AN ASSESSMENT OF BIOTIC INTEGRITY OF THE CHATTAHOOCHEE RIVER, GA

DOWNSTREAM OF ATLANTA, GA

BY

CLIFTON JACKSON JR.

(Under the direction of CECIL A. JENNINGS)

ABSTRACT

During the past decade, indices that assess biological integrity have become an important tool for natural resource managers. A 1990-91 assessment of a 64-km reach of the Chattahoochee River downstream of Atlanta, GA found this reach to be in poor condition. Since then, phosphorous loads in the river have declined and piscivorus striped bass *Morone saxatilis* have been introduced. The biological integrity of the same reach of the Chattahoochee River was reassessed with the methods used in the previous study. Biotic integrity scores increased slightly from those of the 1992 study at three of the four stations. However, there was a decreasing trend in the metric that measures fish health. Improvements in the biotic integrity of this reach was only marginal and remains poor. Further, dramatic reductions in pollution and habitat degradation would be necessary before the biotic integrity of this reach of river improves to fair or good.

Index Words: Index of biotic integrity, Chattahoochee River, Ecological health, Wastewater, Fishes

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CHAPTER 1

INTRODUCTION

The Chattahoochee River south of Atlanta, GA has been the subject of much controversy in recent years. The City of Atlanta is in constant violation of the Federal Clean Water Act, primarily because of insufficient sewer facilities that flood during routine rain events (Jimmy Evans, Georgia Department of Natural Resources - personal communication 1997). Consequently, the river has been plagued by severe eutrophication caused by phosphorus loading (Bayne et. al 1990) and chemical pollution caused by point and non-point sources. Chemicals that are associated with raw sewage can affect fishes by causing poor health (Marcogliese and Cone 2001) and restructuring the food base (Longheed and Chow-Fraser 1998). Poor water quality compromises biotic integrity; however, the biological integrity of the Chattahoochee River often goes unassessed. Biotic integrity is defined as the ability of a body of water to support a biological community similar to what would be expected under pristine condition (Angermeier and Karr 1994).

Karr (1981) found that assessing the health and status of fish communities correlated with the overall health of the aquatic ecosystem in question and formulated an Index of Biotic Integrity (IBI) to yield a point estimate of the health of an aquatic ecosystem. The IBI incorporates information about diversity [species abundance, number of sunfish species (*Lepomis* sp.), number of round-bodied suckers (family Catostomidae), number of intolerant species, number of tolerant species, trophic guilds (percent omnivores, percent insectivores, percent top carnivores), assemblage abundance and a fish health analysis (i.e., percent diseased)] (Karr 1981). The IBI has been widely tested, accepted, and adapted for regional applications (Miller et al. 1988).

There are extensive records of water quality variables of the Chattahoochee River; however, records of the ecological health and fish population data are very limited. Routine assessment of the biotic integrity could reveal trends of the health of the river. A periodic assessment of biotic integrity could also be an important tool for monitoring the synergistic effects of pollutants and physical habitat degradation as well as provide a database for monitoring the effects of changing water quality on the fish community of the Chattahoochee River (Karr 1981).

Mauldin and McCollum (1992) tabulated an IBI on a portion of the Chattahoochee River south of Atlanta in September of 1990 and June of 1991. They used an IBI modified by the Ohio EPA (1988) and found the four sites they sampled to be in poor condition (37-53% of max. score). Recent episodes such as recurring raw sewage spills from rain events, the introduction of a keystone predator (i.e., fingerling striped bass, *Morone saxatilis*) in 1990 and 1992, and Atlanta's recent progress with phosphorus control activities (Nungesser and Frantz 1997) have changed the river.

I wanted to evaluate the river's biological response to the recent changes by applying an IBI to the same reach of river evaluated by Mauldin and McCollum (1992) and compare results. I also wanted to evaluate the changes to IBI scores after incorporating data derived from sampling with gear other than an electrofishing unit.

CHAPTER 2

REVIEW OF LITERATURE

Historically, humans have had a degrading effect on aquatic environments. Watershed modification, chemical contamination, introduction of exotics, and over fishing have had devastating affects in some waterways in North America (Karr et al. 1985). Assessing the damage and establishing recovery criteria for aquatic ecosystems is often a difficult complex task requiring biological and physiochemical knowledge (Hughs et al. 1990). The passage of the Clean Water Act of 1977 set mandates for biological integrity of U. S. water resources and led to many physiochemical water quality standards (Angermeier 1987). However, physiochemical monitoring alone did not reveal all antrophogenic perturbations (Karr 1981). The initial water quality standards were not designed to protect the aquatic ecosystem diversity and integrity, but rather to sustain clean water for consumption, swimming, and fishing (Hughes and Noss 1992). The concept of biological monitoring of aquatic ecosystems grew out of concern for the deficiencies of physiochemical monitoring programs. Monitoring programs of this sort were based on specific levels of contaminants and acute toxicity test (Karr 1981). These types of monitoring criteria are not capable of assessing the temporary, sublethal, or synergistic effects of many contaminants (Karr 1981). Neither does physical and chemical monitoring alone measure the effects of human-induced perturbations such as flow alteration, habitat alteration, siltation and other physical habitat degradation (Karr 1981). The use of biological criteria in assessing ecological health addressed the deficiencies of physical and chemical monitoring. Karr (1987) stated that “Mankind’s failure to use ecological principals to minimize the negative impacts of human activities is arguably the most important failure of the twentieth century”. As the concept of biological monitoring became accepted, the debate began over the appropriate criteria and method for performing the assessment (Karr 1987).

Several groups of aquatic organisms including diatoms, macro-invertebrates, and fishes, were evaluated for their ability to indicate ecological health; however, fishes were the prevalent group used by many biologist (Karr 1981). The advantages of using fish as ecological indicators included a less tedious taxonomic regime, documented life-history data, public appeal, and stress response evaluation (e.g., growth, reproduction, toxicity and bioassay of contaminants) (Karr 1981). The most notable ecological characteristic of fish communities is that they are structurally and functionally diverse and can yield a broad range of information about many types of perturbations (Hocutt 1981).

The concept of evaluating fish community characteristics for determining ecosystem health was proposed around 1900 (Simon 1998). As research on fishes progressed, several other attributes made them highly favorable as ecological indicators. For example, fish have known seasonal distribution patterns, which can be incorporated into the sample regime (Simon 1998). Fish communities are quick to recover from natural disasters (Simon 1998). They also have relatively long life spans (3-15+ years) and are usually present in even the smallest of bodies of water (Simon 1998). Furthermore, their presence or absence can be indicative of the status of various trophic levels and can be directly tied with the regulatory language of the Clean Water Act (i.e., fishable waters mandate) (Karr 1981).

A number of concepts were proposed for using fish communities to evaluate ecological health. The four main approaches were: (1) indicator taxa; (2) multivariate analysis; (3) species diversity indices; and (4) the index of biotic integrity (Fausch et al. 1990). Each of these approaches had several advantages and disadvantages. However, the one essential element that they all share is a sampling regime sufficient to adequately expresses the abundance and/or distribution of all ichthyofauna (Fausch et al. 1990).

Indicator taxa were evaluated for their tendency to decline in abundance after environmental degradation rather than on a series of experiments. This approach has distinct disadvantages including: few guidelines for the selection of such species; declines in populations unrelated to degradation; and the variance of responses to degradation (i.e., regionally, temporally, and with the age of fish) (Fausch et al. 1990).

Indices that attempt to compare similarities among species composition and relative abundance in the degraded location of interest and one or more reference sites also have been used to assess ecological health (Fausch et al. 1990). This procedure and others such as ordination have been used and are found to differentiate among degraded and relatively undegraded sites; however, analyses are complicated and few standard procedures are present for such methods (Fausch et al. 1990).

Species diversity indices are among the more popular techniques of evaluating ecological health. The complexity of their derivation varies; however, all assume that species diversity declines with the onset of environmental degradation (Fausch et al. 1990). Hughes and Noss (1992) defined biological diversity as “the variety of life and its processes”. Whether the threat of extinction poses a serious concern to biological diversity is uncertain; consequently, ecosystem health assessments should be performed on a much broader scale by integrating many other species (Hughes and Noss 1992). Although diversity is easy to calculate, it possesses some major disadvantages when compared to other assessment techniques including: minimal information is gained from its calculation; diversity varies with stream size and region; and issues concerning exotic species are not incorporated in these procedures (Fausch et al. 1990). Species diversity also can increase temporarily with the onset of environmental degradation (Scott and Helfman 2001). Traditional diversity measures do not yield information about changes in fish assemblage structure as a direct result of man-induced perturbations (Hughes and Noss 1992) such as increased urbanization (Scott et al. 1986). Furthermore, environmental policies based on diversity standards tend to promote reactive rather than preventive resource management (Angermeier and Karr 1994).

The Index of Biotic Integrity (IBI) was proposed by Karr (1981) in an effort to produce a more holistic assessment of fish communities that would incorporate many attributes of environmentally disturbed and relatively pristine conditions into one index. Karr and Dudley (1981) defined the concept of biological integrity as "the capability of supporting and maintaining a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of the natural habitats of the region". The original IBI was composed of 12 metrics that related to species composition and richness, and a number of ecological factors (Karr 1981).

The IBI incorporates information about diversity and species abundance including: number of sunfish species (*Lepomis* sp.), number of darters (family Percidae), number of suckers (family Catostomidae), number of intolerant species, percent of tolerant species, trophic guilds (percent omnivores, percent insectivores, percent top carnivores), assemblage abundance, and condition (i.e., percent diseased) (Karr 1981). Scores from each of the metrics yield an important characteristic about the health of the ecosystem. The comprehensive score of all the metrics combined determines the status of the biotic integrity of a stream according to one of six classes. The six classes include 'excellent', comparable to pristine condition, good, fair, poor, very poor, and 'no fish', which means repetitive sampling did not yield any fishes (Karr 1981; Karr et al. 1986). The scoring procedure also allows some room for professional judgement (Karr et al. 1986). IBI's have regional application and require minimal data (i.e., relative-abundance and species composition) (Faush et al. 1984; Miller et al. 1988; Hughes et al. 1990). Factors, such as the synergistic effects of many different contaminants and sublethal effects (e.g., reproduction, growth, health), often are not considered when assessing physiochemical habitat degradation, but may be assessed by indices of biotic integrity (Karr 1981). When performing health evaluations of small Illinois streams with known degradation patterns, the IBI performed a more accurate assessment of the biotic integrity than the commonly used Shannon-Weiner diversity index (Angermeier and Schlosser 1987).

Metrics used to calculate the IBI are sensitive to various perturbations (Karr 1981). Metrics that incorporate species richness and composition with respect to stream size partition the effects of habitat degradation by highlighting the resulting trends such as a decline of intolerant species followed by increases in tolerant species (Karr 1981). Reductions in habitat diversity also can cause hybridization and shifts in the fish community composition from specialist to generalist (Karr 1981). Metrics based on ecological factors yield information related to the proper function and productivity of a water body (i.e., healthy systems have distinct proportions of fishes in trophic guilds such as top carnivores, omnivores, and insectivores) (Karr 1981). Unlike some of the other aforementioned assessment techniques, the IBI gives information about the general health of the fishes, which declines with degradation, by incorporating deformities, erosions, lesions, and tumors into its tabulation (Karr 1981).

Despite the rather large diversity of fish communities, they all respond to perturbations in a similar pattern. Deforestation, poor agriculture practices, siltatation from urban development, and various other land-use practices all typically elicit a decline in abundance of intolerant species, poor reproductive success of certain substrate spawners, and a restructuring of the fish community (usually an increase in omnivores) (Toth et al.1982; Foltz 1982; Scott et al. 1986; Crumby et al. 1990; Brenner et al. 1991; Roth et al. 1996; Scott and Hall 1997; Wang et al. 1997). Other types of physical habitat degradation such as drought and reductions in habitat diversity (e.g., removal of woody debris) typically reduce species richness (Foltz 1982; Toth et al. 1982; Angermeier and Karr 1984). Likewise, chemical habitat degradation (e. g., sewer outfalls and farm effluents) manifests itself in fish communities similarly by declines in species richness and abundance (Katz and Gaufin 1952; Tsai 1973; Oberdorff and Porcher 1994). The responses to perturbations are similar across a variety of habitats and drainage sizes. However, there is not a single IBI that can accurately and consistently evaluate biotic integrity across the rather wide variety of habitats that exist throughout the world; therefore, there is a need for modifications of IBI metrics to address the discrepancy among various habitats.

Although the IBI has been modified numerous times for regions throughout the United States and in several other foreign countries, the conceptual framework remains intact (Simon 1998). Modifications generally are aimed at deleting or exchanging ineffective metrics for those that are better at indicating degradation at the site in question. Most modifications pertain to metrics associated with native species richness. Once an IBI is modified to suit a particular habitat type, it can be used to evaluate similar habitats within an ecoregion (Fausch et al. 1984). Modifications of the IBI in the United States exist for streams in the Western Great Plains (Bramblett and Fausch 1991), Maryland coastal plains (Scott and Hall, Jr. 1997), South Carolina coastal plains (Paller 1994; Paller et al. 1996), and the littoral zone of the Great Lakes (Minns et al. 1994). Schleiger (2000) recently modified the IBI to have applicability within the region of the Chattahoochee River being assessed in this study. However, this publication was unavailable during my study, and this version of the IBI was not calibrated for boat-mounted electrofishing and would be inappropriate for the mainstem Chattahoochee River. Therefore, a version of the IBI modified by the Ohio Environmental Protection Agency (1988) was used to perform the assessment. This version of the IBI was also used in Mauldin and McCollum's (1992) study, which makes data from both studies more comparable.

The Ohio Environmental Protection Agency (OH EPA) compiled many samples from different sized rivers and streams in various conditions of ecological health to modify and adopt an IBI to assess water quality (OH EPA 1988). The IBI used by the OH EPA (1988) was formulated to factor in size of the drainage area and was derived from numerous samples across diverse habitats in Ohio. Each of the metrics selected for this IBI were analyzed, and values were adjusted for flowing waters that require a boat-mounted electrofishing (Table 1)(Ohio EPA 1988). This version of the IBI has been proven to distinguish among pristine and degraded sites across various rivers in Ohio (OH EPA 1988). The twelve metrics incorporated into this IBI range in their ability to assess degradation and are sensitive to specific types of disturbances.

The first metric, the total number of indigenous species, was derived from the observation that these species typically decline with increases in environmental degradation (Karr 1981; Karr et al. 1986). Exotic species are excluded from this metric because their presence is sometimes indicative of poor ecological health (Karr et al. 1986). The range and type of sensitivity of this metric is relatively broad (Ohio EPA 1988).

The second metric, the proportion of round-bodied suckers, was substituted for Karr's (1981) number of darter species because of the sampling bias associated with boat-mounted electrofishing units (Ohio EPA 1988). Both groups decline in abundance in response to environmental perturbations (Ohio EPA 1988). This metric is a good indicator of physical habitat alteration since round-bodied suckers do not tolerate high turbidity and siltation, removal of riffle habitat that is vital to their spawning success, or poor water quality (Ohio EPA 1988). The white sucker (*Catostomus commersoni*) is excluded from this category because of its high tolerance for degraded areas (Ohio EPA 1988).

Table 1. The 12 metrics (Ohio EPA 1988) used to calculate an Index of Biotic Integrity (IBI) for a 64-km reach of the Chattahoochee River downstream of the Atlanta Water-Intake Structure (river km 483-419). The scores can range from 12 to 60; 12 indicates severe degradation, 60 indicates pristine condition.

Metric	Normal (5)	Deviates (3)	Strongly Deviates (1)
Number of Species	>20	10-20	0-10
Number of Sucker Species	>5	3-5	<3
Number of Sunfish Species	>3	2-3	<2
Number of Intolerant Species	>3	2-3	<2
% Tolerant Species	<15%	15-27%	>27%
% Round-body Suckers	>38%	19-38%	<19%
% Omnivores	<16%	16-28%	>28%
% Insectivores	>54%	54-27%	<27%
% Top Carnivores	>10%	5-10%	<5%
Number / 0.5 Km	>400	400-200	<200
% Lithophilic	>50%	50-25%	<25%
Deformities, erosion, lesions, or tumors	≤0.5%	3-0.5%	>3%

The third metric, the number of sunfish species, follows Karr's (1981; Karr et al. 1986) IBI and is used to assesses the condition of pool habitats. Unlike round-bodied suckers, the family Centrarchidae (i.e., excluding *Micropterus* sp.) prefer quiet pool habitat and forage for food from the top to the bottom of the water column (Ohio EPA 1988). Therefore, they have moderate to high sensitivity to perturbations such as habitat alteration and are good indicators of ecological health (Karr et al. 1986).

The fourth metric, the number of sucker species, is included because the family Catostomidae is intolerant of chemical and physical habitat degradation and because of the longevity of the suckers (Karr et al. 1986). This metric is highly sensitive and an excellent indicator of physical and/or chemical environmental degradation (Ohio EPA 1988).

The fifth metric involves the number of intolerant species. This metric is selected because when used, it can evaluate and distinguish integrity among streams that are close to pristine condition (Ohio EPA 1988). This metric was derived from Karr's (1981; Karr et al. 1986) guidelines. Intolerant species often become rare or extirpated once environmental conditions begin to degrade (Karr et al. 1986). The list is derived from historical data (Karr et al. 1986). It should only include 5-10% of the species most sensitive to degradation, and species that become extirpated or extraordinarily rare when IBI scores recede to the 'fair' category (Karr et al. 1986). This is the most sensitive metric (Ohio EPA 1988).

The sixth metric, the proportion of tolerant species, was selected to replace the proportion of individuals as green sunfish (*Lepomis cyanellus*) (Karr et al. 1986). The distribution and numbers of green sunfish varied with stream size; therefore, other fishes that displayed a tendency to thrive in perturbed environments were selected (Ohio EPA 1988). The sensitivity and utility of this metric is observed as environmental conditions degrade from 'fair' to 'poor' (Ohio EPA 1988).

The seventh metric, the proportion of omnivorous species, was selected because of the ability of omnivores to thrive while specialist feeders decrease with instability of the food chain (Karr 1981; Karr et al. 1986). The guidelines require consumption of at least 25% of both plant and animal matter (Karr et al. 1986) and exclude highly variable (e. g., channel catfish) and extremely specialized fishes (e. g., paddlefish) (Ohio EPA 1988).

The eighth metric, the proportion of insectivorous fishes, reflects the status of the insect food base of a river (Karr et al. 1987). The proportion of these fishes typically decrease as the level of disturbance increases because of a loss of diversity in the insect community (Ohio EPA 1988). This metric is moderately sensitive to environmental degradation (Ohio EPA 1988).

The ninth metric, proportion of fishes that are the top carnivores in the system, was extracted from Karr's (1981) original IBI to measure the overall status of the food base. The sensitivity of this metric to perturbations is moderate to high (Karr et al. 1986). Once more, fishes that display feeding plasticity are not included in this metric (Ohio EPA 1988).

The tenth metric, the number of individuals in a sample, was used to incorporate abundance into the IBI (Karr et al. 1986). Generally, environmental disturbance, such as toxic pollutants have been shown to reduce productivity (Ohio EPA 1988). This metric has a low to moderate sensitivity (Karr et al. 1986).

The eleventh metric, proportion of fishes that are simple lithophilic spawners, is a metric aimed at assessing physical habitat degradation (Ohio EPA 1988). Usually, this degradation is caused by excess siltation or simple reductions in this type of habitat caused by perturbations such as flow alteration and gravel mining; and thus, will reduce the number of these fishes (Ohio EPA 1988). This metric replaced Karr's (1981) percentage of hybrid fishes because of the difficulties associated with the original metric that included a lack of historical data and accuracy problems, (i.e., determining normal and standard rates of hybridization) (Ohio EPA 1988) and difficulties with identification (Karr et al. 1986).

The twelfth and last metric assesses the proportion of individuals with deformities, eroded fins, lesions, and tumors (DELT). This metric is aimed at determining the level of sub-acute disturbances caused possibly by chemical pollution, poor nutrition, overcrowding, siltation, and other factors (Karr et al. 1986). This metric often entails the results of sewer and urban run-off (Ohio EPA 1988).

Each metric is scored as either 1, 3, or 5 based on expected and potential scores and the size of the drainage area (Ohio EPA 1988). A total score between 12 and 60 is derived from summing all the metrics, and the individual sites are ranked as very poor (12-22), poor (28-34), fair (40-44), good (48-52), and excellent (58-60) (Karr et al. 1986). The scoring procedure allows some room for professional judgement (Karr et al. 1986). Scores from each of the 12 metrics and summation of the 12 can yield important trends and information when used to monitor ecological health.

CHAPTER 3

METHODS

Study Site

The Chattahoochee River begins in Blue Ridge Province, Georgia and flows southwest through Atlanta, GA; ultimately, it converges with the Flint River in southwest Georgia (Frick et al. 1998). The converged Chattahoochee and Flint rivers are impounded on the border of Georgia and Florida to form Lake Seminole. There are five other impoundments along the Chattahoochee River, and two of these are upstream of the study. The Chattahoochee River serves as the primary source of drinking water and is a destination for treated waste water disposal for Atlanta, GA and other cities that border the river along its 692 km path (Mauldin and McCollum 1992).

A 64-km reach of stream (Figure 1) assessed during 1990-1991 by Mauldin and McCollum (1992) was sampled for the present study. Four sample stations were established along the river: Station 1 was at river kilometer (rkm) 483 (above the Atlanta Waterworks intake facility), Station 2 was at rkm 481 (immediately downstream of Atlanta's and Cobb County's treated wastewater outfalls), Station 3 was at rkm 459 (Hwy 166 bridge), and Station 4 was at rkm 419 (Hwy 16 bridge) (Figure 1). Station 1 is the northernmost station and has the least amount of environmental perturbations. Its average temperature is the coolest of the four because it receives cool-water releases from Morgan Falls Dam, which is about 14 rkm upstream. Sewer outfalls and active dredging operations appear within or immediately upstream of Stations 2 and 3. Station 4 is the southernmost station and is devoid of or a

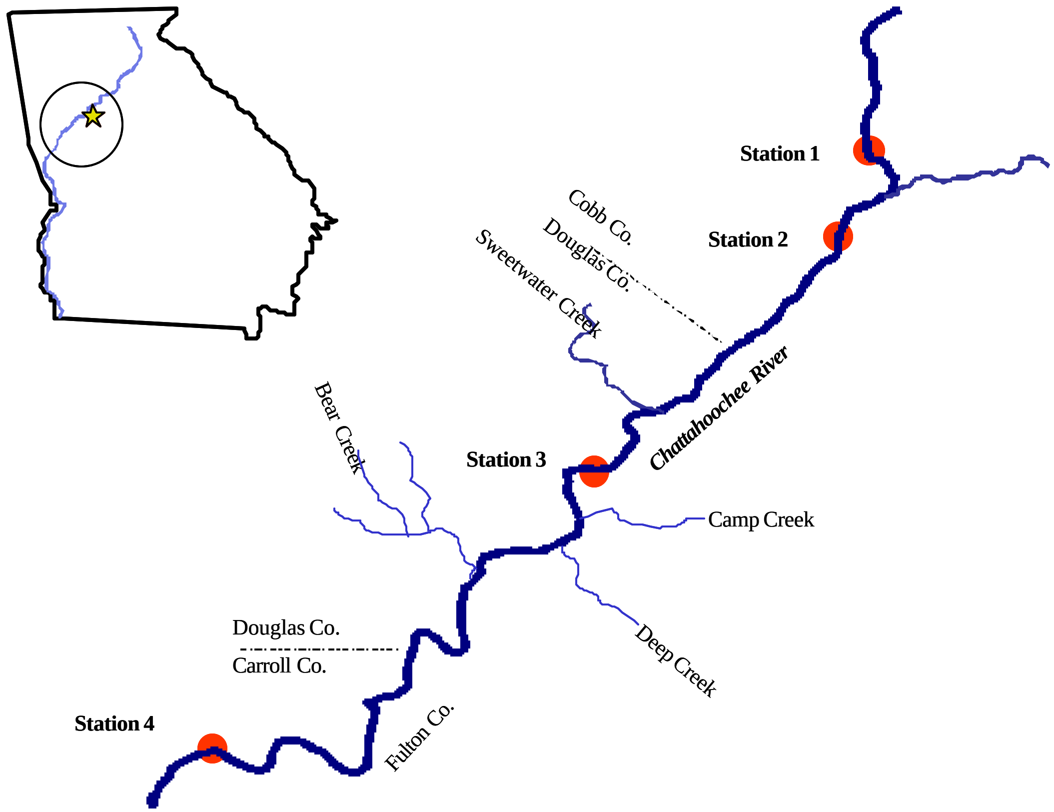

Figure 1. A 64-km reach of the Chattahoochee River near Atlanta, GA assessed with an Index of Biotic Integrity in 1998 Station 1 was at river km 483 (above the Atlanta Waterworks intake facility), Station 2 was at river km 481, Station 3 was at river km 459 (Hwy 166 bridge), and Station 4 was at river km 419 (Hwy 16 bridge).

substantial distance from environmental perturbations such as dredging or treated wastewater discharge. Environmental conditions of this 64-km reach ranged from poor to very poor in the previous assessment (Mauldin and McCollum 1992).

Sampling Fish Populations

Fish populations in the study reach were sampled with boat-mounted electrofishing gear. A Type VI Smith Root electrofishing unit (120 pulses per second; pulsed direct current at 4-6 AMPS) connected to a 5000 watt Honda generator was used to sample the fishes. Six samples per station (three on each bank) were collected at each of the four stations; once in June-July, 1998 and once in September, 1998. Each sample was obtained by electrofishing in a downstream direction for six 15-minute intervals (Figure 2). During each interval, all of the fishes seen were netted and placed into a live well until each fish's total length (mm), weight (g), and taxonomic identification were determined and recorded. The general health (i.e., occurrences of deformities, erosions, lesions, and tumors) of members of the family Centrarchidae was assessed at the study site. Additionally, a number of *Lepomis* species were preserved in 10% buffered formalin and retained to conduct a more thorough external health examination in the lab. Fishes that could not be identified in the field were preserved in 10% buffered formalin and returned to the lab for identification. All fishes that were to be released were transported downstream about 100 meters outside of the study area and returned to the water. After fish from the first interval were processed, the 2nd 15-minute interval was begun, and the remaining samples were conducted in the same manner. The 4th 15-minute interval occurred on the opposite shoreline at about the same starting location as the first (Figure 2). The distance of shoreline sampled during each 15-minute interval was measured with a laser range finder.

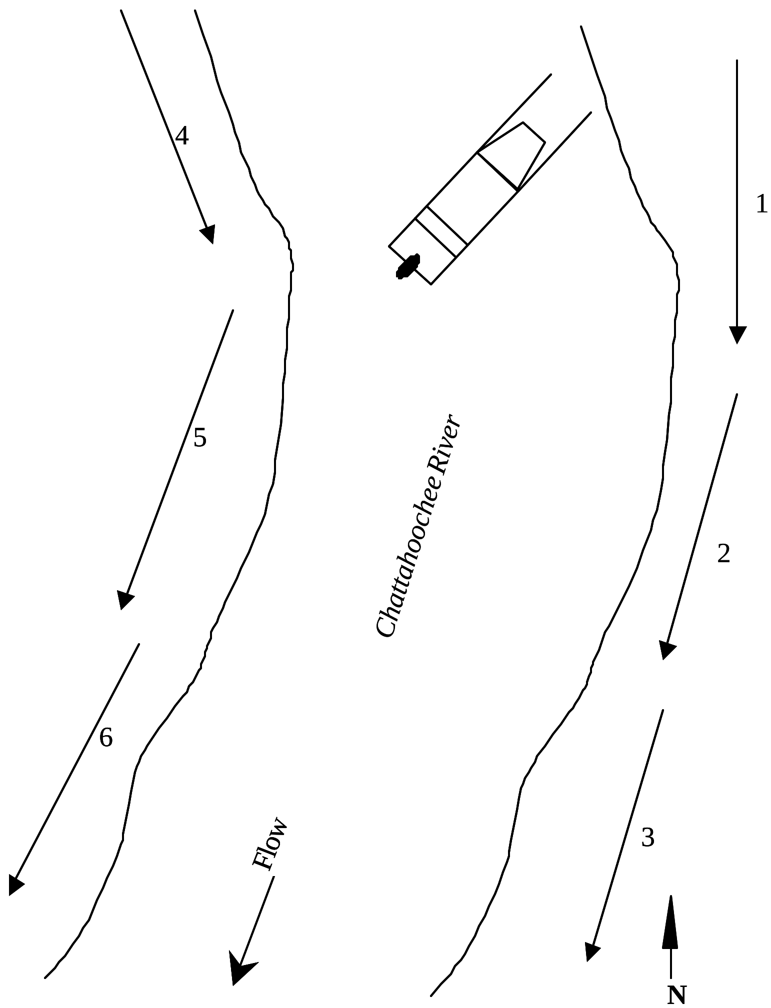

Figure 2. Shoreline electrofishing sample regime for the 1998 Index of Biotic Integrity assessment of four stations within a 64-km reach of the Chattahoochee River near Atlanta, GA

After electrofishing surveys were completed in September, six hoop-nets and 12 wire-mesh minnow traps per station were deployed and fished for two consecutive nights. Three of the hoop-nets were 0.76 meters in diameter (1.27 cm bar-mesh), and three were 1.22 meters in diameter (1.27 bar-mesh). The nets were fished primarily in the main river channel. Minnow traps were weighted with rocks and fished in various intermediate to shallow-water (0.4 - 2.0 meters) habitats. The fishes caught with these gear were identified and released, or preserved in 10% formalin and taken back to the lab for identification.

IBI Application

A version of Karr's (1981) index of biotic integrity modified by the Ohio Environmental Protection Agency (1988) was calculated for each of the four stations. The metrics of this IBI were designed for use in rivers and streams large enough to be sampled by boat-mounted electrofishing gear (OH EPA 1988) and is the same version used by Mauldin and McCollum (1992). In addition to the electrofishing gear, hoop nets and minnow traps also were fished to determine if these gear augmented the total number of species caught and if the IBI scores calculated from the increased catch was different from the electrofish-only IBI scores.

IBI scores were calculated for electrofishing-only data and for electrofishing plus hoop-net and minnow trap data. Scores yielded from the twelve metrics were used to tabulate a cumulative score that was indicative of the biotic integrity at each station. The 1998 IBI electrofishing results were then compared to those in the previous study (Mauldin and McCollum 1992). Afterward, the 1998 electrofishing-only data and electrofishing plus passive gear data was compared to determine if the additional gear improved the IBI scores and should be incorporated into the standard protocol for conducting IBIs on large rivers in this region.

Water Quality

Water quality variables including dissolved oxygen, temperature, turbidity, total phosphorous, and 5-day biological oxygen demand (BOD) were obtained from the Georgia Department of Natural Resources - Environmental Protection Division (GA DNR-EPD) or the United States Geological Survey (USGS).

CHAPTER 4

RESULTS

Abnormally low water conditions (Barber and Stamey 2000) and man-made alterations to the study area rendered some stations unreachable during the September sampling periods. Station 2 was located between two rock weirs that were impassable during low water conditions. Station 4 was relatively shallow and included a substantial amount of exposed bedrock, which severely diminished accessability during low water conditions. Consequently, Stations 2 and 4 were not sampled with either gear during the September sampling period. Stations 1 and 3 were sampled with all gear types during both sampling periods.

Electrofishing

The average length of shoreline sampled during each electrofishing interval was 244.4 meters (SD = 50). This effort resulted in 843 fishes from 35 species (Appendix 1). Eleven species sampled during the present study were not sampled in the 1990-91 study. However, chain pickerel *Esox niger*, brown bullhead *Ameiurus nebulosus*, and yellow bullhead *Ameiurus natalis* sampled during the previous study were not sampled in the present study (Table 2). A single adult striped bass was among the catch of the present study, but none were caught during the previous study.

Table 2. Fishes that were sampled in the Chattahoochee River near Atlanta, GA in during IBI assessments in 1992 and 1998. This is a list of fishes that were sampled exclusively in the 1992 study or the 1998 study but not in both studies.

Study	Species of fishes
Mauldin and McCollum 1992	chain pickerel, brown bullhead, yellow bullhead
Present study	rainbow trout, Alabama hogsucker, bluestripe shiner, golden shiner, flat bullhead, mosquito fish, brook silverside, mottled sculpin, striped bass, redeye bass, shoal bass

Passive Gear Sampling

Stations 1 and 3 were sampled with six hoop-nets per station and 12 minnow traps per station in mid-September, which resulted in about 605 consecutive hours of soak time for hoop-nets, and about 840 hours for minnow traps. The hoop-nets and minnow traps were checked about 42 hours after deployment. The passive gear sampling resulted in 47 fishes from seven species. The passive gear was effective at sampling some species that were not sampled with electrofishing gear. Minnow-traps sampled a few young-of-year bluegill sunfish *Lepomis macrochirus*, redbreast sunfish *Lepomis auritus*, and one yellow bullhead *Ameiurus natalis* that were not sampled with electrofishing gear during this study. The hoop-nets sampled five species of fishes including flathead catfish *Polydictis olivaris* that were not sampled with electrofishing gear in the present study or by Mauldin and McCollum (1992). The hoop-nets also caught channel catfish *Ictalurus punctatus*, bluegill sunfish, redbreast sunfish, and one redeye bass *Micropterus coosea*. Twenty-one of the 42 fishes caught in hoop-nets were catfishes. The minnow traps were empty most (i.e., 83.3%) times when they were checked. Half of the hoop-nets were empty when checked. Although the catch of these two passive gears increased both the number of fishes and the number of species sampled, the increased catch did not affect IBI scores for the respective stations. In fact, all of the IBI scores remained the same.

Fish catch composition and abundance sampled in the study reach varied among the four stations, but some patterns were evident (Appendix 1) (Table 3). Electrofishing samples at Stations 1 and 2 contained more individuals than those from Stations 3 and 4 (Appendix 1) (see also Table 3). The fish population at Station 2 was less diverse than that of the other three stations; it contained only 15 species, whereas the other stations each contained 19 or 20 species. The number of fishes (287) sampled (electrofished) at Stations 1 and 3 in June-July was slightly more than the number (252) sampled in September. Two or three non-native species were sampled at each of the four stations. The additional catch

Table 3. The numbers of fish species and the numbers of fishes sampled by electrofishing four stations on the Chattahoochee River near Atlanta, GA during June-July and September of 1998.

	Number of species			Number of fishes		
Station	total	June-July	September	total	June-July	September
1	19	19	18	311	166	155
2	15	15	*	161	161	*
3	20	13	16	218	121	97
4	20	20	*	143	143	*

* Samples not taken because of low water conditions.

generated by sampling with the passive gears at Station 1 was minimal; it included only one channel catfish, which was sampled with a hoop net, and two bluegill that were sampled with the minnow traps. The supplemental catch at Station 3 included a yellow bullhead, a bluegill sunfish, and a redbreast sunfish that were sampled with minnow traps; 15 channel catfish, five flathead catfish, 15 redbreast sunfish, three bluegill sunfish, and one common carp were sampled with hoop-nets.

Deformities, Erosions, Lesions, and Tumors

Abnormalities were present on some of the fishes examined in this study. Fishes from Station 3 had the highest (10.1%) occurrence of deformities, erosion, lesions, and tumors, Station 4 had the second highest rate at 9.1%, followed by Station 1 at 4.4%, and Station 2 at 1.9%.

Comparison of IBI Scores

Index of biotic integrity scores in this study ranged from 28 to 31 compared to 22 - 32 (Table 4) in the Mauldin and McCollum (1992) study. Five of twelve metrics increased at Station 1, and four of the twelve metrics increased at Station 4 from those of the previous study; one metric (i.e., DELT) declined at Station 1 and all others remained the same at these stations (Mauldin and McCollum 1992) Appendix 2). Consequently, the overall scores at Stations 1 and 4 increased by eight points each. The scores for five metrics at Station 3 decreased, whereas only two metrics increased; the overall score at this station was reduced by six points (Mauldin and McCollum 1992). At Station 2, two metrics had increased values and two values had reduced values. As a result, Station 2 changed the least (2 points) in overall scores from the previous study (Table 4) (Mauldin and McCollum 1992).

Table 4. IBI scores for four stations located on a 64 km reach of the Chattahoochee River near Atlanta, GA. Assessments were performed in 1992 and 1998. The score can range from 12 to 60; larger numbers are indicative of sites with conditions similar to pristine.

	Station							
	1		2		3		4	
Study	Score	Rank	Score	Rank	Score	Rank	Score	Rank
Mauldin and McCollum 1992	22	Very Poor	24	Very Poor	32	Poor	22	Very Poor
Present study	30	Poor	26	Very Poor	26	Very Poor	30	Poor

Values for various IBI metrics were variable between the 1992 study and the current study. In most instances (75%), the metrics that increased among stations were: the number of sucker species, the number of fishes per 0.5 kilometers, and the proportion of tolerant species (Appendix 2). Three metrics that did not change at any of the stations were the proportion of round-bodied suckers, proportion lithophilic spawners, or proportion of carnivores (Appendix 2).

Water Quality

Water quality variables were measured by the GA DNR - EPD. Data from Station 4 were not obtained in 1998 because of the elimination of a water quality monitoring station that was located nearby. Noteworthy differences between the annual averages of the measured variables were not observed between the 1992 and 1998 sample years (Table 5).

Table 5. Water quality variables measured monthly in the study reach of the Chattahoochee River by the Georgia Department of Natural Resources - Environmental Protection Division during 1992 and 1998. The numbers represent a yearly average and the standard deviation in parentheses.

	Temperature (EC)		Dissolved Oxygen (mg/L)		5-Day BOD (mg/L)		Turbidity (HACH FTU)		Phosphorus (mg/L)	
Station	1992	1998	1992	1998	1992	1998	1992	1998	1992	1998
1	16.39	14.54	9.30	8.95	0.81	1.61	21.36	40.00	0.06	0.28
	(6.4)	(2.97)	(1.36)	(1.61)	(0.35)	(0.79)	(23.93)	(54.10)	(0.03)	(0.28)
2	15.67	16.97	9.73	8.19	1.27	1.95	16.17	70.17	0.18	0.13
	(5.38)	(4.19)	(1.68)	(2.40)	(0.51)	(1.45)	(11.75)	(47.31)	(0.07)	(0.19)
3	16.38	17.24	8.84	8.67	1.49	2.14	18.67	47.00	0.19	0.12
	(4.74)	(4.27)	(1.25)	(0.93)	(0.67)	(1.80)	(10.67)	(69.47)	(0.09)	(0.07)
4	16.75	*	8.05	*	1.19	*	19.00	*	0.19	*
	(5.77)		(2.30)		(0.47)		(9.80)		(0.04)	

* Information was not available because of the removal of a monitoring station.

CHAPTER 5

DISCUSSION

In the present study, IBI scores of the four stations along this reach of the Chattahoochee River indicate that improvement in the biotic integrity of this system was minimal or non-existent. I collected a total of 35 species during my electrofishing sampling; Mauldin and McCollum (1992) reported 29 species (Appendix 1). The previous study revealed that the biotic integrity of this area severely deviated from its pristine condition and would rank in the ‘poor’ to ‘very poor’ categories. Substantial increases of 12 - 20 points in IBI scores from the previous study (Mauldin and McCollum 1992) would be necessary to upgrade the biological integrity categories of this area to ‘fair’ or ‘good’. These substantive increases were not evident during my study. IBI scores increased by 2 - 8 points at three of the four stations compared to those of Mauldin and McCollum (1992) (Table 4). Although substantial improvements in IBI scores were not evident from my study, the results indicate that the biotic integrity of this area is not declining and may have improved slightly at Stations 1 and 4.

Mauldin and McCollum (1992) hypothesized that cold water discharges from Morgan Falls Dam displaced sunfishes downstream, which resulted in a lower than expected catch and a resultant reduction in IBI scores at Station 1. Mauldin and McCollum (1992) also suggested that an unexplained high catch rate of fishes, especially bluegill, inflated IBI scores above expected values at Station 3. These two anomalies in IBI scores from the previous study (Mauldin and McCollum 1992) altered the expected ranking of the four stations by suppressing the score of the least degraded area and inflating the score of an area with considerable environmental perturbations. In my study, I did not observe unexpectedly high or unexpectedly low IBI scores at any of the four stations.

The 12 Metrics

Each of the 12 metrics reveals or emphasizes the presence or absence of various forms of environmental degradation and has a range of sensitivity to these perturbations. Scores for the metric related to the total number of species decreased at Station 3 and remained the same at the three other stations (Appendix 2). The decrease in diversity at Station 3 was mainly from a reduction in sunfish species and other insectivores as compared to the previous study (Mauldin and McCollum 1992).

A decline in sunfish species is usually indicative of poor pool habitat since sunfish rely heavily upon this type of habitat (OH EPA 1988). The metric that involved the number of sunfish species (*Lepomis* sp.) declined from a score of five to three only at Station 3 (Table 4). Scores at the remaining stations remained high (i.e., five). The high scores at Stations 1, 2, and 4 are indicative of good pool habitat (OH EPA 1988). The decline of the score at Station 3 could be caused by disturbances in the food base of these fishes (Karr et al. 1986) (Appendix 2). Since sunfishes reside and feed at all depths of pool habitat, this metric has a moderate to high range of sensitivity (Karr et al. 1986; OH EPA 1988).

The metric involving the number of sucker species (family Catastomidae) is included because most species of suckers are generally long-lived and intolerant of physical and chemical habitat degradation (Karr et al. 1986). Therefore, the slight increase in scores (from 1 to 3) compared to those of the previous study (Mauldin and McCollum 1992) at Stations 1, 3, and 4 may be the result of some improvements in habitat (e. g., water quality, fine sediment reduction) (Table 4).

Residual chemicals from wastewater outfalls produce toxic effects on fishes and other aquatic life (Karr et al. 1985). Consequently, the IBI score for this metric continues to deviate strongly from pristine condition, as reported by Mauldin and McCollum (1992) (Table 4). This metric is highly sensitive to environmental degradation (OH EPA 1988) and may be reflecting Station 2's location immediately below sewer outfalls.

A decline in the number species of fishes that are intolerant to environmental perturbations is a strong indicator of decreased biotic integrity (Karr 1981; Karr et al. 1986). The IBI scores for the intolerant species metric deviated strongly from pristine condition at all four stations (Table 4). The score for Station 3 declined from 3 to 1; metric scores at the other stations were unchanged from the previous study (Mauldin and McCollum 1992) (Table 4). Two of the 11 species (bluestriped shiner *Cyprinella callitaenia* and Alabama hog sucker *Hypentelium etowanum*) sampled in my study but not sampled in the previous study are considered intolerant of environmental disturbance (Dr. Mary Freeman, United States Geological Survey - personal communication). The capture of intolerant species (one fish per station) at Stations 1 and 3 did not affect the score of this metric at these stations. The metric scores of all four stations revealed that any changes in water quality in this portion of the Chattahoochee River were not sufficient to improve the metric scores for this area. Schleiger's (2000) recent modifications to the IBI (Karr 1981; Karr et al. 1986) accommodated stream fishes in west-central Georgia but was not designed for boat-mounted electrofishing. However, incorporating some of the proposed changes could affect this metric by increasing the number of fishes in this region that could be classified as intolerant or moderately intolerant (Schleiger 2000) and would have increased the scores (2 - 4 points) for this metric at Stations 1, 3, and 4. However, these changes would not have been enough to elevate the IBI scores in my study from the poor or very poor range. The number of intolerant species is the most sensitive metric of the IBI and can distinguish among streams that are close to being in pristine condition (OH EPD 1988).

Fishes considered tolerant by IBI tend to thrive in perturbed environments (OH EPA 1988). Decreases in the percentage of tolerant species at Stations 1, 3, and 4 reflect slight increases in water quality at those sites (Table 4). The low (i.e., one) score at Station 2 further suggests that sewer outfalls could be causing detrimental effects to fish populations near these facilities. The sensitivity of this metric was designed to discern habitat quality as conditions degrade to the ‘fair’ to poor categories (OH EPA 1988).

The proportion of the fish population that are round-bodied suckers declines with physical and chemical degradation of rivers (OH EPA 1988). None of the four stations displayed increases in this metric (Table 4). In fact, all of the scores continued to deviate strongly as in the previous study (Mauldin and McCollum 1992). Limited water quality data from the Georgia Department of Natural Resources - Environmental Protection Division suggest an much higher increase in turbidity in 1998 compared to 1992 at the study area (Table 4). Since this metric is particularly sensitive to siltation because of the clean gravel substrate requirement for the spawning of these fishes (Balon 1975), turbidity and siltation apparently are deleterious during critical spawning periods (OH EPA 1988).

A shift in a fish community from specialized trophic guilds to a community composed of an abnormally high percentage of omnivorous species reveals increasing levels of degradation resulting from alterations in the food chain (Karr 1981) (OH EPA 1988). Classification of species into the respective trophic guilds was based on standards that exclude species that show feeding plasticity (OH EPA 1988). Stations 1 and 4 had an increase in scores of this metric, and stations 2 and 3 exhibited the same score as in the previous assessment (Mauldin and McCollum 1992) (Table 4). The score of this metric at Station 2 remained low because of a high relative abundance of white sucker and common carp, which thrive in perturbed eutrophic areas (OH EPA 1988).

A low number of insectivorous fishes is indicative of disturbance of the food base (Karr et al. 1986) and may be attributed to environmental perturbations. Station 1 showed a substantive increase (i.e., from one to five) in the score of this metric. Station 2 also increased in this metric (Table 4). Scores of this metric declined at Station 3, and remained the same at Station 4 as compared to the previous study (Mauldin and McCollum 1992). Mauldin and McCollum (1992) suggested that hypolimnetic discharge from Morgan Falls Dam resulted in cooler average water temperatures and may have suppressed the potential score of Station 1; however, this condition did not seem to suppress the score for this metric during the present study. Low water conditions could have reduced the habitat availability for fishes and also slightly elevated water temperatures, which could have increased the efficacy of my sampling effort. The percentage of individuals that are insectivores metric has a mid-range sensitivity of biotic integrity (OH EPA 1988).

The percentage of top carnivores metric as defined by Karr (1981; Karr et al. 1986) was designed to assess the biotic integrity by evaluating the overall status of the food base. The scores of this metric did not change from those of the previous study (Mauldin and McCollum 1992) at any of the four stations. Consequently, the condition of the upper end of the food chain deviated strongly from pristine condition at Station 1, and deviated moderately at Stations 2, 3, and 4. Cooler average water temperatures could have affected the score of this metric at Station 1 as suggested by Mauldin and McCollum (1992). I agree with their assertion and would expect more top carnivores to occupy this portion of the river since Station 1 is generally more protected from degradation than the other down stream stations. The addition of a keystone predator (i.e., adult striped bass) seems to have had minimal or no effect on the biotic integrity of the system. A cool-water release from Morgan Falls Dam at Station 1 may have provided refuge from warm summer temperatures for striped bass during the summers and suitable habitat for rainbow trout. A study of the summer food-habits of adult striped bass (Hess

2000) revealed that they overwhelmingly selected crayfish and other exotic fishes over the native fish fauna as prey items; and thus, probably did not affect community structure of native fishes nor the resultant IBI scores in this study. Furthermore, only one striped bass was sampled during this study. The imperceptible presence of striped bass probably was a result of the relatively low stocking density of fingerlings stocked in 1990 and 1992, limited reproduction and habitat preferences of these fishes (Coutant 1985, Hess 2000). The metric involving the proportion of fishes that are top carnivores has a moderate to high range of sensitivity (OH EPA 1988).

The metric based on the number of individuals per 0.5 kilometer was designed to incorporate abundance into the IBI since productivity often declines with environmental degradation such as toxic pollutants (OH EPA 1988). This metric excludes species that are tolerant of environmental degradation and has a low to moderate range of sensitivity (Ohio EPA 1988). The scores for this metric increased at Stations 1, 2 and 4, but declined at Station 3. An overall increase in the number of fishes sampled during this study as compared to the previous study (Mauldin and McCollum 1992) and a reduction in the proportion of tolerant species mostly increased scores in this metric. Electrofishing the two stations that were not electrofished during the second sample period could have also increased the score for this metric during my study.

The metric based on the percentage of individuals that require clean gravel substrate for spawning success (i.e., simple lithophils) assesses the condition and availability of this habitat (OH EPA 1988). The scores (i.e., 1) for this metric remained low at all four stations. The lack of improvement in this metric strongly suggests physical habitat degradation at these sites. The values of this metric severely deviated from expected values and were less than 50% of the lowest expected value. This metric is sensitive to physical habitat degradation and may be reflective of problems associated with gravel mining, siltation, and flow alteration (OH EPA

1988). Siltation and erosion in this area probably are the factors limiting improvement of the scores for this metric.

Finally, the percentage of individuals with deformities, erosions, lesions, and tumors (DELT) metric was derived to validate the health of the fishes in the community (Karr 1981). Stations 1 and 2 showed declines from the previous assessment, and stations 3 and 4 remained low. The incidence of abnormalities at stations 1, 3, and 4 were 1.3-3.3 times higher than the highest expected value for DELT. Scores at these stations indicate moderate to high levels of environmental stress are affecting this reach of the Chattahoochee River. A number of pesticides (e. g., metolachlor, diazinon, malathion, etc.) exist within this watershed (Frick et al. 1998) and may be a source of stress to resident biota. The DELT metric is sensitive to the chronic low level affects of chemical pollution (e. g., pesticides, urban run-off, sewer overflows) (OH EPA 1988).

Passive Gear Contributions

Catch of the two passive gears did not affect any of the metrics of the IBI. These gear types probably would not contribute enough data to the IBI calculation to be considered a necessary part of the standard sampling protocol. However, hoopnets could be very useful in obtaining catfishes, which are the preferred species for conducting thorough and detailed fish health examinations used to calculate the metric for deformities, erosions, lesions, and tumors.

Phosphorus Reductions

The decrease in Atlanta's phosphorus load from 720 tons/year in 1990-1991 to 517 tons/year in 1995-1996 (Nungesser and Franz 1997) does not seem to have drastically improved IBI scores in the Chattahoochee River. Phosphorous levels in the river remain higher than the 0.100 mg/L EPA recommendation at Stations 2 - 4 in most of the years studied by

Nungesser and Franz (1997) (Table 5). Median phosphorous levels downstream from Atlanta remain higher than the national median (Frick et al. 1998). There are two major sewer outfall facilities (Atlanta's and Cobb County's) located directly above Station 2. Wastewater treatment facilities are a common source of phosphorous (Jones and Lee 1982). Additionally, there are a least two more sewer treatment facilities above Station 3. Consequently, phosphorous levels are noticeably higher at Stations 2 - 4 than those at Station 1. Phosphorous often is the limiting factor controlling eutrophication, and fairly substantial declines are necessary to elicit improvements in water quality (Jones and Lee 1982). Abundance of common carp *Cyprinus carpio*, which typically flourishes in eutrophic environments, declined slightly from 21% of the total number of fishes in the Mauldin McCollum (1992) study to 17% of the total number of fishes in the current study. Although a substantial increase in the biotic integrity was not observed during this investigation, continued declines were not evident either.

The biota of this reach of the river exhibited a slight increase in biotic integrity; however, caution must be taken when interpreting the increase in IBI scores as part of an assessment of the integrity of this system. IBI scores in the present study were 2-8 points higher than the previous study (Mauldin McCollum 1992); despite this increase, these four stations would still be classified as 'poor' or 'very poor' and suggest that there is still much work to be done to improve the integrity of the river. Metrics of the IBI that showed improvements were on the moderate to low end of sensitivity to environmental degradation. Highly sensitive metrics such as the percentage of fishes that are intolerant continue to deviate strongly from expected values at all Stations (Table 4). Schleiger's (2000) proposed regional modification to the IBI metrics for use in streams in west-central Georgia would have increased the scores of this metric at Stations 1, 3, and 4 by including intolerant and moderately intolerant species that have shown precipitous population declines following environmental perturbations. However, the ranking of these stations as poor or very poor (i.e., about 50% of the possible score) would not have been changed by the additional points generated by Schleiger's (2000) modifications.

The range, order, and change in scores from Station 1 to Station 4 seem to be an accurate depiction of the biotic integrity in this reach of the Chattahoochee River. I expected the scores to be the highest at Station 1, decline at Stations 2 and 3, and increase at Station 4. Station 1 is the most protected area (near Atlanta's potable water intake), and I expected that the highest score would come from this location. Mauldin and McCollum (1992) suggested that dam tail-water releases caused a sampling artifact in their study that suppressed the scores for this station. Stations 2 and 3, were located immediately downstream of a number of sewer outfalls, and I expected the point source pollution to decrease species diversity and biotic integrity of these areas as it has in similar studies near sewers (Katz and Gaufin 1952; Tsai 1973). Several metrics declined at Station 3. Mauldin and McCollum's (1992) cited a possible sampling artifact for the higher-than-expected scores at Station 3 (Table 4). Dilution of point source and non-point source chemical pollution at Station 4 should raise the score higher than those at Stations 2 and 3. My assessment illuminated the severity of the affects of the sewer effluent and other environmental perturbations that exist at Stations 2 and 3. The slight improvements in the scores at Stations 1 and 4 reflect small improvements in integrity as evidenced by improvements in several metrics at the mild to moderate range of sensitivity. However, I believe the integrity of Stations 2 and 3 remains severely compromised as evidenced by either similar or lower scores of several metrics compared to those in the previous study (Mauldin and McCollum 1992).

Periodic biotic integrity assessments of waterbodies are useful indicators of habitat degradation (Karr et al. 1985). Measures to reduce pollution, a growing population of exotic species, and a rapidly growing Atlanta population will have dramatic affects on the biotic integrity of the river in the future. Data from the previous study, along with this study and others like it should provide trend data to help evaluate and manage the Chattahoochee River south of Atlanta.

Although this IBI (OH EPA 1988) has been proven to distinguish among pristine and degraded sites across various large rivers in Ohio, region-specific modifications that incorporate regional fauna of large rivers should yield a more accurate assessment in future studies of is region (Millet et al 1988; Lyons et al. 2001). Furthermore, an IBI specifically modified for flow-regulated systems such as the Chattahoochee River may increase the accuracy of biotic assessment for this area (Bowen 1998).

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Appendix 1. Species of fishes sampled (indicated by x) by electrofishing at four stations (km 483-419) on the Chattahoochee River, near Atlanta, GA during June-July and September of 1998. Fishes are arranged by location and time of capture.

		Station 1		Station 2		Station 3		Station 4		Totals	
Species	Common name	Jul	Sep	Jul	Sep	Jul	Sep	Jun	Sep	fish	percent
Lepisosteus osseus	longnose gar				**			x	**	2	0.24
Amia calva	bowfin				**	x		x	**	4	0.47
Dorosoma cepedianum	gizzard shad		x	x	**	x	x	x	**	32	3.80
Oncorhyrnchus mykiss	rainbow trout	x			**				**	3	0.36
Ctenopharyngodon idella	grass carp		x		**				**	5	0.59
Cyprinus carpio	common carp	x	x	x	**	x	x	x	**	144	17.08
Cyprinella calltaneia	bluestripe shiner				**	x			**	1	0.12
Cyprinella lutrensis	red shiner	x			**		x	x	**	4	0.47
Cyprinella venusta	blacktail shiner	x	x		**	x	x	x	**	21	2.49
Notemigonus crysoleucas	golden shiner			x	**				**	2	0.24
Carpiodes cyprinus	quillback	x			**				**	1	0.12
Catostomus commersoni	white sucker	x	x	x	**				**	51	6.05
Hypentelium etowanum	AL hogsucker	x			**				**	1	0.12

Appendix 1. (continued) Species of fishes sampled (indicated by x) by electrofishing at four stations (km 483-419) on the Chattahoochee River, near Atlanta, GA during June-July and September of 1998. Fishes are arranged by location and time of capture.

		Station 1		Station 2		Station 3		Station 4		Totals	
Species	Common name	Jul	Sep	Jul	Sep	Jul	Sep	Jun	Sep	fish	percent
Moxostoma poecilurum	greyfin redhorse				**	x		x	**	2	0.24
Scartomyzon lachneri	greater jumprock				**	x		x	**	4	0.47
Ameiurus brunneus	snail bullhead				**			x	**	2	0.24
Ameiurus platycephalus	flat bullhead			x	**			x	**	3	0.36
Ameiurus catus	white catfish				**	x			**	1	0.12
Ictalurus punctatus	Channel catfish	x	x	x	**	x	x	x	**	46	5.46
Gambusia affinis	mosquitofish				**	x			**	2	0.24
Labidesthes sicculus	brook silverside				**		x		**	1	0.12
Cottus bairdi	mottled sculpin		x	x	**				**	2	0.24
Morone saxitilis	striped bass	x			**				**	1	0.12
Lepomis auritus	redbreast sunfish	x	x	x	**	x	x	x	**	100	11.86
Lepomis cyanellus	green sunfish	x	x	x	**			x	**	21	2.49

Appendix 1. (continued) Species of fishes sampled (indicated by x) by electrofishing at four stations (km 483-419) on the Chattahoochee River, near Atlanta, GA during June-July and September of 1998. Fishes are arranged by location and time of capture.

		Station 1		Station 2		Station 3		Station 4		Totals	
Species	Common name	Jul	Sep	Jul	Sep	Jul	Sep	Jun	Sep	fish	percent
Lepomis macrochirus	bluegill	x	x	x	**	x	x	x	**	289	34.28
Lepomis microlophus	redear sunfish	x	x	x	**		x	x	**	13	1.54
Lepomis punctatus	spotted sunfish		x		**				**	1	0.12
Micropterus coosae	redeye bass		x		**	x			**	3	0.36
Micropterus cataractae	shoal bass				**		x		**	1	0.12
Micropterus punctatus	spotted bass	x	x	x	**	x	x	x	**	18	2.14
Micropterus salmoides	largemouth bass	x	x	x	**		x	x	**	16	1.90
Pomoxis nigromaculatus	black crappie	x	x	x	**	x	x	x	**	30	3.56
Perca flavescens	yellow perch	x	x		**			x	**	10	1.19
Totals	35	19	18	15	**	16	13	20	**	843	100

** Samples not taken because of low water conditions.

Appendix 2. Values for 12 metrics used to calculate an Index of Biotic Integrity (IBI) for the Chattahoochee River on a 64 km reach downstream of the Atlanta Water-Intake Structure. The total score can range from 12 to 60; the smaller the number, the more degraded the site.

	Station 1		Station 2		Station 3		Station 4	
Metric	1992	1998	1992	1998	1992	1998	1992	1998
Number of Species	3	3	3	3	5	3	3	3
Number of Sucker Species	1	3	1	1	1	3	1	3
Number of Sunfish Species	5	5	5	5	5	3	5	5
Number of Intolerant Species	1	1	1	1	3	1	1	1
% Tolerant Species	1	3	1	1	1	3	1	3
% Round-body Suckers	1	1	1	1	1	1	1	1
% Omnivores	1	3	1	1	3	3	1	3
% Insectivores	1	5	1	3	5	3	3	3
% Top Carnivores	1	1	3	3	3	3	3	3
Number / 0.5 Km	1	3	1	3	3	1	1	3
% Lithophilic	1	1	1	1	1	1	1	1
Deformities, erosion, lesions, or tumors	5	1	5	3	1	1	1	1
IBI Score	22	30	24	26	32	26	22	30