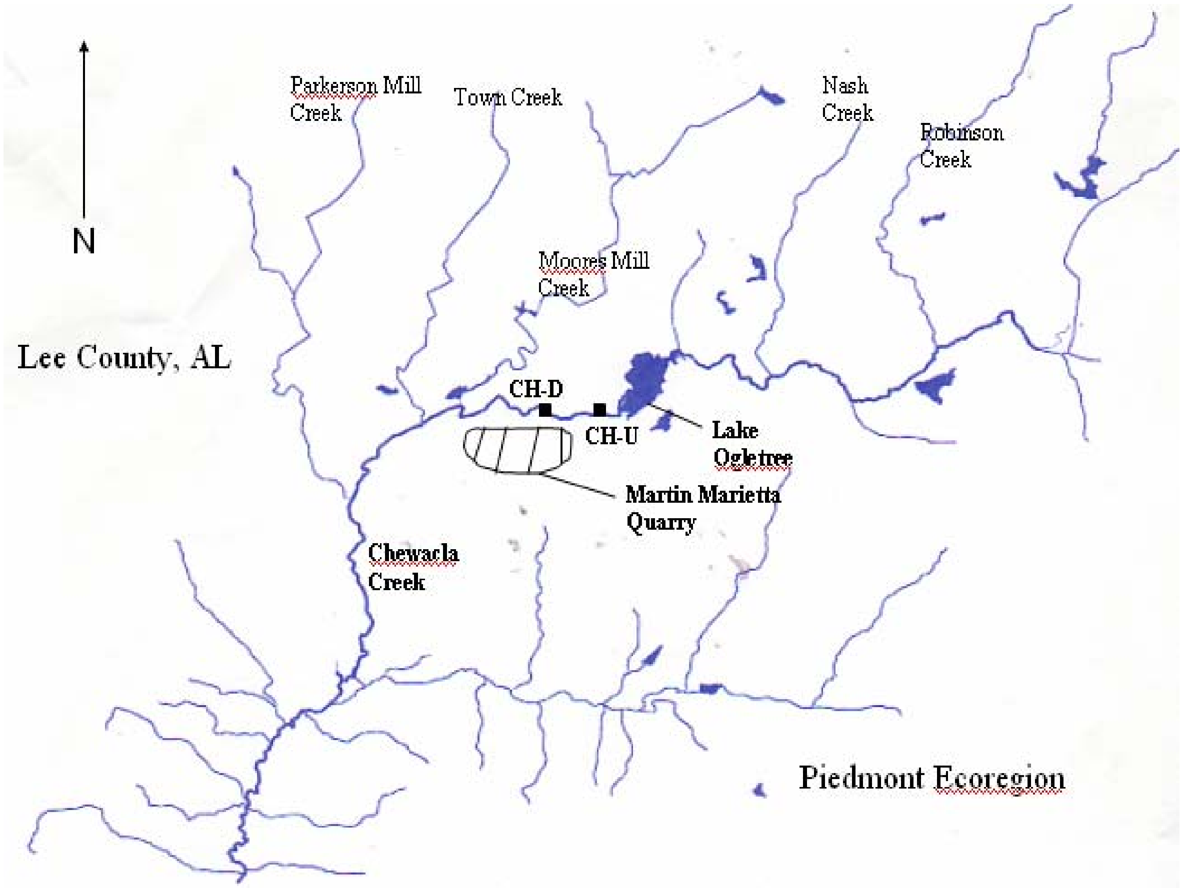


Figure 1. Map showing sampling stations on Chewacla Creek for 2004 and 2005. The locations of the Martin Marietta quarry and Lake Ogletree are included. Sampling stations (CH-U and CH-D) are shown in bold.

Table 3. Physical characteristics measured in Chewacla (CH-U, CH-D) and Cane (Cane-1) creek sites, Lee County, Alabama.

Station	County	Location	Watershed Area (km^2)	Mean Elevation (m)	Mean Stream Width (m)	Mean Depth (m)	Mean Velocity (m/s)
CH-U	Lee	N 32.54702 W 85.45155	90.7	167.1	8.7	0.25	0.17
CH-D	Lee	N 32.54846 W 85.46910	116.5	138.1	7	0.33	0.14
Cane-1	Lee	N 32.63252 W 85.68861	17	172.1	10.9	0.34	0.13

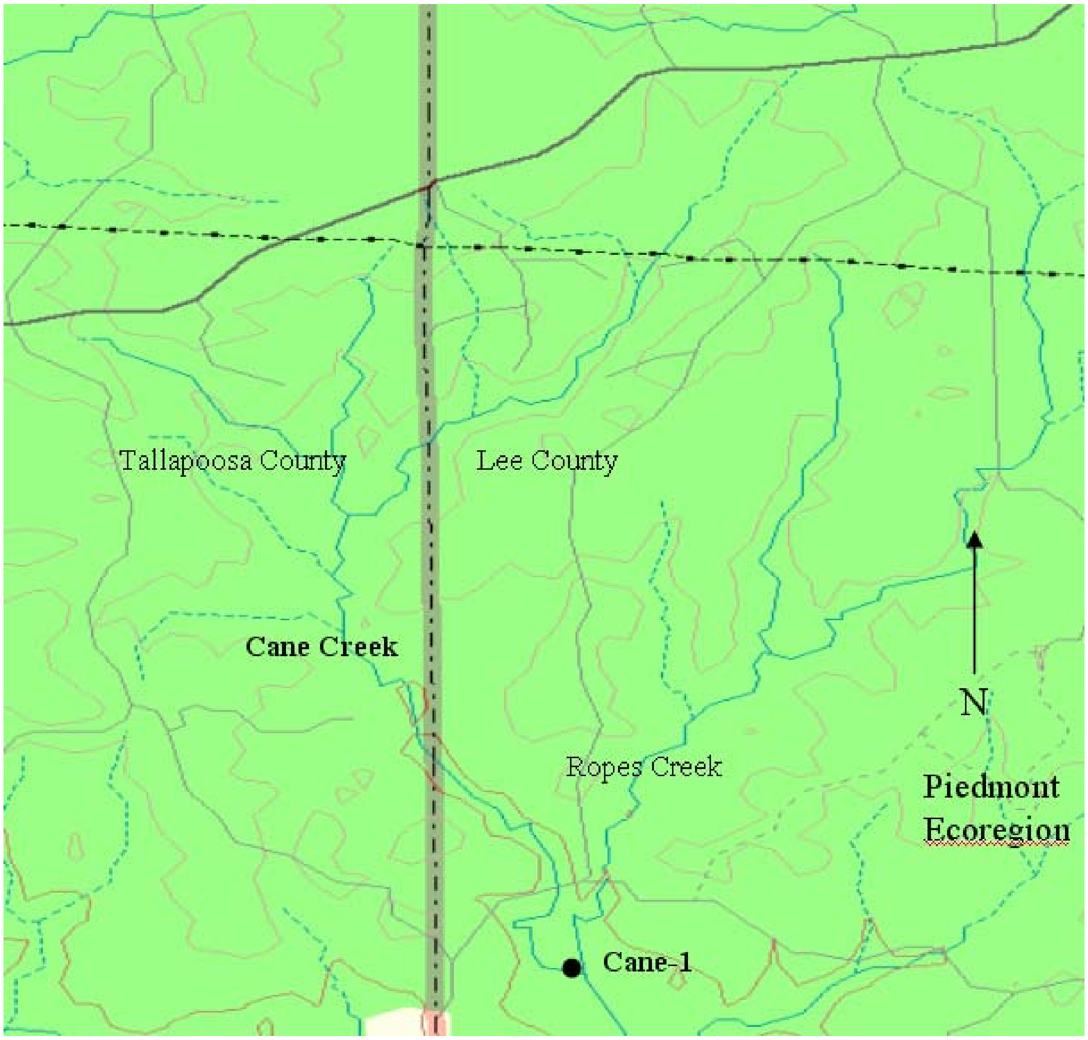


Figure 2. Map showing sampling station in Cane Creek for 2004 and 2005. Sampling station (Cane-1) is shown in bold.

Table 4. Metrics used to assess physical habitat quality at Chewacla and Cane creek sites, Lee County, Alabama during the fall 2006.

Category	Metric
Epifaunal Substrate/Available Cover	Epifaunal substrate/Available cover
	Embeddedness
	Velocity/Depth Regime
	Sediment Deposition
Channel Morphology	Channel flow status
	Channel Alteration
	Frequency of Riffles (or Bends)
Riparian and Bank Structure	Bank stability
	Bank Vegetative Protection
	Riparian Vegetative Zone Width

Table 5. Scoring criteria used to visually estimate physical habitat quality at sites in the Chewacla and Cane Creek watersheds during the fall 2006. A total score is tabulated and compared with the reference stream (Barbour et al. 1999).

Category	Excellent	Good	Fair	Poor
Epifaunal Substrate/Available Cover				
Substrate and cover	20-16	15-11	10-6	5-0
Embeddedness	20-16	15-11	10-6	5-0
Velocity/Depth regime	20-16	15-11	10-6	5-0
Sediment deposition	20-16	15-11	10-6	5-0
Channel Morphology				
Channel flow status	20-16	15-11	10-6	5-0
Channel alteration	20-16	15-11	10-6	5-0
Frequency of riffles (or bends)	20-16	15-11	10-6	5-0
Riparian and Bank Structure				
Bank stability	10-9	8-6	5-3	2-0
Vegetative protection	10-9	8-6	5-3	2-0
Riparian zone width	10-9	8-6	5-3	2-0

Table 6. Assessment used to evaluate physical habitat quality at all sites in Chewacla and Cane creeks during the fall 2006 (Barbour et al. 2006).

Assessment Category	Percent Comparability
Comparable to Reference Site	≥ 90%
Supporting[1]	75-88%
Partially Supporting	60-73%
Non-Supporting	≤ 58%

[1]Potential to support an acceptable assemblage of fishes.

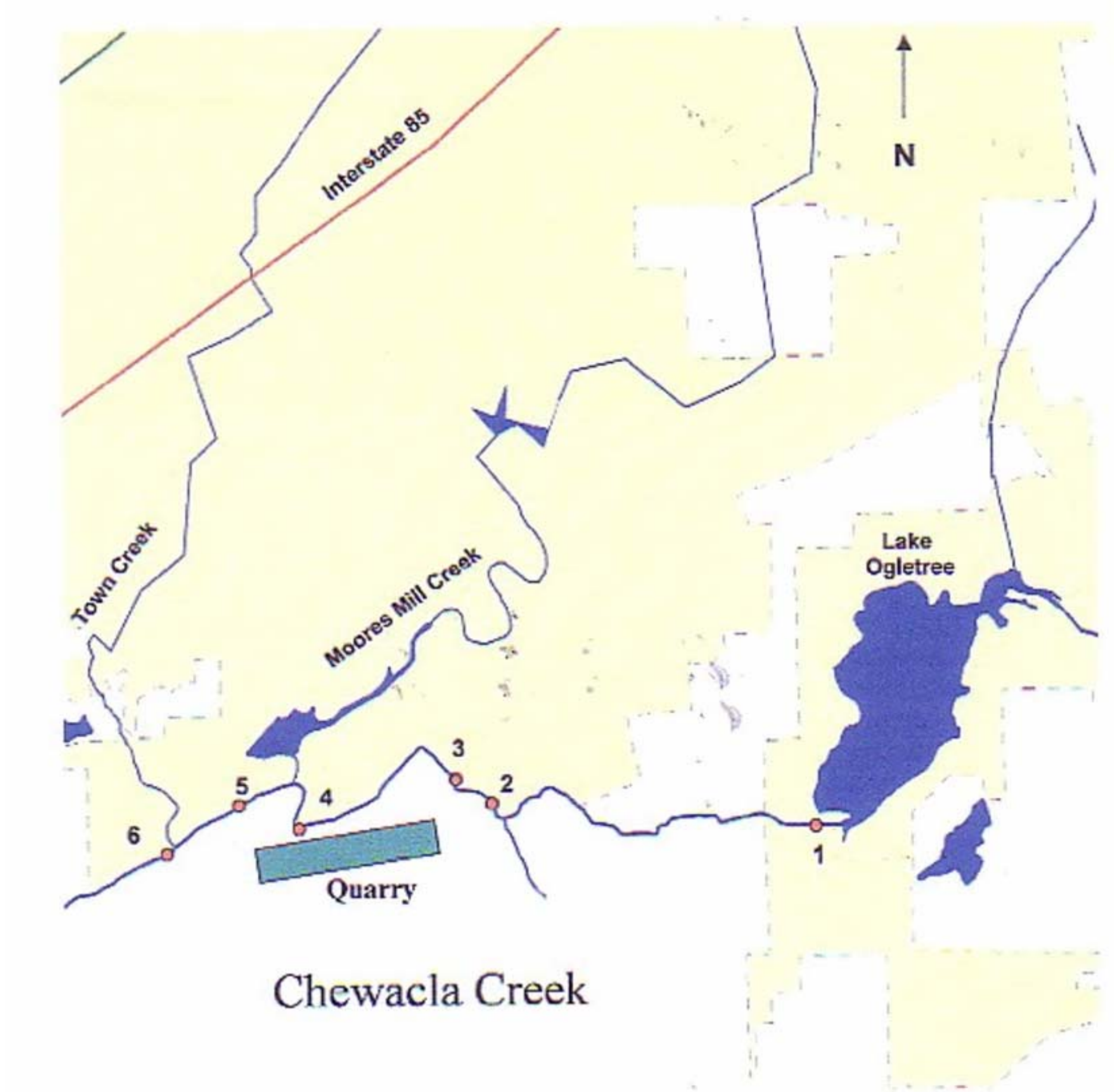


Figure 3. Stations for measuring flow in Chewacla Creek between Lake Ogletree and Martin Marietta quarry. Stream gage stations 1-4 are operated and maintained by the AU Department of Fisheries. Station 5 is a USGS stream gage downstream from the confluence of Moores Mill and Chewacla creeks. Station 6 is a bioassessment site, not a stream gage for measuring flow.

Table 7. IBI metrics and scoring criteria developed for Chewacla (CH-U, CH-D) and Cane (Cane-1) creek sites, Lee County, AL.

	Metric	Watershed Size (km^2)	Expected Value (EV)[1]	Score Calculation
Species richness and composition				
Number of native fish species	#NAT	< 26	17	(#NAT/EV)x100
		26-130	21	
		> 130	25	
Number of native cyprinid species	#CYP	< 26	5	(#CYP/EV)x100
		26-130	7	
		> 130	8	
Number of native round-bodied sucker species	#RBS	< 26	2	(#RBS/EV)x100
		26-130	3	
		> 130	3	
Number of intolerant species	#INT	< 26	2	(#INT/EV)x100
		26-130	4	
		> 130	6	
Trophic composition				
% tolerant	%TOL	all sizes	0.10	100-(%TOL/EV)x100
% top carnivores	%TC	all sizes	0.04	(%TC/EV)x100
% *Lepomis*	%LEP	all sizes	0.30	100-(%LEP/EV)x100
% benthic invertivores	%BI	all sizes	0.20	(%BI/EV)x100
% non-lithophilic spawners	%NLS	all sizes	0.35	(%NLS/EV)x100
% omnivores and herbivores	%OH	all sizes	0.20	100-(%OH/EV)x100
Fish abundance and condition				
Number of fishes per 100 m	#Sample	all sizes	200-300	100
% DELTs + hybrids	%DH	all sizes	0.005	100-(%DH/EV)x100

[1]EV = metric value expected for relatively healthy Piedmont streams. Expected values were determined using fish data from Cane Creek, ADEM reference data and previous applications of the IBI.

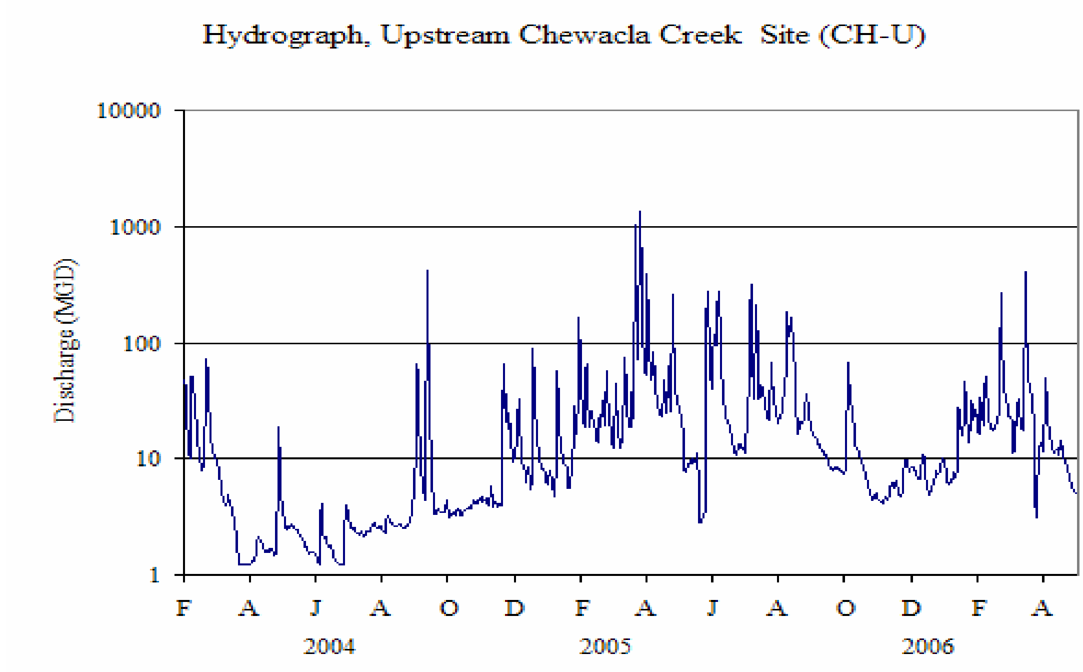
Table 8. Index of Well-Being (Iwb) scoring criteria and integrity classes for wadeable Piedmont streams. Adapted from GA DNR (2005) and Ohio EPA (1987).

Iwb Score	Watershed Size (km^2)	Integrity class
≥ 8.1	< 40	Excellent
≥ 9.6	≥ 40	
8.1 - ≥ 7.3	< 40	Good
9.6 - ≥ 8.6	≥ 40	
7.3 - ≥ 5.7	< 40	Fair
8.6 - ≥ 6.6	≥ 40	
5.7 - ≥ 4.9	< 40	Poor
6.6 - ≥ 5.6	≥ 40	
< 4.9	< 40	Very poor
< 5.6	≥ 40	

Table 9. Physical habitat assessment metrics, individual metric and total scores, percent comparison, and final habitat assessment for sites in the Chewacla (CH-U, CH-D) and Cane (Cane-1) creek watersheds, September 2006.

Category	CH-U	CH-D	Cane-1
Epifaunal Substrate/Available Cover			
Substrate and cover	20	18	20
Embeddedness	20	20	20
Velocity/Depth regime	15	15	18
Sediment deposition	20	20	17
Channel Morphology			
Channel flow status	18	20	20
Channel alteration	20	20	20
Frequency of riffles (or bends)	18	20	20
Riparian and Bank Structure			
Bank stability	20	20	20
Vegetative protection	20	20	20
Riparian zone width	20	18	20
Total score	191	191	195
Percent comparison with reference stream	97.9	97.9	
Habitat assessment	C[1]	C	

[1]N = non-supporting; C = comparable to reference site; S = supporting.



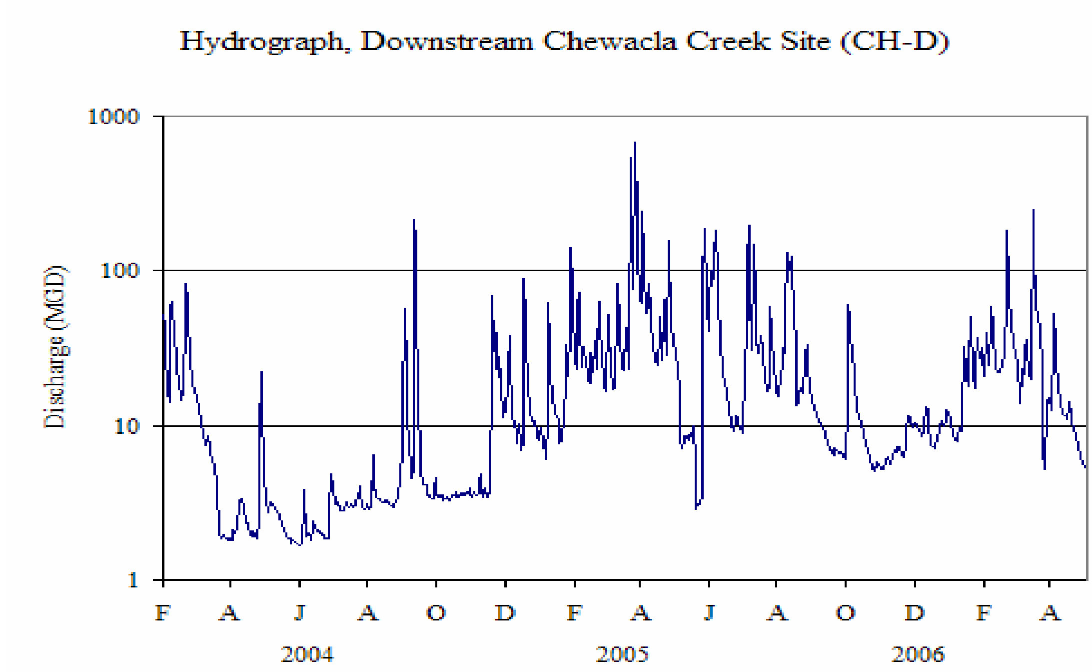


Figure 4. Hydrographs showing mean daily discharge for the upstream and downstream Chewacla Creek sites from August 2004 to April 2006.

Table 10. Physicochemical data measured at sites in Chewacla (CH-U, CH-D) and Cane (Cane-1) creeks during the fall of 2004 and 2005.

	2004			2005		
Variable	CH-U	CH-D	Cane-1	CH-U	CH-D	Cane-1
Dissolved Oxygen (mg/L)	8.2	10.0	10.8	7.50	8.20	9.80
Temperature (^{o}C)	15.5	14.0	12.5	18.5	18.0	15.8
pH	7.05	7.1	6.74	7.01	7.12	6.71
Conductivity (µS)	128.0	137.8	5.0	106.6	126.9	65.3
Turbidity (NTU)	4.40	1.39	62.10	3.60	1.03	3.00
NO_3-N (mg/L)[1]	0.063	0.09	0.058	0.051	0.079	0.012
NH_3-N (mg/L)[1]	0.027	0.006	0.095	0.04	0.02	0.03
O-PO_4 (mg/L)[2]	0.004	0.002	0.000	0.004	0.005	0.004
T-PO_4 (mg/L)[2]	0.02	0.006	0.006	0.02	0.01	0.02
Total alkalinity (mg/L)	52.50	62.50	26.25	46.25	56.25	27.50
Total hardness (mg/L)	45.65	61.91	19.49	51.05	55.14	29.61
TSS (mg/L)[2]	4.34	0.52	2.96	3.84	1.09	2.38

Table 11. Total abundance, percent relative abundance and total number of fishes by family collected at sites in the Chewacla (CH-U, CH-D) and Cane (Cane-1) creek watersheds, 2004 and 2005.

Family	CH-U	CH-D	Cane-1	%
Cyprinidae	166	69	262	30.2
Catostomidae	151	24	74	15.1
Ictaluridae	18	4	7	1.8
Cottidae	0	5	0	0.3
Fundulidae	6	5	0	0.67
Centrarchidae	491	174	89	45.8
Percidae	37	56	9	6.2
Total	869	337	441	100.0

Table 12. Comprehensive list of fish species including species abundance and total abundance for sites in Chewacla and Cane creeks during the two-year study period, 2004 and 2005. TC = tolerance class, FG = feeding guild, P = pioneer, INT = intolerant, TOL = tolerant, MI = intermediate. CH-U, CH-D = upstream and downstream Chewacla Creek sites, Cane- 1= reference site in Cane Creek. Species names follow Boschung and Mayden (2004).

Family	Common Name	TC	FG	CH-U	CH-D	Cane-1
Cyprinidae						
Campostoma oligolepis	Largescale stoneroller	[*]P	HER	[1]106	59	14
Cyprinella gibbsi	Tallapoosa shiner	INT	INS	0	0	33
Cyprinella venusta	Blacktail shiner	MI	INS	53	10	57
Hybopsis lineapunctata	Lined chub	INT	INS	0	0	10
Luxilus crysocephalus	Striped shiner	TOL	INS	0	0	27
Nocomis leptocephalus	Bluehead chub	P	O	0	0	14
Notropis baileyi	Rough shiner	MI	INS	0	0	103
Notropis buccatus	Silverjaw minnow	MI	INS	0	0	4
Notropis texanus	Weed shiner	MI	INS	7	0	0
Catostomidae						
Hypentelium etowanum	Alabama hogsucker	MI	O	20	23	33
Minytrema melanops	Spotted sucker	TOL	O	29	0	0
Moxostoma duquesnei	Black redhorse	MI	INV	94	0	41
Moxostoma poecilurum	Blacktail redhorse	MI	INV	8	1	0
Ictaluridae						
Ameiurus natalis	Yellow bullhead	TOL	O	13	0	2
Ictalurus punctatus	Channel catfish	MI	O	4	0	0
Noturus funebris	Black madtom	MI	INV	0	0	4
Noturus leptacanthus	Speckled madtom	MI	INS	1	4	1
Fundulidae						
Fundulus olivaceous	Blackspotted topminnow	MI	INS	6	5	0
Cottidae						
Cottus carolinae	Banded sculpin	MI	TC	0	5	0
Centrarchidae						
Ambloplites ariommus	Shadow bass	INT	TC	1	0	0
Lepomis auritus	Redbreast sunfish	MI	INV	43	3	15
Lepomis cyanellus	Green sunfish	TOL/P	TC	2	2	10
Lepomis gulosus	Warmouth	MI	TC	0	0	4
Lepomis macrochirus	Bluegill sunfish	TOL	INV	251	64	5
Lepomis megalotis	Longear sunfish	TOL	INV	124	87	36
Lepomis hybrid		TOL	INV	1	0	0
Lepomis microlophus	Redear sunfish	MI	INV	4	0	0
Lepomis miniatus	Redspotted sunfish	MI	INV	47	16	0
Micropterus coosae	Redeye bass	INT	TC	0	0	16
Micropterus salmoides	Largemouth bass	MI	TC	6	0	0
Micropterus punctulatus	Spotted bass	MI	TC	12	2	3

Table 12. – continued.

Table 12. – continued.

Percidae						
Etheostoma stigmaeum	Speckled darter	MI	INS	17	13	5
Etheostoma tallapoosae	Tallapoosa darter	INT	INS	0	0	1
Percina maculata	Blackside darter	INT	INS	0	0	3
Percina nigrofasciata	Blackbanded darter	TOL/P	INS	20	43	0
	Total Count			[2]869	337	441

[1]species abundance, [2]total abundance by site

Table 13. IBI metrics, raw metric scores, and total IBI scores for sites in Chewacla and Cane creeks, 2004 and 2005.
CH-U, CH-D = upstream and downstream sites in Chewacla Creek, Cane-1 = reference site in Cane Creek.

	CH-U		CH-D		Cane-1	
Metric	2004	2005	2004	2005	2004	2005
Species richness and composition						
Number of fish species	95.2	76.2	57.1	51.9	100	100
Number of cyprinid species	42.9	28.6	28.6	28.6	100	100
Number of round-bodied sucker species	100	100	33.3	66.7	100	100
Number of intolerant species	25.0	0	0	0	100	100
Trophic composition						
% tolerant	0	0	0	0	40.8	0
% top carnivores	60.3	60.2	46.0	19.4	100	100
% *Lepomis*	0	0	0	0	63.3	0
% benthic invertivores	27.0	13.6	86.0	100	87.2	92.1
% non-lithophilic spawners	39.5	16.3	9.2	29	83.0	60.1
% omnivores and herbivores	54.0	82.0	43.6	76.4	25.5	29.5
Fish abundance and condition						
Number of fishes per 100 m	0	50.0	50.0	50.0	100	50.0
% DELTs + hybrids	0	0	0	100	100	100
∑ metric scores	443.9	426.9	353.8	522.0	999.8	831.7
IBI Score[1]	**37.0**	**35.6**	**29.5**	**43.5**	**83.3**	**69.3**

[1]IBI Score = ∑metric scores / 12

Table 14. IBI and Iwb scores with associated integrity classes for all sites in Chewacla (CH-U, CH-D) and Cane (Cane-1) creeks, 2004 and 2005.

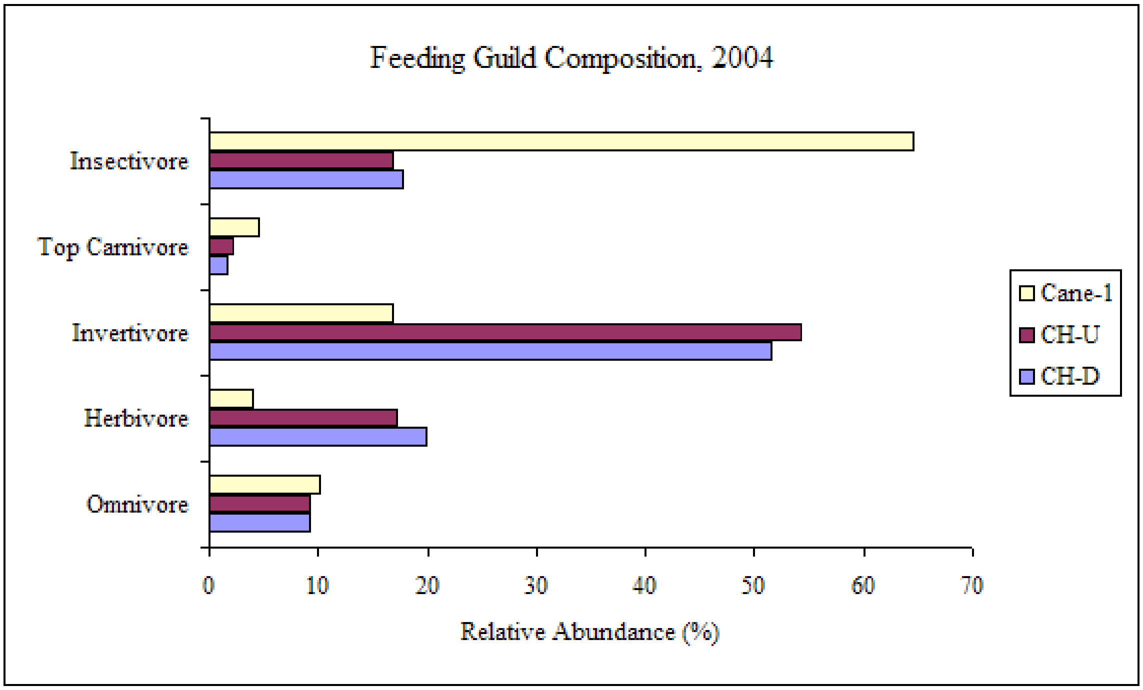
	2004		2005	
Site	Score	Integrity Class	Score	Integrity Class
IBI				
CH-U	37.0	Poor	35.6	Poor
CH-D	29.5	Poor	43.5	Poor
Cane-1	83.2	Excellent	66.9	Good
Iwb				
CH-U	9.4	Good	7.8	Fair
CH-D	6.0	Poor	6.4	Poor
Cane-1	13.5	Excellent	12.1	Excellent

Table 15. Core IBI metrics, and *F*-test results between raw metric values across sites.

Metric	*F*-value	*p*-value
Species richness and composition		
No. native species	5.25	0.102
No. native cyprinid species	39.38	0.006
No. native round-bodied sucker species	15.56	0.024
No. intolerant species	1.83	0.324
Trophic composition		
% tolerant	5.10	0.106
% top carnivores	1.61	0.363
% *Lepomis*	18.28	0.019
% benthic invertivores	32.56	0.009
% non-lithophilic spawners	7.34	0.067
% generalist feeders	16.51	0.022
Fish abundance and condition		
No. of individuals in sample per 100 m	2.73	0.211
Percent DELTs + hybrid species	2.09	0.270

Table 16. Descriptive measures of fish assemblage structure at sites in the Chewacla (CH-U, CH-D) and Cane (Cane-1) creek watersheds, 2004 and 2005.

	2004			2005		
	CH-U	CH-D	Cane-1	CH-U	CH-D	Cane-1
Abundance (N)	521	186	327	332	148	114
Richness (S)	21	12	19	17	13	19
Diversity (H′)	2.38	1.97	2.35	1.98	2.05	2.54
Evenness (E)	0.514	0.600	0.553	0.429	0.600	0.674



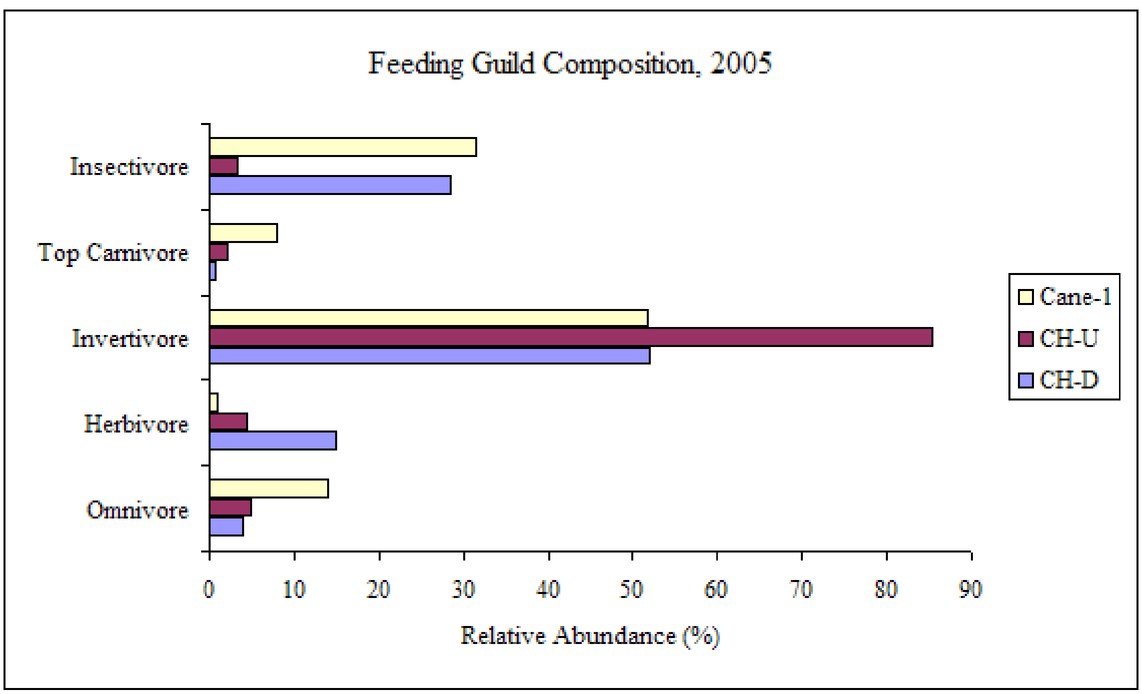


Figure 6. Feeding guild composition – percent total number of fishes collected by feeding guild at sites in Chewacla and Cane creeks, 2004 and 2005.

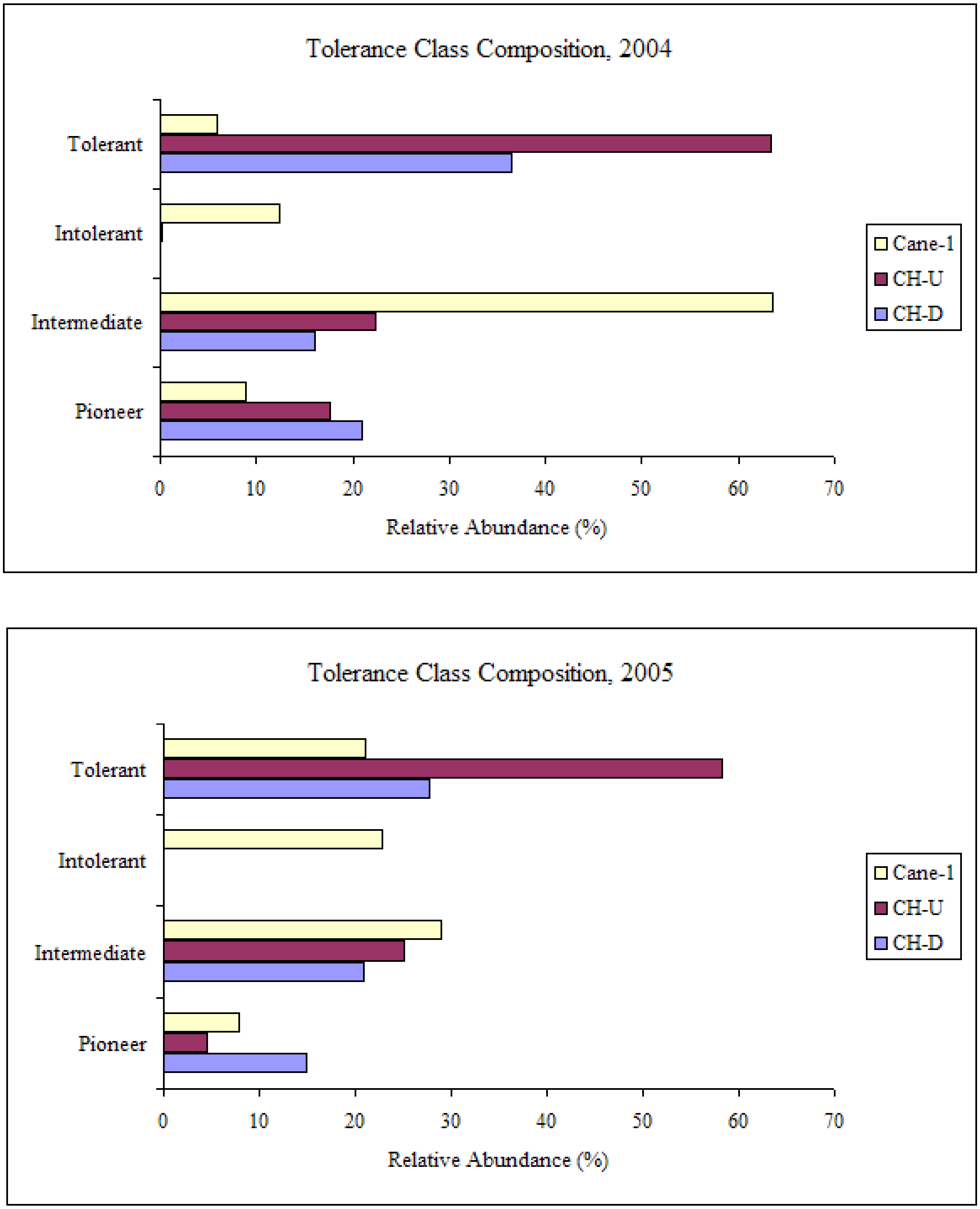


Figure 7. Tolerance class composition – percent total number of fishes collected by tolerance class in Chewacla and Cane creeks, 2004 and 2005.

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