APPLICATION OF AN INDEX OF BIOTIC INTEGRITY TO STREAMS IN THE LOWER OUACHITA MOUNTAINS ECOREGION, ARKANSAS

By

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CHAPTER I

INTRODUCTION

Background

The question, "What are the effects of timber management activities on the ecological health of streams?" has received much attention in research the past several years. Much has been learned about increased nutrient loading, sedimentation and water yield following clearcutting (Beasley et al. 1987; Miller et al. 1988a & 1988b) as well as increased sedimentation from the construction and maintenance of forest roads (Swift 1984; Wooldridge and Larson 1980). However, much is left to be discovered about the long-term and cumulative effects of forest management activities on the aquatic biota.

Concern over our nation's degraded water quality resulted in some of the earliest environmental legislation: The Water Pollution Control Act of 1966; the Federal Water Pollution Control Act Amendments of 1972 (FWPCA); and the Clean Water Act of 1977. Consequently, conspicuous improvements in surface water quality and point source pollution control have been made in the last twenty years. However, we are discovering that the biotic integrity of our water resources has continued to decline (Karr et al. 1985; Judy 1984; Schindler et al. 1989).

Efforts to restore the quality of our water resources have focused primarily on chemical and physical water quality parameters, with the naive assumption that as these criteria improved, improvements in biological quality would always follow (Karr et al. 1986). One nationwide U.S. Environmental Protection Agency (EPA) study found otherwise. This study revealed that 56% of the stream segments with water resource degradation had a reduced fishery potential because of chemical problems; however, 50% were impaired by degradation in physical habitat and 67% by flow alteration (Judy et al. 1984). Likewise, an evaluation of instream biota in Ohio indicated that 36% of impaired sites were not identified using chemical-based criteria (Ohio EPA 1988).

The FWPCA of 1972, commonly known as the Clean Water Act, calls for the restoration and maintenance of the chemical, physical and biological integrity of the nation's waters. Integrity is defined as the quality or state of being complete; unimpaired (Morris 1969). Ecological integrity is attainable only when chemical, physical and biological (or biotic) integrity occur simultaneously (USEPA 1990). Thus, EPA's response to the Clean Water Act is an aquatic ecosystem approach to water-quality monitoring, including the use of integrated physical, chemical, and biological assessment techniques (MacDonald et al. 1991; USEPA 1989; USEPA 1990). Several comprehensive methods have been designed and tested in the evaluation of forest management practices on stream ecological integrity and in the evaluation of forest best management practices or BMPs (Platts et al. 1983; USEPA 1989; Rinne 1990; MacDonald et al. 1991; Clingenpeel 1994; Dissmeyer 1994).

Biotic integrity, a major component of ecological integrity, has been defined as "the ability of an aquatic ecosystem to support and maintain a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of the natural habitats within a region" (Karr and Dudley 1981). Biologists have tried a variety of approaches to assess biotic integrity, including the use of indicator species (Hilsenhoff 1977, 1982; Ryder and Edwards 1985; Schaeffer et al. 1985), diversity indices (Wilm and Dorris 1968; Kaesler et al. 1978; Osborne et al. 1980), the Index of Community Well-Being (Gammon et al. 1981; Yoder et al. 1981; Ohio EPA 1987), relative abundance of desirable species (Coble 1982; Swink and Jacobs 1983), evaluations of physical habitat conditions (Terrell et al. 1982; Stalnaker 1982), and the Index of Biotic Integrity (Karr et al. 1986).

The Index of Biotic Integrity (IBI) is the focal point of this paper. It was designed to use a range of fish assemblage attributes to evaluate stream biotic integrity. The strength of the IBI is its ability to integrate information from individual, population, community, zoogeographic, and ecosystem levels into a single ecologically-based index of the quality of a water resource (Karr et al. 1986). The IBI uses a number of measures, or metrics, that are scored to evaluate biotic integrity of a sample site with respect to conditions found in an undisturbed or reference stream within the same geographic region. High scores indicate sites with minimal perturbation; sites of reduced quality have lower scores (Karr et al. 1986).

Researchers have successfully utilized the IBI to evaluate stream biotic integrity in a variety of geographic regions within the U.S. (Leonard and Orth 1986; Fausch and

Schrader 1987; Hughes and Gammon 1987; Steedman 1988; Bramblett and Fausch 1991). Adapting the IBI to these regions required modifications to accommodate regional differences in fish distribution, assemblage structure and function (Miller et al. 1988). Problems remain in adapting the IBI to regions of low species richness, establishing tolerance rankings and scoring criteria, and adjusting scoring criteria for differences in stream gradient and location of streams of similar size in the drainage network (Miller et al. 1988; Osborne et al. 1992).

In spite of these difficulties, the IBI serves as a functional tool in biological monitoring of water resource quality. As a result, the Tennessee Valley Authority, and the states of Illinois, Kentucky, Ohio, and Vermont have all incorporated the IBI into their monitoring or standards programs (Miller et al. 1988; Ohio EPA 1987). The IBI has also been included in monitoring programs designed to evaluate BMP effectiveness on forest lands (Dissmeyer 1994).

Justification and Objectives

Small streams in the Ouachita Mountains region exist in watersheds where timber management activities have occurred at various levels of intensity since the late 1800's (Smith et al. 1984). Though research has been done to quantify short-term impacts of clearcuts to streams in small Ouachita Mountains watersheds (Beasley et al. 1987; Miller et al. 1988a & 1988b), little has been done to assess cumulative effects of forest management activities on the biotic integrity and ecological health of streams in the Ouachita Mountains. In light of that need, the objectives of this study were to:

1. Modify the IBI to reflect regional differences in fish community structure in mid-sized streams in the Lower Ouachita Mountains Ecoregion.

2. Use the IBI to compare ecological integrity in streams of forested watersheds receiving varying intensities of timber management.

3. Relate differences in IBI scores with corresponding differences in chemical, physical, or aquatic macroinvertebrate community characteristics.

CHAPTER II

LITERATURE REVIEW

Timber management activities can impact streams by increasing sediment and nutrient concentrations; altering streamflow; altering the form and amount of organic material; increasing temperature and autotrophic production; and simplifying channel structure and habitat complexity (Brown and Krygier 1970; Troendle and King 1985; Beasley et al. 1987; Miller et al. 1988a and 1988b; Allan 1995). These changes are often followed by changes in the populations of benthic macroinvertebrates and fish (Boschung and O'Neil 1981; Matlock and Maughan 1988; Bisson et al. 1992).

Habitat simplification results in a decrease in species diversity and composition (Rutherford et al. 1987 and 1992; Bisson et al. 1992). Greater light penetration and autotrophic production results in an increase in standing crop biomass at all trophic levels (Murphy et al. 1981; Hawkins et al. 1982; Clingenpeel 1994; Allan 1995).

Riparian vegetation influences instream water quality through its insulating effect on water temperature. Taxa adapted to cool waters are likely to be eliminated by temperature increases following canopy removal (Barton et al. 1985).

Sedimentation reduces permeability to water movement in interstitial spaces. This, in turn, affects the delivery and removal of gases and nutrients, and restricts movement by aquatic biota (Allan 1995). Additionally, spawning habitat is reduced for many fish species (Beschta 1978; Scrivener and Brownlee 1989; Sweeney 1992), and benthic macroinvertebrate habitat is impaired (Allan 1995).

The effects of logging on channel structure require a much longer recovery period than does recovery of shading due to forest re-growth, which will occur in approximately 10-30 years (perhaps less time in warmer climates). Therefore, the effects of an open canopy do not appear to be as serious as changes to the channel and streambed (Allan 1995).

Biomonitoring/ IBI

Karr (1981) proposed that by carefully monitoring fish communities, the health or "biotic integrity" of a water resource could be assessed. He asserted that biomonitoring was a more sophisticated and environmentally sound approach than merely focusing on contaminant levels. Chemical monitoring misses many anthropogenic impacts such as flow alterations, habitat degradation, and heated effluents (Karr 1981).

Karr et al. (1986) suggested that environmental factors affecting aquatic biota be grouped into five major categories: energy source, water quality, habitat quality, flow regime, and biotic interactions. Efforts to restore or maintain water quality by modifying factors in one of these categories will probably fail if factors in another category limit biotic integrity (Karr et al. 1986). On agricultural lands, for example, efforts to remove pollutants such as pesticides and fertilizers would not result in improved biotic integrity if the stream had been channelized, thus impairing habitat quality.

Several taxa have been targeted in biomonitoring: diatoms (Patrick 1975; Somashakar 1988); benthic invertebrates (Resh and Unzicker 1975; Hilsenhoff 1977; Lang et al. 1989; USEPA 1989); and fish (Karr 1981; Karr et al. 1986; USEPA 1989). Ideally, all of these taxa should be included in a comprehensive biomonitoring program. However, time and financial restraints often prevent such an ideal biomonitoring program.

Fish occur in almost all streams and in a variety of trophic levels throughout the aquatic food web. Therefore, fish community assessment can give an integrated view of watershed condition (Karr et al. 1986). Fish populations are relatively stable during the summer season when most sampling occurs. They are relatively easy to identify, and extensive fish data bases already exist. In addition, the public relates well to programs dealing with fish (Karr et al. 1986).

The IBI is a holistic approach to fish community assessment. It assesses 12 biological attributes or metrics, which combine to assess biotic integrity (Karr et al. 1986). The 12 metrics fall into three main categories: species richness and composition (metrics 1-6), trophic composition (metrics 7-9), and fish abundance and condition (metrics 10-12). Metrics include:

1. Total number of fish species
2. Number and identity of darter species
3. Number and identity of sunfish species
4. Number and identity of sucker species
5. Number and identity of intolerant species
6. Proportion of individuals as green sunfish

7. Proportion of individuals as omnivores
8. Proportion of individuals as insectivorous cyprinids (minnows)
9. Proportion of individuals as piscivores (top carnivores)
10. Number of individuals in sample
11. Proportion of individuals as hybrids
12. Proportion of individuals with disease, tumors, fin damage and skeletal anomalies

The main concept of the IBI is that an ecologically healthy stream has a more diverse and complex fish community than a degraded stream. More complex habitats allow for more biotic complexity and diversity. Polluted streams usually have a simple fish community dominated by a few tolerant species that can out compete less tolerant species (Karr et al. 1986).

Scores of 5, 3, or 1 are assigned to each metric according to whether its value approximates, deviates somewhat from, or deviates strongly from the value obtained at a relatively undisturbed reference site(s) in the same geographic region (Karr et al. 1986). Where undisturbed streams do not exist, it is necessary to depend on least-disturbed streams or sites, along with the use of historical fish collection data. The 12 metric scores are totaled to obtain an IBI score which can then be assigned to the appropriate integrity class (excellent, good, fair, poor, very poor, no fish), which gives a qualitative description of integrity or degradation.

Karr et al. (1986) warn that precautions must be followed in using the IBI. The IBI is most effective when interpreting extensive amounts of data from complex fish communities with biotic integrity assessment as the goal. IBI should not be used, for example, with the objective of single species management. Nor should a single assessment at a single site be used to make a final judgement on the ecological health of the entire stream. IBI is best suited as a screening tool to identify impacted sites which require additional monitoring, and for assessing trends over time at specific sites (Karr et al. 1986).

Professional judgement must be used in every step of the IBI process: selection of appropriate sampling sites and sampling methodology, application of historical fish collection data, modification of IBI metrics, calculation of scoring criteria, and interpretation of results. "The ultimate arbiter of the quality of a sample is a competent ichthyologist or aquatic ecologist who is familiar with the local fish fauna. In addition, users of IBI may want to confer with regional resource managers." (Karr et al. 1986, p. 12).

Species Richness and Composition Metrics

Some of the species richness and composition metrics have required modification in almost all regions where the IBI has been applied, other than the midwestern USA. These metrics were deleted or replaced because the taxa were either not native, were depaurperate, were not present in sufficient abundances, or were not expected in the stream size or habitat (Miller et al. 1988). Other species, such as minnows (Cyprinidae) and sculpins (Cottidae), have been added (Fausch and Schrader 1987; Hughes and Gammon 1987).

Leonard and Orth (1986) applied the IBI to small coolwater streams in the Appalachian Plateau region of West Virginia. A modified IBI based on six fish community attributes closely correlated with independent rankings of stream degradation. This study also related another challenge for use of the IBI: the use of fish tolerance ratings. Species tolerance might vary among geographic locations. Of even more concern, however, is the fact that a species that is intolerant of some forms of perturbation might be tolerant of other forms of disturbance. Leonard and Orth (1986) found that northern hog suckers (*Hypentelium nigricans*) and rock bass (*Ambloplites rupestris*), the least tolerant sucker and sunfish species in Karr's (1981) tolerance ranking, were present in substantial numbers in their second-most degraded stream. They suggested that these species were actually tolerant of poor water quality as long as oxygen concentrations remain high. They also found that central stonerollers (*Campostoma anomalum*), while tolerant of other water quality variations, are sensitive to turbidity and siltation. The fact that tolerance rankings are often specific to the type of perturbation is an area that deserves much attention in further use and development of the IBI (Leonard and Orth 1986).

Trophic Composition Metrics

Of the trophic composition metrics, the proportion of individuals as insectivorous cyprinids has been most often modified (Miller et al. 1988). Leonard and Orth (1986) found that proportions of specialist and generalist species were a more accurate indicator of trophic composition than was the number of insectivorous cyprinids and omnivores. For example, some insectivorous cyprinids, such as the creek chub (*Semotilus atromaculatus*), inhabit the most degraded streams, and some non-omnivorous generalized feeders dominate degraded streams. Feeding strategy should, therefore, be considered as a component of species tolerance (Leonard and Orth 1986). Fausch and Schrader (1987) used the proportion of individuals that were specialized invertebrate feeders as a trophic composition metric for the IBI in Northeastern Colorado.

Leonard and Orth (1986) and Fausch and Schrader (1987) deleted the top carnivore metric because of an absence of carnivores. Hughes and Gammon (1987) replaced the top carnivore metric with proportion of catchable salmonids, because the major carnivore in the Willamette River is tolerant of degraded conditions.

Fish Abundance and Condition Metrics

Almost all IBIs have included a fish abundance metric in which the greatest fish abundances receive the highest score (Karr et al. 1986; Leonard and Orth 1986; Fausch and Schrader 1987; Hughes and Gammon 1987). Scoring criteria for this metric was adjusted in Steedman's (1988) IBI for streams in southern Ontario. High abundances were associated with warm, enriched agricultural streams; very low abundances were associated with degraded urban streams (Steedman 1988).

Certain disturbances, such as clearcutting activity, can result in higher fish (and invertebrate) abundances, which may or may not affect biotic integrity. Hawkins, Murphy and Anderson (1982) found invertebrate abundances to be greater immediately after clearcutting because of increased solar insolation and higher water temperatures. Murphy, Hawkins and Anderson (1981) found higher salmonid biomass and densities to be greater in recent clearcuts compared with forested sections, perhaps because of greater availability of prey and improved foraging in unshaded streams.

Leonard and Orth (1986) noted that some species have a greater incidence of anomalies than others (IBI metric #12). For example, the parasitic trematode blackspot

(*Uvulifer* spp., formerly *Neascus* spp.) has a high incidence of occurrence in central stonerollers (Berra and Au 1978). Certain abnormalities may have a stronger correlation with habitat characteristics than with the degree of stream degradation (Leonard and Orth 1986). High incidence of blackspot disease has been reported for unpolluted, unperturbed streams (Berra and Au 1978). This could be related to the suitability of a stream for abundance of intermediate hosts (Van Duijn 1973). Though Leonard and Orth (1986) recognized this possibility, they felt that this metric (proportion of individuals with anomalies) was still valid. Steedman (1988) found the incidence of blackspot disease to be one of five metrics (along with species richness, local indicator species, abundance of large piscivores, and fish abundance) that cumulatively accounted for 87% of the variation in IBI scores.

Correlation of IBI with Land Use

Steedman (1988) calibrated the modified IBI to land use (urbanization, forest cover and riparian forest) in 10 watersheds on a variety of spatial scales. Significant IBI/land use relationships were found with whole-basin IBI estimates and for IBI estimates from individual stream reaches. Land use immediately upstream of sample stations was most strongly associated with stream integrity as measured by the IBI.

Roth (1994) found a significant relationship between the IBI and land use in an agricultural watershed in southeastern Michigan. The IBI declined with increasing agricultural land use, which explained approximately half of the variation in water quality as measured by biotic integrity (Allan 1995).

Limitations of the IBI

Bramblett and Fausch (1991) modified the IBI for use within the Southwestern Tablelands Ecoregion in southeastern Colorado, to assess effects of U.S. Army mechanized infantry training on the biotic integrity of the Purgatoire River. A significant increase in IBI scores at several sites over a 6-year period resulted primarily because of a decrease in abundance of a tolerant omnivorous fish species.

Because the fish fauna of the Purgatorie River (like other Great Plains streams) is tolerant of the variable physiochemical and low habitat diversity conditions, certain human perturbations that mimic natural environmental variation will likely cause little change (Bramblett and Fausch 1991; Rapport et al. 1985), whereas other perturbations that shift conditions beyond the natural range should cause large changes (Bramblett and Fausch 1991). For example, either moderate additions of sediment from erosion or added flow fluctuations may have little detectable effect. Natural disturbances, such as floods, may also negate other human disturbances such as nutrient input or removal of large woody debris (Bramblett and Fausch 1991). Thus, anthropogenic disturbances likely to degrade the aquatic ecosystem of the Purgatorie River (and other streams of the Great Plains) may have the opposite effect of those that cause change in lotic fish communities of coldwater streams and more mesic systems in the midwestern USA (Bramblett and Fausch 1991).

In regions of low species richness the IBI has proven difficult to apply and has resulted in extensive modification. Adapting the IBI to those regions required that metrics be replaced, deleted, or added to accomodate regional differences in fish distribution and assemblage structure and function (Fausch and Schrader 1987; Hughes and Gammon 1987; Miller et al. 1988).

Biotic integrity now provides a basis for biotic asssessment of surface waters by the Environmental Protection Agency (USEPA 1990), Tennessee Valley Authority (Miller et al. 1988), and numerous states (USEPA 1991). Vermont, Ohio, Kentucky, and Illinois have included the IBI in their monitoring or standards programs (Miller et al. 1988). The EPA has incorporated the IBI into its Rapid Bioassessment Protocols (RBP) designed to provide aquatic biota data for planning and management functions such as screening, site ranking, and trend monitoring (USEPA 1989).

Benthic Macroinvertebrate Indices

Benthic macroinvertebrates have also been used in measuring biotic integrity (USEPA 1989; Ohio EPA 1989; Shackleford 1988). Benthic macroinvertebrates are good indicators of localized conditions; their communities reflect the effects of short-term environmental impacts; sampling is relatively easy; aquatic macroinvertebrates serve as a primary food source for many fish species; they are abundant in most streams; and most state water quality agencies routinely collect such data (USEPA 1989).

Biological impairment of benthic macroinvertebrate communities may be indicated by the absence of generally pollution-sensitive macroinvertebrate taxa such as Ephemeroptera, Plecoptera, and Trichoptera (EPT); excess dominance by any particular taxon, especially pollution-tolerant taxa such as some Chironomidae and Oligochaeta; low overall taxa richness; or significant shifts in community composition relative to the reference condition (USEPA 1989). Most aquatic macroinvertebrate bioassessment indices are based on these concepts (Lenat 1988; USEPA 1989; Kerans and Karr 1994). Criteria used to characterize biological condition using EPA's Rapid Bioassessment Protocols for benthic macroinvertebrates (USEPA 1989) include:

1. Taxa Richness
2. Family Biotic Index or Modified Hilsenhoff Index
3. Ratio of Scrapers/Filering Collectors
4. Ratio of EPT and Chironomid Abundances
5. % Contribution of Dominant Family
6. EPT Index
7. Community Similarity Index
8. Ratio of Shredders/Total

Scoring criteria are generally based on percent comparability to the reference station. Streams or sites assessed are then classified as non-impaired (comparable to the best situation to be expected within an ecoregion); moderately impaired (fewer taxa due to loss of most intolerant forms); or severely impaired (few taxa are present, and if high densities exist they are dominated by one or two taxa) (USEPA 1989).

Habitat Assessment

Habitat, as affected by instream and surrounding topographical features, is a major determinant of aquatic community structure and function (USEPA 1989). Both the quality and quantity of available habitat are critical to biotic integrity. The pattern, distribution, and complexity of habitat types within a stream drainage have a significant affect on fish production and abundance (Gorman and Karr 1978; Foltz 1982; Schlosser 1982; Rankin 1989; Burton 1991; Lobb III and Orth 1991). Many aquatic species have specific habitat requirements, including depth, velocity and substrate type (Moyle and Baltz 1985; Gorman 1987; Allan 1995).

A number of habitat assessment techniques have been utilized (USEPA 1989; Burton 1991; Rankin 1989; Clingenpeel and Cochran 1992). In the Rapid Bioassessment approach (USEPA 1989), habitat assessment is partitioned into primary (substrate and instream cover), secondary (channel morphology) and tertiary (riparian and bank structure) parameters. Habitat characteristics are weighted (primary characteristics receive the strongest values, etc.) to reflect their degree of importance to biological communities. A total score is obtained and is compared to conditions represented by a reference stream or reference site to determine the level of biological health.

The Basin Area Stream Survey (BASS) methodology (Clingenpeel and Cochran 1992) was designed to help monitor the effectiveness of BMPs in the Ouachita National Forest. BASS habitat assessment does not result in a qualitative index as does the RBP approach. BASS quantifies (by actual measurement or visual estimate) elements relating to habitat type, width, depth, bottom substrate, instream cover, and riparian cover. These measurements can be used to compare streams in watersheds under various management regimes to assess BMP effectiveness. In comparing three pairs of streams in managed vs. reference watersheds in the Ouachita National Forest with the BASS methodology, Clingenpeel (1994) found that no single factor was indicative of adverse cumulative effects from silvicultural activities for all three years in which data was collected. However, some associations occurred for two of three years. For example, instream cover and shade canopy was greater for reference streams than for streams in managed watersheds.

Physio-Chemical Water Quality Factors

The U.S. EPA (1989) recommends that water quality assessment be done along with habitat assessment and biosurveys to determine if stream impairment exists. Typical water quality parameters include temperature, dissolved oxygen (DO), pH, conductivity, total suspended solids (TSS) and turbidity. Naiman et al. (1992) found that nitrogen, phosphorus, turbidity, temperature, and intragravel DO were water quality elements fundamental to evaluating ecological stream health. Richards et al. (1993) included concentrations of total nitrate, total phosphate, ortho-phosphate, ammonia-N, and nitrite+nitrate-N as major environmental factors that influence aquatic macroinvertebrate community structure. Clingenpeel and Cochran (1992) recommended analysis of additional water quality elements, including a number of micronutrients and heavy metals.

CHAPTER III

METHODS AND MATERIALS

Streams were chosen for this study with the objective of selecting headwater streams (2nd-3rd order) within the Lower Ouachita Mountains Ecoregion (Omernik 1987; Giese et al. 1987; Clingenpeel 1994) that were similar in size and geomorphology, and located within watersheds that have been managed under a variety of timber management schemes.

The Ouachita Mountains consist of long east-west oriented ridges formed under geologic pressures of folding and faulting. Rock formations in the study area are primarily sandstone and shale. Soils are primarily silty clay and silty loam of medium texture and moderate permeability. These soils are deep in stream valleys and shallow and stony on ridge tops. Forest vegetation is a mixture of oak-hickory and shortleaf pine. The primary land use in the Ouachita Mountains is forestry. Some small-scale farming and mining also occur (Foti 1978).

Mean annual precipitation in the study area ranges from 50-54"(127-137 cm). Mean January temperature is 42-44 degrees F (6-7 degrees C), and mean July temperature is 80 degrees F (27 degrees C) (Foti 1978).

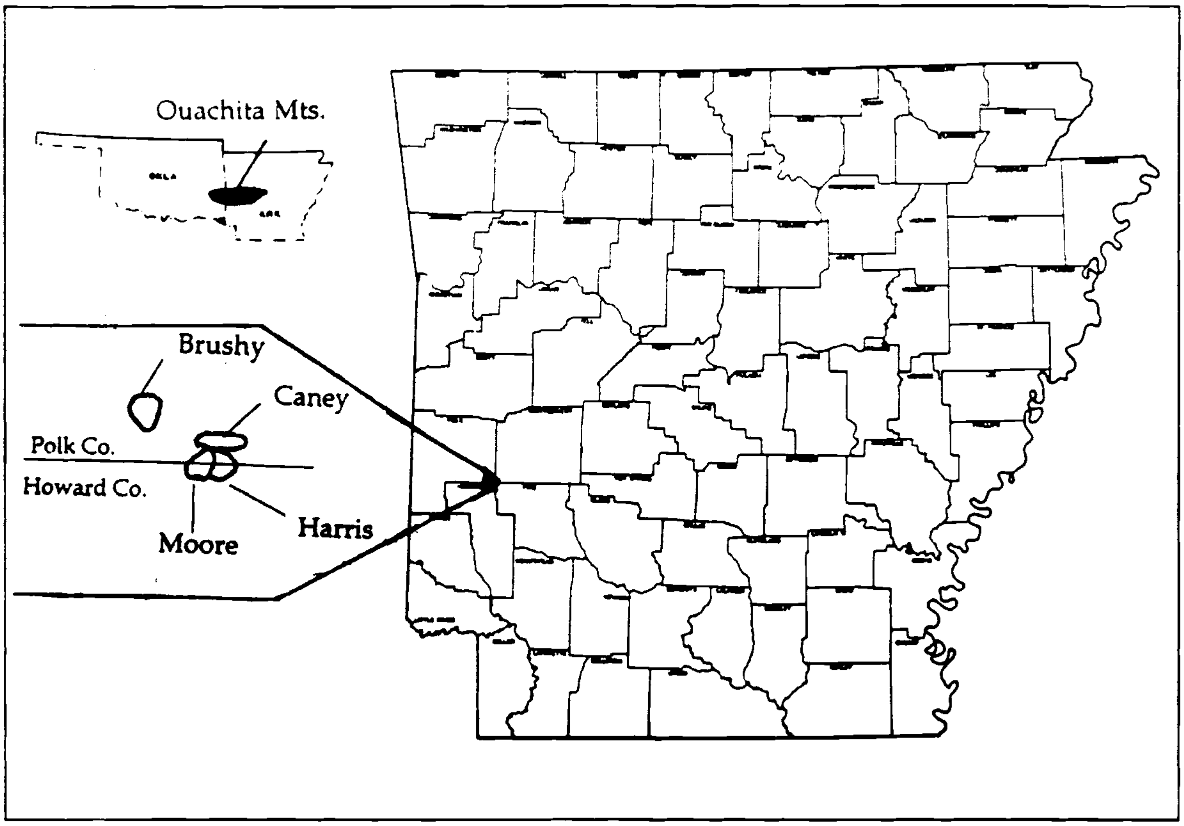


Figure 1. Location of the four watersheds in this study.

The study included portions of four perennial headwater streams that drain into the Cossatot River. Caney and Brushy Creeks are located within the Ouachita National Forest. Moore and Harris Creeks are located on lands owned and managed by Weyerhaeuser Company. The four study streams are located within a 10-mile radius.

Caney Creek, the reference stream, is situated in the Caney Creek Wilderness Area in Polk County, Arkansas. The Caney Creek Wilderness Area is closed to motorized vehicles. Timber has not been commercially harvested since 1948. The majority of the timber in the Caney Creek watershed is in the 81-100+ age classes (Clingenpeel 1994). Any anthropogenic influences on Caney Creek probably result from recreational use. An eight-mile long hiking trail is located adjacent to Caney Creek.

Brushy Creek is also located within the Ouachita National Forest in Polk County. Timber in this watershed has been under an uneven-aged management regime. The majority of trees in this watershed are in the 61-100+ age classes (Clingenpeel 1994). Four private inholdings are located within the Brushy Creek study area. At least two of these inholdings contain some pastureland and livestock. Forest Service roads are located in close proximity to Brushy Creek and cross the creek in several locations.

Moore and Harris Creeks are located south of Caney and Brushy Creeks. Portions of these streams in the study area are located within Polk and Howard Counties, Arkansas. These watersheds contain loblolly pine plantations of varying age classes. Logging roads and bridges intersect Moore and Harris Creeks in a number of locations, and cattle grazing occurs within both watersheds.

Table 1. Stream and watershed characteristics.

	Caney	Brushy	Harris	Moore
Stream length (km)	10.3	8.8	7.1	8.8
Watershed area (km2)	22.8	28.2	26.2	28.7
Riffle-run/pool ratio	1/1.4	1/0.8	1/1.5	1/2.2

Watershed areas were obtained using a Geographic Information System (GIS). Watershed boundaries were digitized into the GIS using 1:24,000 scale USGS topographic maps. A map wheel was used with the same maps to obtain stream lengths.

Stream profiles, elevation versus stream length, were developed using 1:24,000 scale USGS topographic maps (Appendix A). Each stream was divided into three sections based on changes in gradient. Biological sampling and habitat assessment was done every 1000 meters throughout each section (upper, middle, and lower) of each stream. A random number between 1 and 100 was generated to determine the number of meters to move upstream (from the lowest point in the study area) to identify the first area to be sampled on each stream. That same number was used to locate the starting point in the next two sections.

Each sample area consisted of four consecutive habitats (generally two riffle-pool sequences) or reaches. Habitat reaches were separated using block nets. Sampling began with electroshocking, and was followed with physical habitat measurements and macroinvertebrate sampling (Clingenpeel 1994).

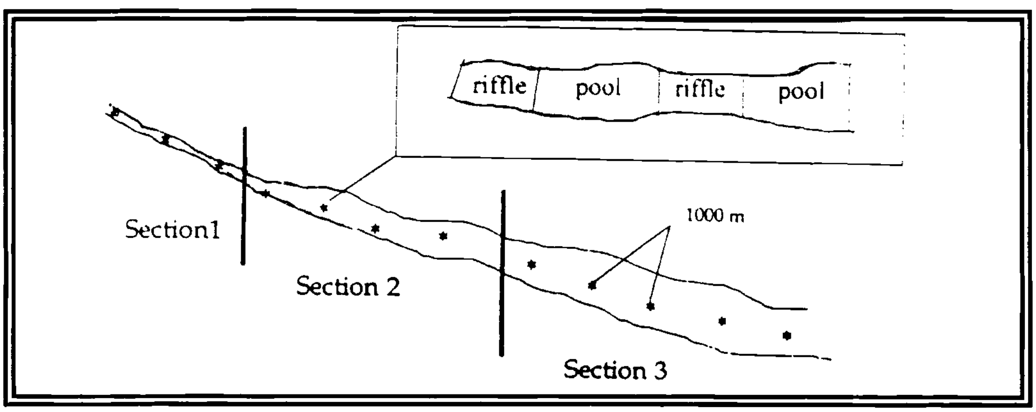


Figure 2. Sampling design.

Habitat Assessment

Habitats within each sampling area were identified and coded using the Ouachita National Forest Habitat Typing Field Guide based (McCain, et al. 1990). This system is based on stream channel morphology and fish habitat utilization research from the U.S. Pacific Northwest. Three habitat types were added to reflect geomorphological differences in the Ouachita Mountains.

Three primary habitat types are distinguished based on water depth: riffles, runs and pools (proceeding from shallow to deep water). 25 habitat types stem from these three main categories. Riffle habitat types are differentiated based on water surface gradient and water depth. Runs are characterized by lower gradient and less surface agitation than riffles. Run habitat types are differentiated based on water velocity and depth. Pools are differentiated at two levels: 1) position of the pool in the stream channel (secondary channel, backwater, lateral, or main channel), and 2) cause of the scour (obstruction, blockage, constriction, or merging flows) (McCain, et al. 1990).

Sample areas for each stream were identified alpabetically, beginning with the lowermost area and moving upstream. Habitat reaches within each area were identified numerically (A1-A4, B1-B4, etc.). Each habitat within each sample area was flagged with the appropriate habitat type number (0-24).

Physical description of each habitat reach was accomplished using BASS methodology (Clingenpeel and Cochran 1992). The length and width of each reach were measured to the nearest tenth of a meter. Mean bank full width was visually estimated to the nearest meter.

Water depths were measured (to the nearest centimeter) along a mid-reach transect at the waters edges, one quarter, half, three quarters across, and at the thalweg.

Substrate composition was estimated and expressed as a percentage of the total area of the reach. Substrate was classified into bedrock, boulder (>30cm), cobble (8-30 cm), gravel (1-8 cm), sand (1mm-1cm), and fines (<1mm) (Bovee and Cochnauer 1977). Embeddedness was visually estimated by examining "sediment lines" on several cobble-sized stones, and estimating the percentage of the stone buried in sediment.

Fish cover was estimated as a percent of the habitat area. Categories included undercut banks, large woody debris (d>0.15 m), small woody debris (d<0.15 m), terrestrial vegetation overhanging the stream (height < 0.3 m. above the water), white water, boulders, bedrock ledges, vegetation clinging to substrate, and rooted vegetation (Platts et al. 1987).

Left and right stream bank angles were measured in degrees using a clinometer (undercut bank = < 90 degrees; vertical bank = 90 degrees; sloping bank = > 90 degrees) (Platts et al 1987). Bank stability was estimated for each bank as a percentage of the bank that was intact and non-erodable. Riparian vegetation was classified as brush, grass, forest or barren. Canopy closure was obtained using a spherical densiometer while facing upstream in the middle of the habitat reach (Clingenpeel and Cochran 1992).

Biological Assessment

Habitat reaches were separated using block nets prior to sampling for fish. Sampling was conducted during late spring and early summer to reduce variation caused by seasonal fish migration and recruitment of young-of-the-year fish (Angermeier and Karr 1986). Fish were sampled by electrofishing with pulsed direct current using a Smith-Root battery powered backpack electrofisher (Hendricks et al. 1989). Each habitat was sampled by making a thorough pass up through and back to the lower end of the reach. One person operated the electrofisher, and three persons netted the stunned fish. Captured fish were identified, counted and released. Representative specimens were collected and preserved in 10% formalin for verification in the lab. Young-of-the-year fishes were excluded because of difficulty in identification and because of scoring bias caused by the influence of recruitment (Karr et al. 1986; Angermeier and Karr 1986).

One aquatic macroinvertebrate sample per sample area (four consecutive habitat reaches) was taken using a D-frame kick net. Kicking of the substrate just upstream of the net was done for five minutes throughout a low gradient riffle or similar habitat (Clingenpeel 1994). Specimens were picked from the net as well as from rocks and organic matter that accumulated in the net. Specimens were preserved in ethyl alcohol for later identification using Merrit and Cummins (1984) and McCafferty (1981). Time and resource limitations prevented the collection of multiple habitat macroinvertebrate samples.

<u>Chemical and Physical Water Quality</u>

Chemical and physical water quality data were collected by sampling one reach in each stream section (three collections per stream) within a two-day period following biological sampling. A Hydac Meter was used to obtain conductivity, water temperature, and pH measurements. A YSI 55 Meter was used to obtain dissolved oxygen (DO) and air temperature readings. Two grab samples of water were collected for subsequent analyses, including TSS, nephelometric turbidity, alkalinity, total nitrates and total phosphorus, using methods described in Greenburg et al. (1985).

Statistical Analyses

Proc GLM in SAS (SAS 1989) was used to perform statistical analyses. Using stream sections as experimental units, analysis of variance (ANOVA) was used to test the equality of IBI means. Parametric tests follow the assumptions of equal variances and normally distributed data. Because these assumptions cannot be made with the use of indices like the IBI, ANOVA was done on both raw IBI scores and on ranks (non-parametric) (Conover 1981). The Fisher Least Significant Difference Procedure (LSD) was then used to isolate statistically significant differences in IBI scores among streams. Contrasts were built to determine if significant differences occurred between the reference and less-managed stream versus the two intensively managed streams; between reference and uneven-aged management stream; and between even-aged management streams.

A correlation analysis was run with IBI, habitat assessment, biological and physio-chemical variables to determine significant linear relationships between IBI scores and these variables, as well as among the variables.

CHAPTER IV

RESULTS AND DISCUSSION

IBI/ Fish

The IBI was modified to reflect differences in fish assemblages in the Cossatot River watershed of the Ouachita Mountains as compared to streams in the midwestern United States, where the original IBI was developed (Table 4 contains the fish list).

IBI metrics that were deleted include the following:

* *Number and identity of darter species.* Only two species of darters were collected, the orangebelly darter (*Etheostoma radiosum*) and logperch (*Percina caprodes*). Only two logperch individuals were collected, and they were from a single site on Moore Creek. Other drainage basins within the Ouachita Mountains Ecoregion contain several darter species, and could utilize this metric. Because darter species are so limited within the Cossatot River Drainage, however, this metric was deleted.

Number and identity of sunfish species. Two common sunfish species were collected, the longear (*Lepomis megalotis*) and green sunfish (*Lepomis cyanellus*). Three bluegill sunfish (*Lepomis macrochirus*) were collected; however, this species has been widely stocked throughout Arkansas, and it is possible that these fish escaped from stock ponds and are not a native species.

The primary use of the IBI in this study was to detect differences in fish assemblages relating to the effects of silvicultural activities. It is doubtful that silvicultural activities would result in the elimination of either of the two common sunfish species in the Ouachita Mountains (Rutherford et al. 1992). Therefore the number of sunfish species metric was deleted. However, the proportion of green sunfish to longear sunfish or smallmouth bass is noteworthy among streams in this study, as later analysis and discussion indicate.

Number and identity of sucker species. Only two sucker species were collected, the creek chubsucker (*Erimyzon oblongus*) and golden redhorse (*Moxostoma erythrurum*); and only three golden redhorse individuals were collected. This metric might be appropriate for larger streams within the Ouachita Mountains Ecoregion where there is a greater abundance of sucker species. The three metrics mentioned above were replaced by the number and identity of minnow (Cyprinidae) species, which will be addressed later in this section.

* *Number and identity of intolerant species.* This metic was replaced with a somewhat less restrictive metic, the number of sensitive species. Only one species collected in this study is considered threatened, rare, or of special concern according to the U.S. Fish and Wildlife Service, Arkansas Natural Heritage Commission, and Arkansas Game and Fish Commission (USDA Forest Service 1990). Only one species collected is described as intolerant according to Karr et al. (1986). For that reason this metric was

broadened to include species that are sensitive, but perhaps not totally intolerant, to environmental disturbances. This metric has been similarly broadened in other geographic areas (Ohio EPA 1988).

* *Proportion of individuals as omnivores.* According to Karr et al. (1986) omnivores are defined as species with diets composed of at least 25% plant material and at least 25% animal material. After consulting the literature, I concluded that only one species collected in this study, the bluntnose minnow (*Pimephales notatus*), should be classified as an omnivore. This species was collected in only two of the sample streams.

* *Proportion of individuals as insectivorous cyprinids.* The premise of this metric is that the relative abundance of insectivores (or insectivorous cyprinids) will decrease with degradation due to changes in the insect abundance resulting from changes in water quality, energy sources, or instream habitat (Karr et al. 1986). Using this metric in my study resulted in the reference stream attaining the lowest score among the four sample streams. This could be due to Caney Creek's size or rate of primary production; or it could relate to the fact that one of the predominant insectivorous cyprinids (*Notropis boops*) is somewhat tolerant to stream impacts and might have a high relative abundance in more impacted sites.

* *Proportion of individuals as hybrids.* Very few fish were identified as being hybrids. This could be due in part to inexperience in field identification. However, it appears that minimal channel alteration has occurred in the study streams. Without major channel modification, habitat heterogeneity usually results in habitat partitioning among fish species and should prevent significant hybridization (Karr et al. 1986). After

examining the fish data (Appendix B), eight metrics were selected to describe fish community structure and function in the Cossatot River Drainage (Table 2). Scoring criteria were then set for each metric according to Karr et al. (1986). For metrics 1-6 values were plotted against stream order for each sample area. A 95-percentile line was fitted by eye, and the area below that line was trisected to achieve qualitative ratings of 5, 3, or 1 (Appendix D). Metrics 1-4 are related to stream order; thus the points produced a right triangle (Karr et al. 1986). Stream order (Strahler 1964) was determined using USGS 1:24,000 topographic maps. Metrics 5 and 6 are not a function of stream order; thus scores are the same regardless of stream size. The trisection method was also attempted for metric # 7. However, values set with this method would have resulted in low scores for all four streams. An analysis of variance (SAS 1989) was run to test the equality of means of fish abundances. There were no significant differences between treatment and reference streams, so all stream sections were assigned scores of 5. General ratings set in Karr et al. (1986) were used to score metric # 8.

Table 2. IBI metrics selected for this study.

Category	Metric
Species richness	1. Total number of fish species
and composition	2. Number and identity of minnow (Cyprinid) species
	3. Number of sensitive species
	4. Proportion of individuals as green sunfish
Trophic composition	5. Ratio of generalist to specialist feeders
	6. Proportion of individuals as top carnivores
Fish abundance	7. Number of individuals in sample
and condition	8. Proportion of individuals with disease or other anomaly

As additional data is examined, perhaps other metrics can be added to the ones I selected. However, as Figure 3 illustrates, these metrics are sensitive to the full range of biotic integrity.

Metric	Low---Biotic Integrity---High
1. Total number of species	----------------------------------
2. Total number & composition Cyprinids	----------------------------------
3. Number sensitive species	-----------------
4. Proportion green sunfish	-------------------
5. Generalists/Specialists ratio	----------------------------------
6. Proportion top carnivores	-----------------
7. Fish abundance	-------------------
8. Proportion fish with anomalies	----------

Figure 3. Range of primary sensitivity for each IBI metric (based on Karr et al. 1986).

A description of each metric and metric scores for each stream are given below:

1. *Total number of fish species (exluding exotics and hybrids).* The number of fish species supported by similar streams in a given region generally decreases with environmental degradation (Karr et al. 1986). This metric has rarely, if ever, been modified.

Scores: Caney - 5; Brushy - 5; Harris - 5; Moore - 5

2. *Number and identity of minnow (Cyprinid) species.* Cyprinid species are abundant throughout Arkansas and represent 58% of the fish species collected in this

study. Cyrinids are a diverse family in terms of trophic composition and tolerance levels, and thus are of great value as ecological indicators of water quality (Robison and Buchanan 1988). Fausch and Schrader (1987) used this metric in an IBI for streams in northeastern Colorado.

Scores: Caney - 5; Brushy - 5; Harris - 5; Moore - 5

3. *Number of sensitive species*. This metric replaced Karr's number of intolerant species. Because this metric was designed to distinguish streams of the highest quality, only species that are highly intolerant to a variety of disturbances should be considered intolerant, because this metric is designed to distinguish streams of the highest quality (Ohio EPA 1988). Because few such intolerant species exist in the study streams, the metric was broadened to include species that are somewhat intolerant to environmental disturbances. Table 4 lists tolerance levels as described for this study. Only those species listed as sensitive were included in this metric.

As is often the case with "categorizing," few distinct boundaries actually exist when it comes to tolerance levels. Some species, such as the central stoneroller, are sensitive to some forms of perturbation and tolerant of others (Leonard and Orth 1986; Ebert and Filipek 1988). The tolerance and trophic groupings listed in Table 4 were compiled after consulting a variety of published sources, focusing on those specific to Arkansas. The list was then reviewed by two noted researchers (H.W. Robison and W.J. Matthews, personal communication).

Scores: Caney - 5; Brushy - 5; Harris - 5; Moore - 3

4. *Proportion of individuals as green sunfish.* In the Midwest the relative abundance of green sunfish increases in degraded streams (Karr et al. 1986). My data and personal observation support this premise for fish assemblages in the Cossatot River drainage. Because the green sunfish is primarily a pool species, this metric could be biased against Moore Creek, which has the lowest riffle-run/pool ratio, a difference which does not appear to result from anthropogenic influences. For that reason, I calculated separate IBI scores for riffles and pools. These two scores were then averaged for the actual metric score. Substitute metrics which would eliminate this bias include proportion of green sunfish to longear sunfish, and proportion green sunfish to smallmouth bass (less tolerant species than the green sunfish). Both of these metrics showed the same trend as did the proportion of green sunfish metric (Table 3).

Scores: Caney - 5; Brushy - 5; Harris - 4; Moore - 4

Table 3. Results of metric #4 and alternative metrics.

Metric	Caney	Brushy	Harris	Moore
% Green sunfish	0.2	0.4	2.2	2.7
Ratio green:longear	1:24	1:17	1:5	1:4
Ratio green:smallmouth	1:8	1:2	2:1	6:1

5. *Ratio generalist/specialist feeders.* Initially, I dealt with this concept in two separate metrics: proportion of specialist feeders and proportion of generalist feeders. For the fish assemblages in this study, however, these two metrics seemed to be directly

inversely related. For that reason I combined the two into a single metric. Most of the sample streams in this study were comprised of insectivorous or invertivorous fish (Table 4), which may indicate a healthy food supply and minimally degraded conditions (Karr et al. 1986). The generalist:specialist metric should be more sensitive to trophic level changes in minimally impacted streams than the insectivorous cyprinid metric (Leonard and Orth 1986). The relative abundance of generalist or opportunistic feeders should increase in degraded sites as specialist feeders emigrate or are eliminated due to impacted food resources.

Leonard and Orth (1986) used the concept of generalist and specialist feeders as two metrics in their IBI. "Generalists" were defined as species that eat a wide range of foods or that adapt readily to shifts in food availability. "Specialists" utilized fewer food resources and exhibited little capability for major diet shifts. Trends and scoring for this metric followed that for insectivorous cyprinids and omnivores metrics in the original IBI (Leonard and Orth 1986).

Because of time and resource limitations, I was unable to categorize fish species under the definitions used by Leonard and Orth (1986). Instead, the specialist category in this study is comprised of benthic specialist feeders, which consisted primarily of orangebelly darters (*Etheostoma radiosum*). Although the orangebelly darter is one of the more adaptable species, it is most abundant in optimal habitat: clear upland streams with a gravel and cobble substrate and moderate to swift current (Robison and Buchanan 1988), and it is considered a management indicator species for the Ouachita Mountains Ecoregion

(USDA Forest Service 1990). Other species that are categorized as benthic specialist feeders include the creek chubsucker, golden redhorse, and logperch.

Because the orangebelly darter, primarily a riffle-run species, is the most abundant of the benthic specialist feeders in this study, I calculated separate scores for riffle-run and pool habitats. These two were then averaged to obtain the actual metric score, thus giving equal consideration to both major habitat types and minimizing differences due to natural morphological variability among streams. This would not be done if the lower riffle-run/pool ratio was caused by anthropogenic activity such as channelization.

Scores: Caney - 5; Brushy - 4; Harris - 3; Moore - 4

Table 4. Tolerance and trophic groups.

Species	Common Name	Tolerance	Trophic Group
Campostoma anomalum	Central stoneroller	M	H
Lythrurus snelsoni	Ouachita Mountain shiner	S	I/G
Lythrurus umbratilis	Redfin shiner	M	I/G
Luxilus chrysocephalus	Striped shiner	S	I/G
Notropis boops	Bigeye shiner	M	I/G
Pimephales notatus	Bluntnose minnow	M	O/G
Semotilus atromaculatus	Creek chub	T	C/G
Erimyzon oblongus	Creek chubsucker	T	I/S
Moxostoma erythrurum	Golden redhorse	S	I/S
Ameiurus natalis	Yellow bullhead	T	C/G
Fundulus catenatus	Northern studfish	S	I/G
Fundulus olivaceus	Blackspotted topminnow	M	I/G
Labidesthes sicculus	Brook silverside	T	I/G
Lepomis cyanellus	Green sunfish	T	C/G
Lepomis megalotis	Longear sunfish	M	I/G
Micropterus dolomieu	Smallmouth bass	S	TC
Etheostoma radiosum	Orangebelly darter	S	I/S
Percina caprodes	Logperch	M	I/S

Tolerance key: S-sensitive; M-moderately tolerant; T-tolerant
Trophic key: H-herbivore; I-insectivore-invertivore; O-omnivore;
C-carnivore; TC-top carnivore; G- generalist; S-specialist

6. *Proportion of individuals as top carnivores.* Viable and healthy populations of such top carnivore species as smallmouth bass, walleye, and pike indicate a healthy, trophically diverse community (Karr et al. 1986). Carnivorous species such as the creek chub were not included in this metric because they are opportunistic and generalist type feeders. In this study the smallmouth bass was the only top carnivore. Smallmouth bass are primarily upland stream inhabitants and are less tolerant to habitat alteration than the other black basses, especially relative to high turbidity and siltation (Robison and Buchanan 1988). This species is also considered a management indicator species for the Ouachita Mountains Ecoregion, with the assumption that population changes may indicate the effects of forest management activities (USDA Forest Service 1990). Scores for this metric were low for all streams. This could have been partially due to sampling inefficiency. The smallmouth bass is a difficult fish to electrofish because of its speed and elusiveness.

Scores: Caney - 3; Brushy - 1; Harris - 3; Moore - 1

7. *Number of individuals in sample.* In similar streams, poor quality sites are generally expected to support fewer fish than sites of higher quality (Karr et al. 1986). However, biomass at all trophic levels often increases following clear-cutting (Allan 1995). Murphy et al. (1981) found salmonid biomass and densities to be greater in recent clear-cuts compared with forested stream sections. My data agree with those findings. Fish abundance was greater in the two more intensively managed watersheds (Table 5).

Scores: Caney - 5; Brushy - 5; Harris - 5; Moore - 5

Table 5. Fish abundance for the four sample streams.

	Caney Creek (reference)	**Brushy Creek** (uneven-aged mgt.)	**Harris Creek** (even-aged mgt.)	**Moore Creek** (even-aged mgt.)
Fish Abundance (individuals/100m^2)	56	45	86	83

8. *Proportion of individuals with disease or other anomaly.* Severely degraded streams often yield a high number of fish in poor health (Karr et al. 1986). Parasitism has been shown to correlate with poor environmental condition (Mahon 1976). Other such conditions include tumors, fin damage or other deformities, discoloration, excessive mucus, and hemorrhaging. Although a small incidence of such abnormalities exists in streams of high ecological integrity, these problems are generally more common in more degraded streams (Karr et al. 1986).

The most common abnormality found in this study was the occurrence of blackspot disease, a parasitic trematode that appears as a tiny black spot in the flesh of fish, especially certain species such as the central stoneroller (Berra and Au 1978). Blackspot disease has been deleted from some IBIs because the presence of this parasite and varying degrees of infestation may be natural and not related to environmental degradation (Berra and Au 1978; Berra and Au 1981; Ohio EPA 1988; Lyons 1992). Incidence and abundance of some parasites could be related to the suitability of a stream for intermediate hosts (Van Duijn 1973). Hosts of the blackspot trematode include the ram's horn snail (*Helisoma*) and the belted kingfisher (*Mergaceryle alcyon*).

Moore Creek is the only sample stream that yielded a high number of fish with anomalies (5.2%), and the vast majority was blackspot disease. Blackspot disease was found in less than 1% of the fish collected in the other three streams. There are at least two reasons besides increased degradation that could attribute to the higher incidence of blackspot disease in Moore Creek:

1) Moore Creek has a greater proportion of pool habitats than the other sample streams: Moore Creek's riffle-run/pool ratio is 37% lower than that of the reference stream, and 33% lower than Harris Creek. Pennak (1953) notes that *Helisoma* spp. exist in all types of habitats, but are most common in quiet waters. For streams, therefore, *Helisoma* is likely to be more prevalent in pools than in riffles. This factor alone would not account for the great increase in occurrence of black spot in Moore Creek, which was over five orders of magnitute greater than the other streams.

2) Although I did not quantitatively measure snail abundance (snails were not included in the macroinvertebrate kick net samples), it is likely that, due to a more open canopy, the rate of primary production is higher in Moore Creek than in the other sample streams, which would support an abundant snail population. Organic suspended sediment was also greatest in Moore Creek. Snails feed on periphyton or detritus (McCafferty 1981), and thus could have been more abundant in Moore Creek. Fish and aquatic macroinvertebrate abundances as well as the relative abundance of central stonerollers (the predominant black spot fish host) were higher in Harris Creek than in Moore Creek, however, and incidence of anomalies in that stream was only 0.3%

Rosenberg and Resh (1993) categorize one species of the ram's horn snail (*Helisoma anceps*) as being somewhat tolerant of organic pollution. Also, a small study done on Baker Creek (Balkenbush et al. 1995), a second order stream in the Cossatot

River drainage, showed the occurrence of blackspot to be significantly higher at a site adjacent to grazing lands as compared to a less disturbed reference site. Riffle/run:pool ratios were similar between sample sites, and a higher proportion of fish from riffle-run habitats were collected than fish from pools, especially at the impacted site. Although this condition merits additional consideration, I feel that the evidence in this study supports the inclusion of blackspot disease in this metric.

Scores: Caney - 5; Brushy - 5; Harris - 5; Moore - 1

Table 6. IBI scores for each metric.

	1	2	3	4	5	6	7	8	Totals	%
Caney	5	5	5	5	5	3	5	5	**38**	**95**
Section 1	5	5	5	5	5	3	5	5	38	95
Section 2	5	5	5	5	5	5	5	5	40	100
Section3	5	3	3	5	5	1	5	5	32	80
Brushy	5	5	5	5	4	1	5	5	**35**	**88**
Section 1	5	5	5	5	4	5	1	5	35	88
Section 2	5	5	5	5	5	1	5	5	36	90
Section 3	3	5	3	5	4	1	5	5	31	78
Harris	5	5	5	4	3	3	5	5	**35**	**88**
Section 1	5	5	5	4	4	3	5	5	31	78
Section 2	5	5	5	3	3	3	5	5	34	85
Section 3	5	5	3	5	4	1	5	5	33	83
Moore	5	5	3	4	4	1	5	1/5	**28**	**70**
Section 1	5	5	3	4	4	3	5	1/5	30	75
Section 2	5	5	3	4	4	1	5	1	28	70
Section 3	5	3	1	4	5	1	5	5	29	73

Because data were obtained from only four streams, I did not feel confident enough to assign integrity classes based on IBI scores to each study stream. By converting the IBI scores to percentages, however, it was possible to fit the scores into integrity classes designed by other researchers (Table 7). According to Karr et al. (1986), Caney, Brushy and Harris Creeks would be considered good to excellent, and Moore Creek would be classified as fair. According to IBI ratings set for Wisconsin streams (Lyons 1992) all four streams would be classified within the excellent integrity class.

Table 7. Integrity classes and related attributes, from Karr et al. (1986).

Integrity class	Attitributes
Excellent	Comparable to the best situations without human disturbance; all regionally expected species for the habitat and stream size are present with a full array of age (size) classes; balanced trophic structure.
Good	Species richness somewhat below expectation, especially due to the loss of the most intolerant forms; some species are present with less than optimal abundances or size distributions; trophic structure shows some signs of stress.
Fair	Signs of additional deterioration include loss of intolerant forms, fewer species, highly skewed trophic structure; older age classes of top predators may be rare.

Lyons (1992) concluded that for the Wisconsin version of the IBI, differences among IBI scores of 10% or less could be caused by sampling error and/or natural variation, and thus are not significant. Differences of 10-25% may indicate true changes in biotic integrity, especially among streams with high IBI scores. Additional sampling or supplementary data may be needed to determine if such differences are actually significant. Differences of greater than 25% probably indicate real differences in biotic integrity and environmental quality (Lyons 1992). Most evident in this study is the difference between Moore Creek and the other three streams. This point will receive further attention in the statistical analyses discussion.

Also noteworthy is the drop in score between sections two and three for Caney and Brushy Creeks (the two smallest streams). Species richness declines rapidly as one moves upstream in a second order section (Harrel et al. 1967). The IBI should not be used for sites where only a few species occur naturally (Lyons 1992).

Habitat Assessment

Habitat assessment should support and complement bioassessment, approximating conditions of a reference or "best attainable" situation (USEPA 1989). Results of the BASS habitat inventory are divided into the following categories: physical dimensions, substrate composition and embeddedness, instream cover, and riparian cover.

Table 8. Overview of average habitat reach dimensions.

Stream	Reach Length (m)	Bankful Width (m)	Water Width (m)	Water Area (m2)	Depth (cm)	Thalweg Depth (cm)	Width to Depth Ratio
Caney	**22.3**	**26**	**4.9**	**118.2**	**16.2**	**32**	**1.6**
Section 1	22.0	36	5.7	125.6	19.8	36	
Section 2	23.6	24	5.1	132.5	17.8	36	
Section 3	19.2	23	3.5	72.7	7.8	16	
Brushy	**27.2**	**36**	**5.5**	**166.7**	**16.4**	**32**	**2.2**
Section 1	35.2	35	7.3	285.5	21.0	42	
Section 2	26.0	36	5.6	144.9	16.8	30	
Section 3	19.0	36	3.0	51.7	10.2	20	
Harris	**23.2**	**40**	**6.5**	**159.7**	**19.6**	**38**	**2.0**
Section 1	21.3	53	7.9	171.2	22.2	45	
Section 2	26.1	40	6.6	186.9	22.2	41	
Section 3	20.1	29	5.1	103.0	13.0	25	
Moore	**22.8**	**38**	**6.2**	**140.2**	**17.6**	**37**	**2.2**
Section 1	21.8	39	9.7	212.1	27.6	60	
Section 2	23.3	40	6.0	136.2	17.0	36	
Section 3	23.0	34	4.4	101.0	12.2	25	

Physical dimensions. Table 8 gives an overview of habitat reach dimensions for each stream. In selecting a reference stream it is not only important to find the least-disturbed stream in the region, but to minimize natural variabiliy caused by such factors as geomorphology, topography, soils, vegetation, etc. The shape of the Caney Creek watershed is long and narrow, with steeper topography and higher elevations than those of the treatment streams. As a result, the width-to-depth ratio is lower for Caney than for the other three study streams. Caney Creek's orientation is unique as well, flowing east to west rather than north to south. Ideally, data from other least-disturbed streams should have been utilized to characterize reference conditions, in order to minimize variability caused by natural differences in stream attributes. However, time limitations did not allow for inclusion of additional data in this study.

Substrate composition and embeddedness. A variety of substrate and habitat types is desirable to support aquatic organisms. The presence of rock and gravel in flowing streams is generally considered the most desirable habitat (USEPA 1989). As shown in Table 9, substrate composition was varied and a high percentage of cobble and gravel occurred in all four streams.

Table 9. Overview of substrate composition and embeddedness (percentages).

	Bedrock	Boulder	Cobble	Gravel	Sand	Fines	Embed.
Caney	**19**	**23**	**28**	**18**	**6**	**6**	**17**
Section 1	21	25	26	17	0	13	15
Section 2	26	23	26	17	5	5	16
Section 3	9	23	31	22	14	2	20
Brushy	**18**	**17**	**35**	**17**	**1**	**12**	**11**
Section 1	17	16	39	23	2	3	6
Section 2	21	14	36	15	0	14	14
Section 3	17	19	30	14	0	20	12
Harris	**20**	**26**	**26**	**16**	7	**5**	**26**
Section 1	23	23	27	15	0	12	27
Section2	19	27	26	16	10	3	29
Section 3	17	29	26	17	10	2	23
Moore	**21**	**26**	**26**	**19**	**1**	**9**	**25**
Section 1	29	31	21	16	2	3	15
Section 2	14	30	33	20	0	6	30
Section 3	20	20	25	21	0	18	30

Embeddedness is the degree to which larger rocks are surrounded by fine sediment. This indicates suitability of the stream substrate as habitat for benthic macroinvertebrates and fish (USEPA 1989). The degree of embeddedness appears to be significantly higher in Harris and Moore Creeks, which could result from an increased sediment load following clearcutting or road construction. This was not substantiated by the percentage of fines found in each stream, which could have been because of sampling error; or much of the fine sediment could have been flushed downstream. A more rigorous quantitative approach is needed to accurately assess such primary habitat parameters as substrate composition and embeddedness.

Instream cover. Large and small woody debris represent a significant energy source in small streams (Vannote et al. 1980). Logs and tree roots also provide excellent

Instream cover. Large and small woody debris represent a significant energy source in small streams (Vannote et al. 1980). Logs and tree roots also provide excellent habitat, as do submerged or emergent vegetation, undercut banks, bedrock ledges, etc. All study streams except Brushy averaged less than 1% woody debris covering habitat reaches. Boulders and bedrock ledges predominated instream cover, especially in Caney and Harris Creeks (Table 10).

Table 10. Habitat reach averages for instream cover characteristics by percentages.

Stream	Undercut Banks	Large Woody Debris	Small Woody Debris	Terrestr. Vegetat.	White Water	Boulders	Bedrock Ledges
Caney	**8**	**0.1**	**0.2**	3	**3**	36	**41**
Section 1	11	0	0	10	2	37	42
Section 2	8	0	0.2	1	4	36	47
Section 3	5	0.4	0.4	0	1	35	24
Brushy	**5**	**0.8**	**0.3**	**3**	**7**	**8**	**26**
Section 1	5	0.1	0.3	1	9	5	11
Section 2	4	0.3	0.4	6	8	8	32
Section 3	4	3	0.3	0	2	14	35
Harris	**9**	**0.1**	**0.7**	**3**	**4**	**40**	**40**
Section 1	10	0.2	1	1	4	30	46
Section2	13	0.1	0.9	6	3	37	48
Section 3	4	0	0.2	2	4	56	20
Moore	7	**0.2**	**0.3**	**2**	4	**17**	**26**
Section 1	2	0	0	0	2	10	16
Section 2	5	0.1	0.1	3	7	31	16
Section 3	11	0.6	0.8	2	1	9	42

Riparian cover. Well-vegetated banks are usually stable regardless of bank undercutting. The ability of vegetation, bedrock and other materials on the stream banks to prevent or inhibit erosion is important in determining stream channel stability and instream habitat. Riparian cover dominated by shrubs and trees provides instream cover, a source of coarse particulate organic matter (CPOM) and solar insulation. Using BASS methodology it was difficult to distinguish how prevalent shrubs were in the habitat, because habitats were classified as either forest, shrub, grass, or barren; not a combination of these types. Although shrubs were present in all four streams, the predominant riparian vegetation for all four streams was classified as forest. Bank stability for the reference stream was 13-20% higher than the treatment streams, which could have resulted in higher erosion and sedimentation in the treatment streams.

The most noticeable difference in riparian cover, however, was canopy closure. Percentage of overhanging vegetation was 17-29% less for treatment streams than for the reference stream. This could have resulted in increased solar penetration and primary production in the treatment streams.

Table 11. Habitat reach averages for riparian cover characteristics.

Stream	Clinging Veg. %	Rooted Veg. %	Left Bank Angle	Left Bank Stability %	Canopy Closure %	Right Bank Angle	Right Bank Stability %
Caney	**2**	**7**	**150**	**76**	**83**	**136**	**78**
Sect. 1	0	11	144	73	77	136	76
Sect. 2	3	7	149	78	83	129	82
Sect. 3	0	2	156	75	88	156	68
Brushy	**2**	**4**	**136**	**68**	**66**	**140**	**59**
Sect. 1	0	5	129	70	49	137	53
Sect. 2	3	4	141	66	69	137	58
Sect. 3	1	2	137	70	85	148	68
Harris	**6**	**8**	**126**	**76**	**61**	**131**	**75**
Sect. 1	1	7	123	80	52	120	70
Sect.2	11	13	135	74	57	129	81
Sect. 3	3	3	114	73	76	147	70
Moore	**0**	**4**	**137**	**66**	**54**	**142**	**69**
Sect. 1	0	1	145	68	28	143	69
Sect. 2	1	3	135	69	45	131	67
Sect. 3	0	8	136	61	78	152	72

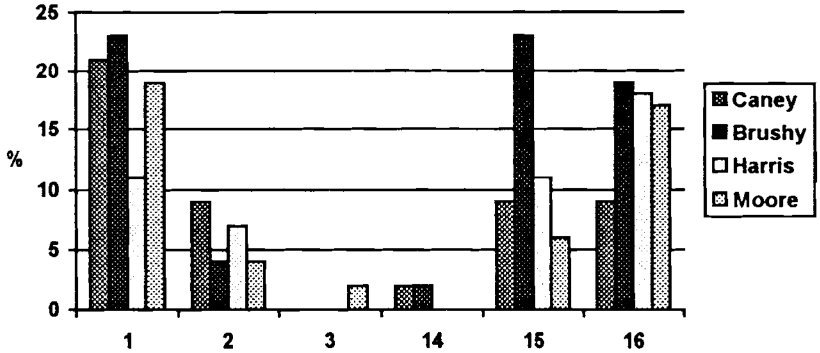
Habitat characteristics such as substrate composition, embeddedness, canopy closure, and instream cover are certainly important in relation to biotic integrity. A higher degree of quantification and/or qualification of these attributes during inventory would probably make such relationships more apparent.

Table 12 shows the proportion of each of the three primary habitat types for each stream. All four streams appear similar in composition, except Brushy Creek, which has a much higher number of runs. This could have been influenced by channel modification caused by a number of road crossings across Brushy Creek, which tends to result in more shallow, homogenous reaches. This factor could also have influenced the lower fish abundance in Brushy Creek.

Table 12. Primary habitat type percentages by stream (by number, not area).

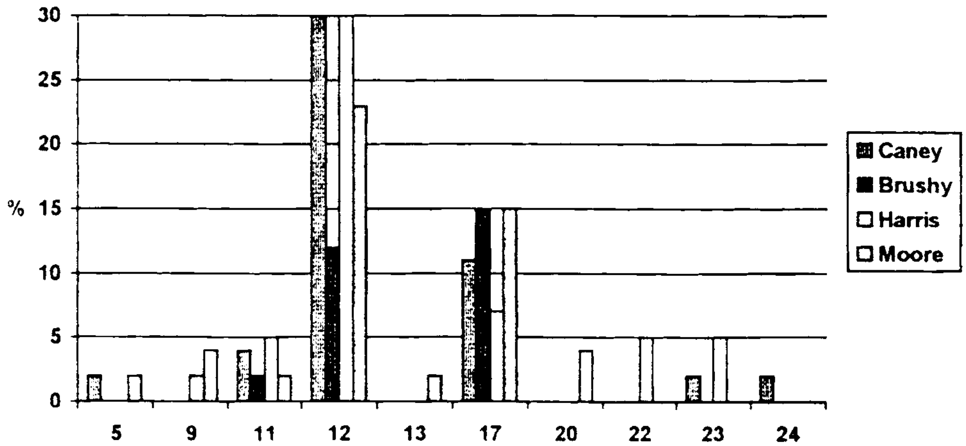
	Riffles	Runs	Pools
Caney	30	20	50
Brushy	27	44	29
Harris	18	29	53
Moore	25	23	52

Figures 4 and 5 break the habitat types down further into specific types of riffles, runs and pools as described using the Ouachita National Forest Habitat Typing Field Guide (McCain et al. 1990).



1-low gradient riffle; 2-high gradient riffle;
3-cascade; 14-glide; 15-run; 16-step run

Figure 4. Riffle-run habitat types by stream: Frequency of occurrence.



5-backwater pool, boulder; 9-plunge pool; 11-lateral scour, rootwad;
12-lateral scour, bedrock; 13-dammed pool; 17-mid-channel pool;
20-lateral scour, boulder; 22-corner pool; 23-step pool; 24-bedrock sheet

Figure 5. Pool habitat types by stream: Frequency of occurrence.

Aquatic Macroinvertebrates

The method selected to assess the macroinvertebrate community was the Rapid Bioassessments of Lotic Macroinvertebrates developed by the Arkansas Department of Pollution Control and Ecology (Shackleford 1988). This method, similar to the EPA Rapid Bioassessment Protocols, uses a combination of semi-quantitative and qualitative measures incorporated into seven biometrics. It has been tested and used extensively in Arkansas (Shackleford 1988). The Mean Biometric Score, which indicates the degree of impairment, can be used to rank monitoring stations in terms of water quality (Shackleford 1988). Biometric scores and corresponding biotic integrity status are shown in Table 13.

Table 13. Biometric scores and integrity status.

Biometric Scores	Mean Biometric Score	Biotic Integrity Status
4	3.5 - 4.0	No impairment indicated
3	2.6 - 3.4	Minimal impairment indicated
2	1.6 - 2.5	Substantial impairment indicated
1	1.0 - 1.5	Excessive impairment indicated

Following is a brief description of each metric.

1. *Dominants in Common (DIC).* A comparison of the dominant tax can be used to identify changes in community structure. Benthic studies have shown that tolerant species are present in almost all streams, but dominate only in degraded systems (Lenat 1988; Shackleford 1988).

2. *Common Taxa Index (CTI).* This qualitative metric deals with the presence or absence of taxa and is expressed by the formula: CTI = TIC/max (Ta,Tb),

where TIC = taxa at common between treatment and reference stream; Ta = total number of reference taxa; Tb = total number of treatment taxa.

3. *Quantitative Similarity Index (QSI).* This metric compares two communities in terms of presence-absence and relative abundances of taxa. The minimum relative abundance is obtained for each taxa that the two sites (reference and treatment) have in common. These values are then summed to obtain the QSI value. Identical communities have a value of 100 and totally different communities have a value of 0.

4. *Taxa Richness.* The number of taxa present is commonly used as a measure of community health (USEPA 1989; Platts et al. 1983). The percent change from the reference to treatment site is used to express the degree of disturbance (Shackleford 1988).

5. *Indicator Assemblage Index (IAI).* The objective of this index is to measure the change in relative abundances of tolerant and intolerant organisms. It integrates pollution tolerance and relative abundance of selected taxonomic groups. In general, Ephemeropterans, Plecopterans and Trichopterans (EPT) are sensitive to pollutants while Chironomids and Annelids (CA) are relatively tolerant (USEPA 1989; ADPCE 1986). The IAI is expressed as follows: IAI = 0.5 (%EPTb/ %EPTa + %CAa/ %Cab), where a = reference stream and b = treatment stream.

6. *Missing Genera.* A decline in the number of taxa from the orders EPT is associated with environmental stress (USEPA 1989). This metric examines the EPT genera that are present at the reference site and absent from the treatment site. It

combines a quantitative measure of richness and a qualitative measure of intolerant groups in measuring the relative change in EPT richness.

7. *Functional Group Percent Similarity (FGPS).* Significant change in an aquatic community's function can indicate interference with the energy flow mechanisms of the ecosystem (Shackleford 1988). Functional groups with respect to the processing of nutritional resources have been established (Merritt and Cummins 1984). These functional groups include shredders, collector-gatherers, filterers, scrapers, macrophyte piercers, and predators. The FGPS determines the relative change in community function between reference and treatment sites by applying the QSI equation to the relative abundances of functional groups.

Table 14 shows individual and mean biometric scores for each study stream. Caney Creek served as the reference site, with which each of the other streams were compared; therefore it has an understood score of 4.

Table 14. Biometric scores.

	# 1	# 2	# 3	# 4	# 5	# 6	# 7	Mean Score
Brushy	4	4	3	4	3	4	4	3.7
Harris	3	3	3	4	4	4	4	3.6
Moore	4	3	4	3	3	4	4	3.6

Mean biometric scores indicate no impairment in any of the managed streams as compared to the reference stream. Individual metric scores indicating minimal impairment for Harris Creek deal primarily with similarity characteristics, whereas those metrics indicating impairment for Moore Creek also involve common taxa, taxa richness, and indicator assemblages. Aquatic macroinvertebrate numbers, relative abundances, and biometric comparisons are found in Appendix C.

It should be noted that macroinvertebrate sampling was restricted to riffle type habitats. Sampling of CPOM macroinvertebrates and pool benthos would have given a more comprehensive view of the macroinvertebrate community structure and function.

Physio-Chemical Water Quality Factors

Table 15 lists the physical-chemical water quality factors measured with mean values for each of the four streams. Overall, minimal impairment is indicated. Differences among the streams are most apparent for turbidity, TSS, and conductivity. Turbidity and TSS each have a statistically significant inverse relationship with IBI score (Table 18).

Table 15. Physical-chemical water quality measures.

	Caney	Brushy	Harris	Moore
Air temperature (C)	25.7	26.9	28.1	26.3
Water temperature (C)	21.5	23.3	21.8	21.8
Dissolved oxygen (mg/l)	8.6	8.1	8.7	8.3
pH	6.5	6.8	7.0	6.7
Conductivity (um/cm)	15.7	20.6	26.1	21.9
Turbidity (NTU)	3.3	6.3	9.7	12.0
Alkalinity (mg/l as CaCO3)	1.9	5.1	7.6	4.8
Calcium (mg/l)	0.6	1.2	1.9	1.1
Magnesium (mg/l)	0.6	0.8	0.7	0.7
Potassium (mg/l)	0.6	0.7	0.7	0.8
Sodium (mg/l)	0.1	1.1	1.9	2.0
Nitrate-N (mg/l)	0.04	0.03	0.05	0.04
Total dissolved phosphorus (mg/l)	0.01	0.01	0.02	0.02
Total suspended solids (mg/l)	1.0	1.2	2.2	4.0
Sediment, fixed (mg/l)	0.6	0.8	1.3	1.7
Sediment, volatile (mg/l)	0.3	0.3	0.8	2.3

Statistical Analyses

Results of parametric and nonparametric analyses were similar (Table 16). Rank transformations for ANOVA resulted in a significance level of .04, adequate to reject the null hypothesis of equal IBI means among streams. The resultant LSD test revealed significant differences (p=.05) between Moore Creek and each of the other three streams.

Table 16. LSD test results for IBI scores (p=.05).

Stream	T Grouping	Mean	N	**Nonparametric** T Grouping	------- Mean
Caney	A	36.7	3	A	9.7
Brushy	A	34.0	3	A	7.8
Harris	B A	32.7	3	A	6.5
Moore	B	29.0	3	B	2.0

Contrasts showed a significant difference (p=.05) between the reference stream and stream in a less intensively-managed watershed versus the two streams in intensively managed watersheds (Caney, Brushy vs. Harris, Moore) (Table 17). Contrasts between Harris and Moore Creeks were significantly different (p=.046); but not between Caney and Brushy Creeks (p=.346).

Table 17. Statistical contrasts among stream IBI scores.

Contrasts	***p* value (parametric)**	***p* value (nonparametric)**
Caney,Brushy vs. Harris, Moore	0.016	0.012
Caney vs. Brushy	0.214	0.346
Harris vs. Moore	0.105	0.046

Table 18. Correlation matrix with IBI, physical, chemical, habitat, and biological variables.

	IBI	pH	Phos	Con	TSS	Turb	Fish	Mac	Can	Fine	Emb	WW	Swd	Lwd
IBI	---	-.28	-.19	-.49	-.69*	-.69*	-.58	-.20	.37	-.13	-.51	-.07	-.22	-.26
pH	-.28	---	-.16	.68*	.47	.58*	.54	.49	-.47	-.25	.40	.48	.43	-.19
Phos	-.19	-.16	---	-.15	-.36	-.18	-.15	-.38	.31	.54	-.30	-.46	-.08	.97*
Con	-.49	.68*	-.15	---	.42	.70*	.52	.47	-.56	-.19	.43	.49	.40	-.18
TSS	-.69*	.47	-.36	.42	---	.76*	.54	.22	-.52	.01	.64*	.39	.30	-.27
Turb	-.69*	.58*	-.18	.70*	.76*	---	.48	.29	-.84*	-.17	.47	.66*	.23	-.21
Fish	-.58	.54	-.15	.52	.54	.48	---	.32	-.13	.06	.71*	.08	.86*	-.02
Mac	-.20	.49	-.38	.47	.22	.29	.32	---	-.15	-.56	.63*	.03	.12	-.42
Can	.37	-.47	.31	-.56	-.52	-.84*	-.13	-.15	---	.35	-.06	-.91	.05	.39
Fine	-.13	-.25	.54	-.19	.01	-.17	.06	-.56	.35	---	-.01	-.38	.23	.64*
Emb	-.51	.40	-.30	.43	.64*	.47	.71*	.63*	-.06	-.01	---	-.09	.54	-.23
WW	-.07	.48	-.46	.49	.39	.66*	.08	.03	-.91*	-.38	-.09	---	-.01	-.56
Swd	-.22	.43	-.08	.40	.30	.23	.86*	.12	.05	.23	.54	-.01	---	.05
Lwd	-.26	-.19	.97*	-.18	-.27	-.21	-.02	-.42	.39	.64*	-.23	-.56	.05	---

* Statistically significant (p = .05)

Phos - Total dissolved phosphorus
Con - Conductivity
TSS - Total suspended solids
Turb - Turbidity
Fish - Fish abundance
Mac - Macroinvertebrate abundance
Can - Canopy closure
Fine - % fines
Emb - Embeddedness
WW - Water width
Swd - Small woody debris
Lwd - Large woody debris

Table 18 is a correlation matrix including all variables that showed significant relationships. Turbidity and TSS showed the strongest correlation with IBI. Fish and macroinvertebrate abundance probably show an inverse relationship with IBI because the two intensively managed streams were higher in fish and macroinvertebrate abundances, but not in biotic integrity. This might also account for the positive correlation between embeddedness and fish and macroinvertebrate abundance.

Conclusion and Recommendations

The objectives of this study were to : modify the IBI to reflect fish assemblage differences within the Lower Ouachita Mountains Ecoregion; to use the IBI to measure biotic integrity in forested watersheds under different management regimes; and to relate differences in IBI scores to corresponding differences in chemical, physical, or aquatic macroinvertebrate community characteristics.

An IBI with eight metrics, sensitive to a wide range of biotic integrity, was developed. The IBI detected differences among the four study streams. These differences seemed to be most strongly associated with increased turbidity and TSS in the treatment watersheds. Metrics which revealed differences in biotic integrity among the four streams were: number of sensitive species, proportion of green sunfish, generalists:specialists ratio, proportion of top carnivores, and proportion of fish with anomalies. Species richness and abundance metrics resulted in high scores for all four streams.

The IBI score for Moore Creek (even-aged management watershed) was significantly lower than scores for the other three streams. Further inspection of this stream could reveal specific impacts (road crossings, grazing pressure, timber management practices) which might be contributing to reduced biotic integrity. The IBI could then be used to assess specific sites and impacts. Further monitoring would be necessary before making a judgment on the ecological health of Moore Creek. IBI scores for the other treatment streams indicated good to excellent biotic integrity.

Aquatic macroinvertebrate analyses indicated no impairment for all four study streams according to the Rapid Bioassessments of Lotic Macroinvertebrates (Shackleford 1988). However, sampling was limited to riffle-run habitats. Sampling of CPOM and pool benthos might reveal more significant differences in macroinvertebrate community structure among the sample streams.

Habitat assessment showed that Moore and Harris Creeks are larger streams than Brushy and Caney Creeks. This factor, as well as more intensive silvicultural practices, could contribute to the more open vegetative canopy over Moore and Harris Creeks. Greater fish and macroinvertebrate abundances in these two streams was probably a result of increased solar penetration and primary production. Embeddedness was greater on all treatment streams as compared to the reference stream, and was highest on Moore and Harris Creeks. None of the habitat characteristics showed a statistically significant relationship with IBI scores. More rigorous measurement techniques might reveal significant relationships, especially concerning characteristics like embeddedness and substrate composition.

Data from this study supports the use of the IBI in the assessment of silvicultural and related impacts in the Lower Ouachita Mountains Ecoregion. Further refinement should enable its use within a comprehensive water quality management program to assess site impacts from timber management and related activities; to monitor biotic integrity within streams over time; and to assess effectiveness of forestry best management practices on stream ecological integrity and water quality goals.

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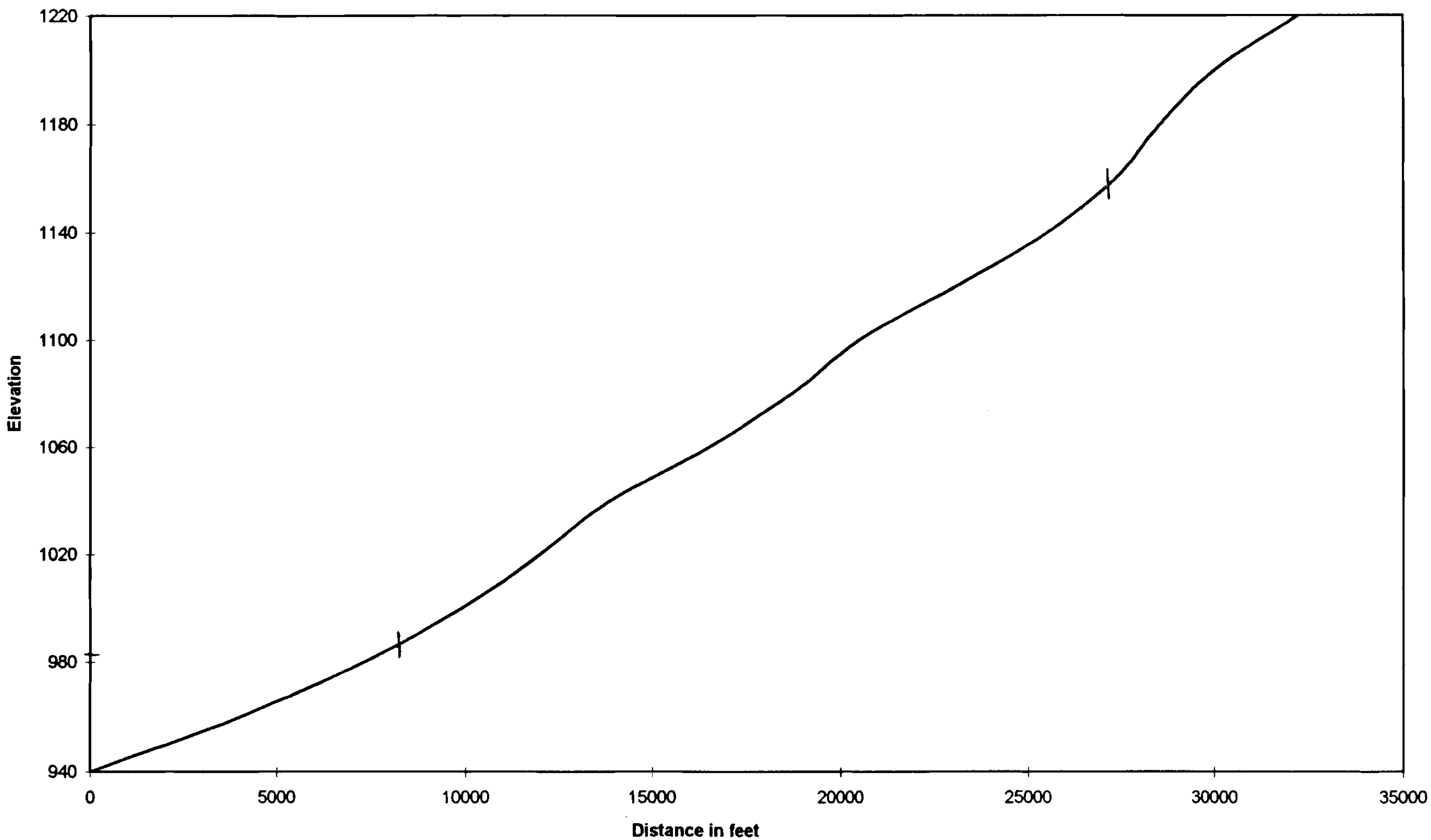
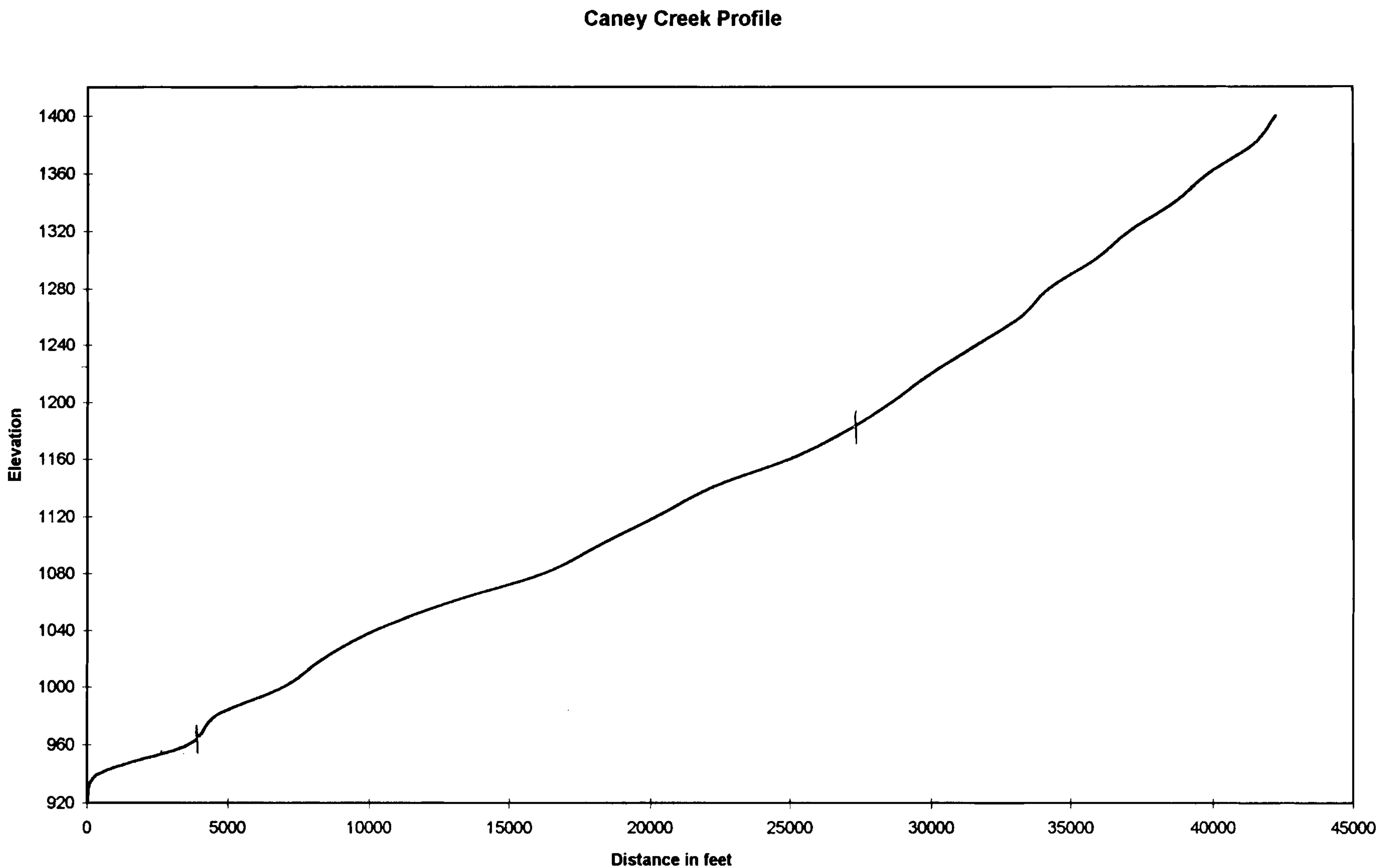
APPENDIX A

STREAM PROFILES

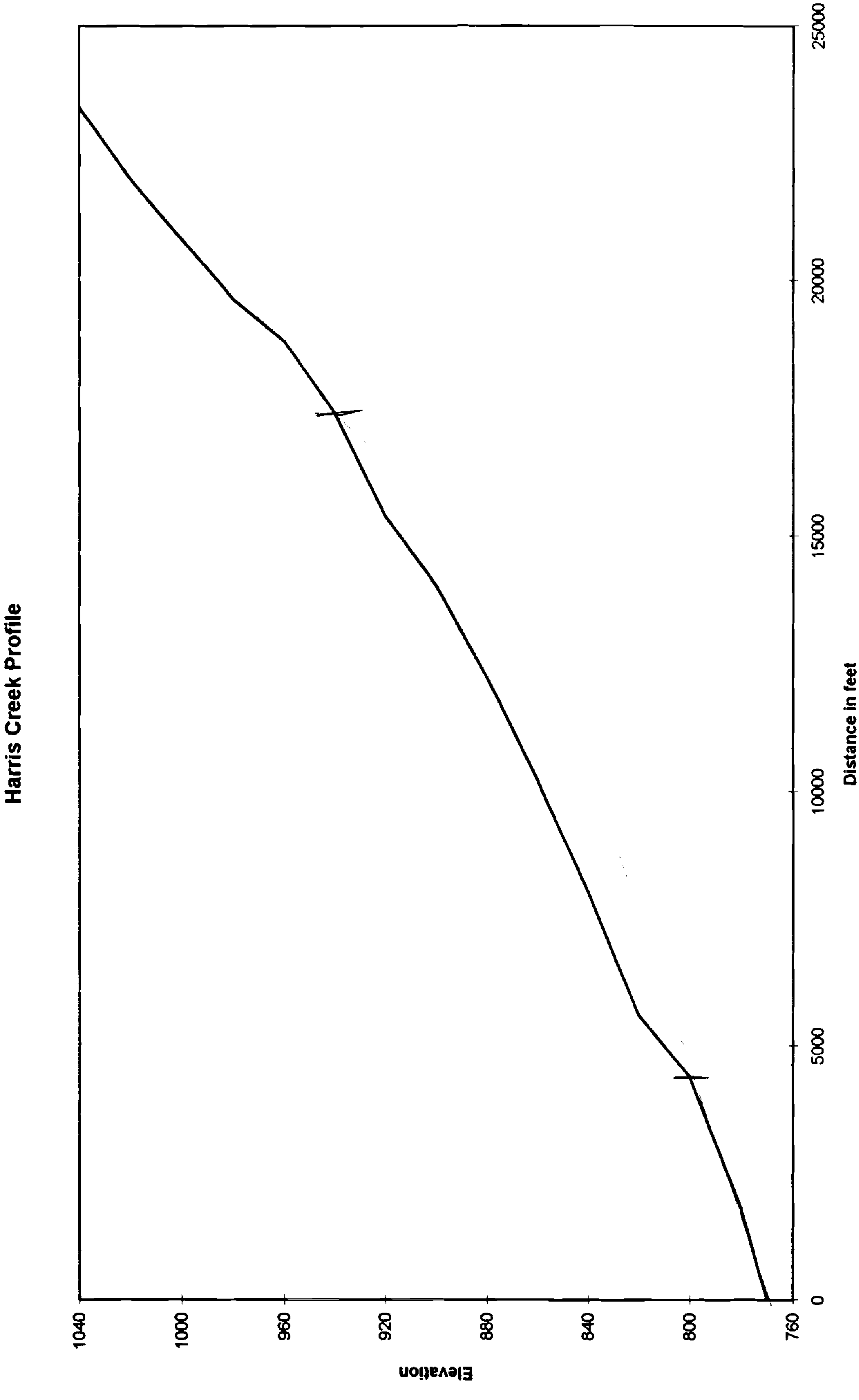
Brushy Creek Stream Profile





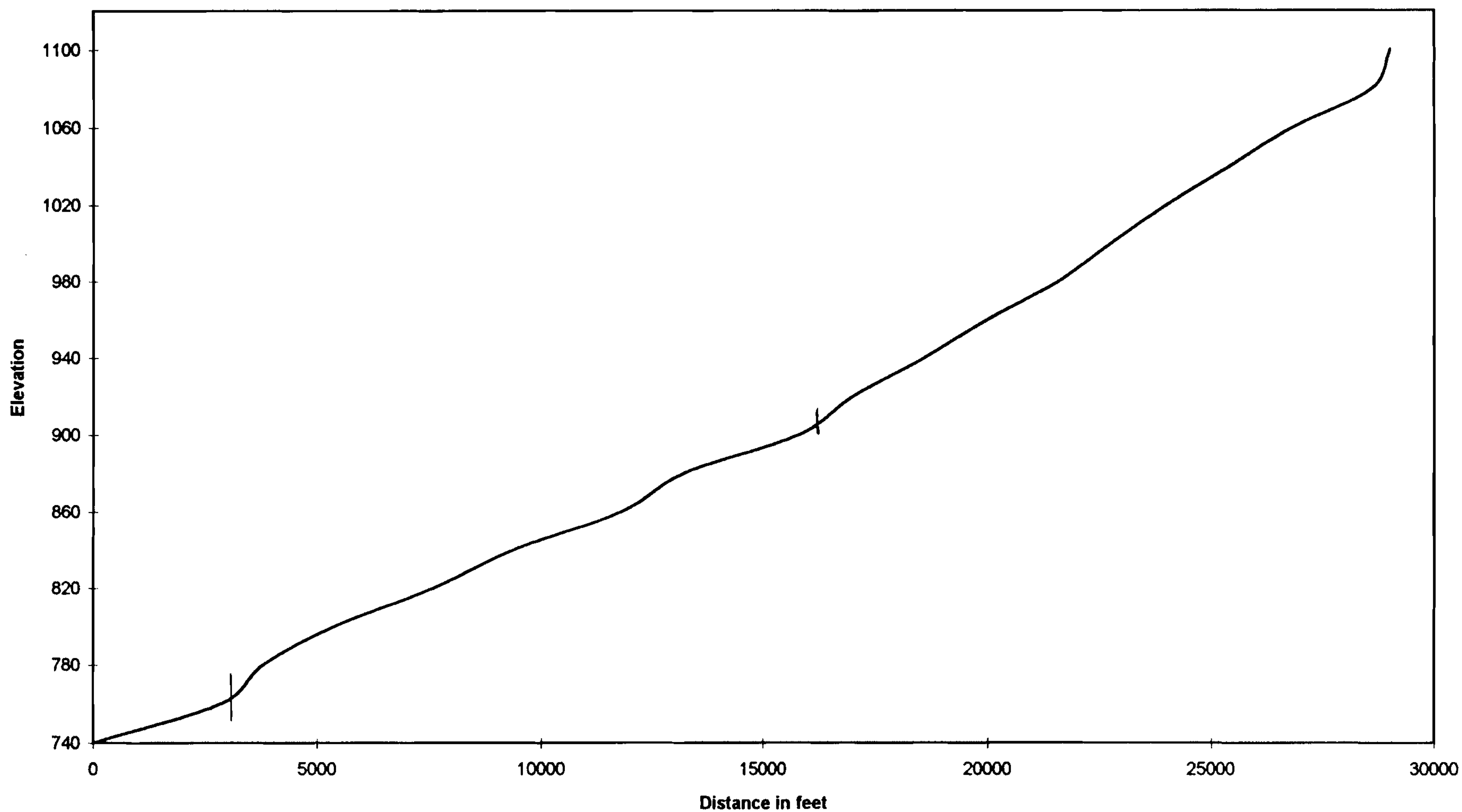
Caney Creek Profile
Elevation
1400
1360
1320
1280
1240
1200
1160
1120
1080
1040
1000
960
920
0
5000
10000
15000
20000
25000
30000
35000
40000
45000
Distance in feet

70



Harris Creek Profile
Elevation
1040
1000
960
920
880
840
800
760
0
5000
10000
15000
20000
25000
Distance in feet

Moore Creek Profile



APPENDIX B

FISH BY HABITAT TYPE PER STREAM

Brushy (A-M)

Fish Per Habitat Type

	Riffle ------------	Run ----------------------			Pool --------------------			Total	Relative Abundance		
	1	**2**	**14**	**15**	**16**	**11**	**12**	**17**		Section 1	Section 2	Section 3
Bigeye Shiner	29	18		43	20		17	8	135	6	3	0
Blackspotted Topminnow	1						3	15	19	1	0	0
Bluegill								1	1			
Central Stoneroller	218	26	70	125	248	1	38	88	814	17	27	6
Creek Chub	131	11	52	146	270	24	48	33	715	1	18	53
Green Sunfish	1						2	13	16	1	0	0
Longear Sunfish	2	14		21	13		58	159	267	17	3	0
Northern Studfish	1	1	6	2	4		12	4	30	1	1	0
Orangebelly Darter	473	100	47	238	463	2	122	86	1531	36	40	39
Ouachita Mt. Shiner				4	69		22		95	8	0	0
Smallmouth Bass	2	6		7	5		7	9	36	2	1	0
Striped Shiner	59	31		33	26		34	19	202	9	4	0
Yellow Bullhead	3	1	3	9	21	3	8	15	63	1	2	2
									3924	100	99	100

13 species

Caney (A-N)

Fish Per Habitat Type

	Riffle		Run				Pool					Total	Relative Abundance Section 1	Section 2	Section 3
	1	**2**	**24**	**14**	**15**	**16**	**5**	**11**	**12**	**17**	**23**				
Bigeye Sh	11	5	5	0	42	17	1	0	72	10	0	163	6	6	0
Stoneroller	275	87	7	3	209	85	33	20	425	73	63	1280	48	31	34
Creek Ch	126	4	2	5	14	30	0	65	126	41	34	447	2	7	27
Green SF	1	0	0	0	0	0	0	0	5	1	0	7	0	0	0
Longear SF	2	0	17	5	4	1	0	2	125	9	0	165	3	7	1
N. Studfish	1	1	0	1	0	0	0	3	1	1	1	9	1	0	0
OB Darter	502	100	13	34	100	137	7	56	284	21	22	1276	31	36	35
OM Shiner	0	0	0	0	0	0	0	0	4	3	0	7	1	0	0
SM Bass	6	2	3	1	5	4	0	1	30	5	0	57	2	2	0
Striped Sh	34	7	1	2	23	12	0	0	65	4	3	151	5	6	1
Y. Bullhead	25	1	1	2	3	10	0	5	66	5	0	118	1	4	2
												3680	100	99	100
11 species															

Harris Creek

Fish per Habitat Type

	Riffle		Run		Pool							Total	Relative Abundance		
													Section 1	Section 2	Section 3
	1	**2**	**15**	**16**	**9**	**11**	**12**	**13**	**17**	**22**	**23**				
Bigeye Sh	120	125	205	156	7	42	100		23	10	69	857	20	11	8
Bluegill			1				1					2	0	0	0
Bluntnose		3	10									13	1	0	0
Stoneroller	373	322	432	455	4	38	313		40	35	373	2385	44	45	13
Creek Chu	36	1	45	257	57	1	135		2	104	38	676	0	7	47
Creek CS				3		1	10					14	0	0	0
Gold Redh				3								3	0	0	0
Green SF	3	6	6	11		7	54		36	8	2	133	1	3	1
Longear	2	25	68	19	2	46	369		158	20	8	717	12	14	5
N. Studfish	2	1	27	1		1	7		1		1	41	1	0	0
OB Darter	202	95	177	331	8	17	97		29	23	41	1020	16	15	23
OM Shiner	1	8	35				12		7			63	2	1	0
SM Bass	10	11	7	2		2	21		4	1	2	60	1	1	0
Striped Sh	1										1	2	0	0	0
Y. Bullheac	1	5	7	7	3		19		8	14	3	67	1	1	2
												6053	99	98	99

15 Species

Moore Creek

Fish per Habitat Type

	Riffle			Run		Pool								Total	Relative Abundance		
	1	2	3	15	16	5	9	11	12	13	17	20	22		Section 1	Section 2	Section 3
Bigeye Sh	15	15	95	88	98	12	44	70	37	5	29	6		514	14	6	8
BS Topmin			7											7	1	0	0
Bluntnose	1			17	2		7	17	15	1	4	1		65	0	1	1
B. Silversi			1		1	1			7		8			18	1	0	0
Stoneroller	375	175	69	343	402	8	53	100	413	21	284	59		2302	36	45	32
Creek Chu	100		1	64	117			5	91		154			532	1	2	18
Creek CS		2		9	2			30	27		35			105	0	0	4
Green SF	8	6		9	24	4	13	20	41	5	31	3		164	3	3	2
Logperch									2						0	0	0
Longear	28	33	14	31	94	7	32	40	258	19	143	35		734	25	16	4
N. Studfish					1						1	1		3	0	0	0
OB Darter	467	68	11	160	270	4	4	20	239	5	241	12		1501	15	25	28
Redfin Sh	1		3	6	27		2	16	5		4			64	2	0	1
SM Bass	3		4	2	3		1		7	2	4	1		27	1	1	0
Y. Bullheac	6				8		1	4	17		5	1		42	1	1	1
															100	100	99
15 Species																	

APPENDIX C

AQUATIC MACROINVERTEBRATE DATA

AND BIOMETRICS

Aquatic Macroinvertebrate Numbers and Relative Abundance Per Stream											
				Caney--------		Brushy--------		Harris----------		Moore----------	
ORDER	FAMILY	GENUS	STAGE	No.	RA	No	RA	No.	RA	No.	RA
Coleoptera	Psephenidae	Psephenus	larvae	277	19	296	27	270	17	328	19
	Elmidae	Optioservus	adult	52	3	7	1	23	1	10	1
	Elmidae	Stenelmis	adult	56	4	16	1	28	2	53	3
	Elmidae	Ordobrevia	larvae	18	1	0	0	10	1	14	1
	Elmidae	Ancyronyx	adult	1	0	0	0	0	0	0	0
	Gyrinidae	Dineutus	adult	1	0	0	0	8	1	1	0
	Gyrinidae	Gyrinus	adult	0	0	0	0	0	0	1	0
	Dytiscidae	Oreodytes	adult	0	0	0	0	0	0	1	0
Diptera	Tipulidae	Hexatoma	larvae	40	3	13	1	1	0	6	0
	Empididae		larvae	29	2	10	1	12	1	15	1
	Dixidae	Dixella	larvae	0	0	0	0	0	0	3	0
	Ephydridae		larvae	0	0	1	0	0	0	0	0
	Chironomidae		larvae	3	0	42	4	39	2	77	5
	Simuliidae		larvae	2	0	1	0	0	0	3	0
	Tabanidae		larvae	0	0	0	0	1	0	3	0
Nematocera			pupae	3	0	0	0	0	0	0	0
Ephemeropter	Heptageniidae	Macdunnoa	larvae	142	9	223	20	290	19	243	14
	Oligoneuriidae	Isonychia	larvae	25	2	61	6	180	12	161	9
	Siphlonuridae	Ameletus	larvae	3	0	23	2	9	1	3	0
	Baetidae	Baetis	larvae	0	0	1	0	3	0	2	0
	Baetidae	Pseudocloeon	larvae	12	1	18	2	2	0	7	0
	Caenidae	Caenis	larvae	0	0	0	0	2	0	3	0
	Leptophlebiid	Leptophlebia	larvae	0	0	1	0	1	0	7	0
	Ephemerellida	Attenella	larvae	0	0	0	0	0	0	1	0
	Ephemerellida	Serratella	larvae	0	0	1	0	0	0	0	0
Hemiptera	Gerridae	Gerris	adult	3	0	1	0	2	0	15	1
	Gerridae	Trepobates	adult	3	0	1	0	0	0	0	0
	Gerridae	Metrobates	adult	1	0	0	0	0	0	0	0
	Nepidae	Ranatra	adult	1	0	0	0	0	0	1	0
	Belostomatida	Lethocerus	adult	1	0	0	0	0	0	0	0
	Belostomatida	Belostoma	adult	1	0	0	0	1	0	0	0
	Veliidae	Rhagovelia	adult	1	0	4	0	0	0	2	0
	Veliidae	Microvelia	adult	0	0	0	0	1	0	0	0
	unknown		nymph	1	0	0	0	0	0	0	0

Aquatic Macroinvertebrate Numbers and Relative Abundance Per Stream

				Caney--------		Brushy--------		Harris----------		Moore----------	
Hirudinoidea	Glossiphoniidae		adult	1	0	7	1	0	0	1	0
	Hirudinidae		adult	0	0	1	0	3	0	5	0
Isopoda	Asellidae		adult	2	0	0	0	0	0	0	0
Lepidoptera	Pyralidae	Petrophila	larvae	0	0	1	0	0	0	0	0
	unknown		pupae	0	0	0	0	0	0	1	0
Megaloptera	Corydalidae	Corydalus	larvae	18	1	45	4	34	2	27	2
	Corydalidae	Nigronia	larvae	22	1	13	1	14	1	16	1
	Sialidae	Sialis	adult	0	0	0	0	0	0	1	0
Odonata	Gomphidae	Gomphus	larvae	7	0	8	1	22	1	11	1
	Gomphidae		adult	0	0	0	0	1	0	1	0
	Gomphidae	Hagenius	larvae	9	1	1	0	2	0	2	0
	Aeshnidae	Boyeria	larvae	0	0	2	0	0	0	0	0
	Calopterygida	Calopteryx	larvae	0	0	0	0	0	0	2	0
	Coenagrionida	Argia	larvae	0	0	4	0	11	1	6	0
Plecoptera	Perlidae	Neoperla	larvae	283	18	119	11	51	3	314	18
	Perlidae	Claassenia	larvae	5	0	6	1	0	0	2	0
	Perlidae	Acroneuria	larvae	4	0	14	1	3	0	13	1
Trichoptera	Philopotamida	Chimarra	larvae	58	4	19	2	41	3	126	7
	Hydropsychida	Cheumatopsy	larvae	392	26	60	5	330	21	137	8
	Hydropsychid	Cheumatopsy	pupae	0	0	1	0	1	0	0	0
	Helicopsychid	Helicopsyche	larvae	5	0	1	0	38	2	37	2
Prosobranchia (sbclass)			adult	1	0	47	4	119	8	9	1
Oligochaeta (class)			adult	48	3	37	3	10	1	33	2
				1531	100	1106	100	1563	100	1704	100

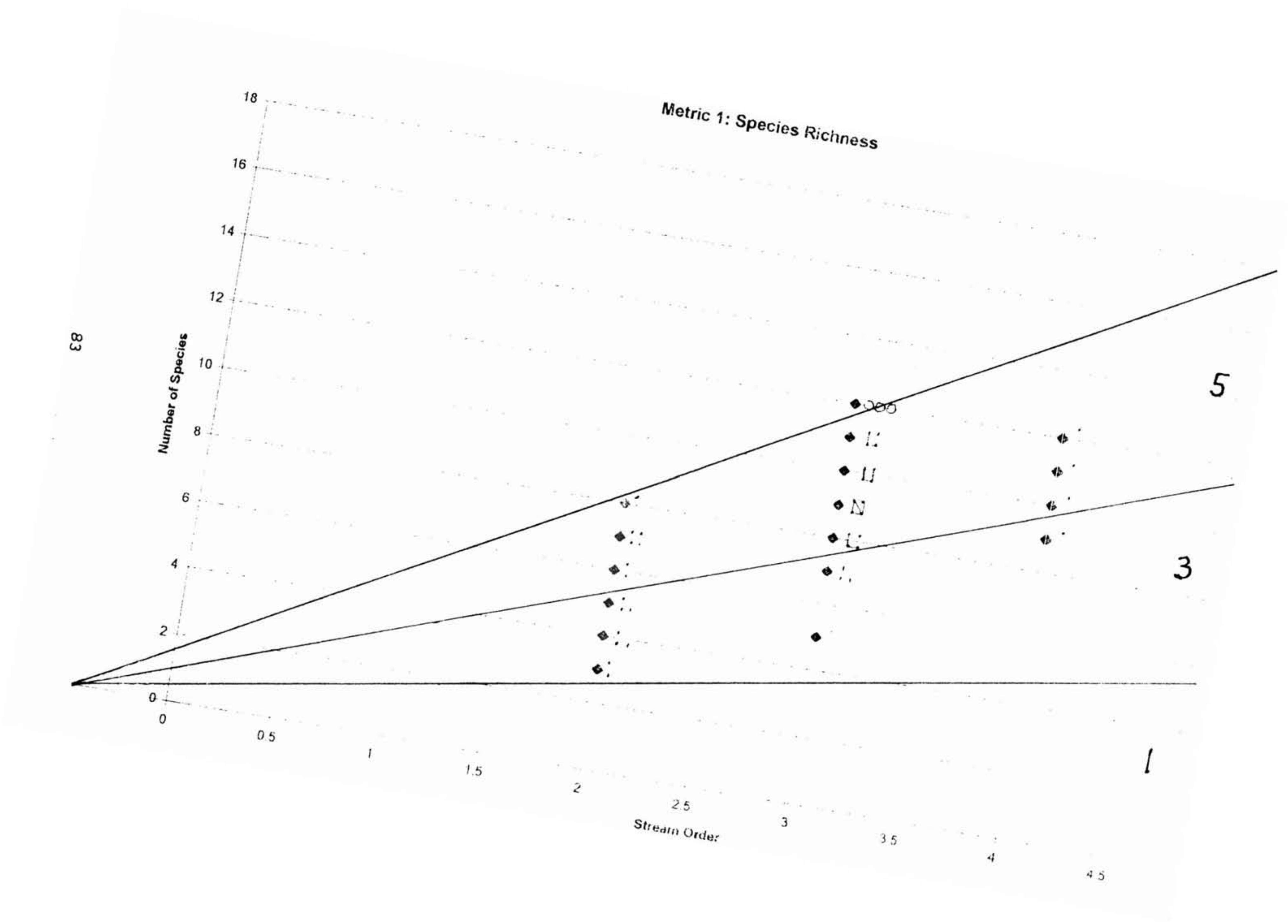
Aquatic Macroinvertebrate Biometrics

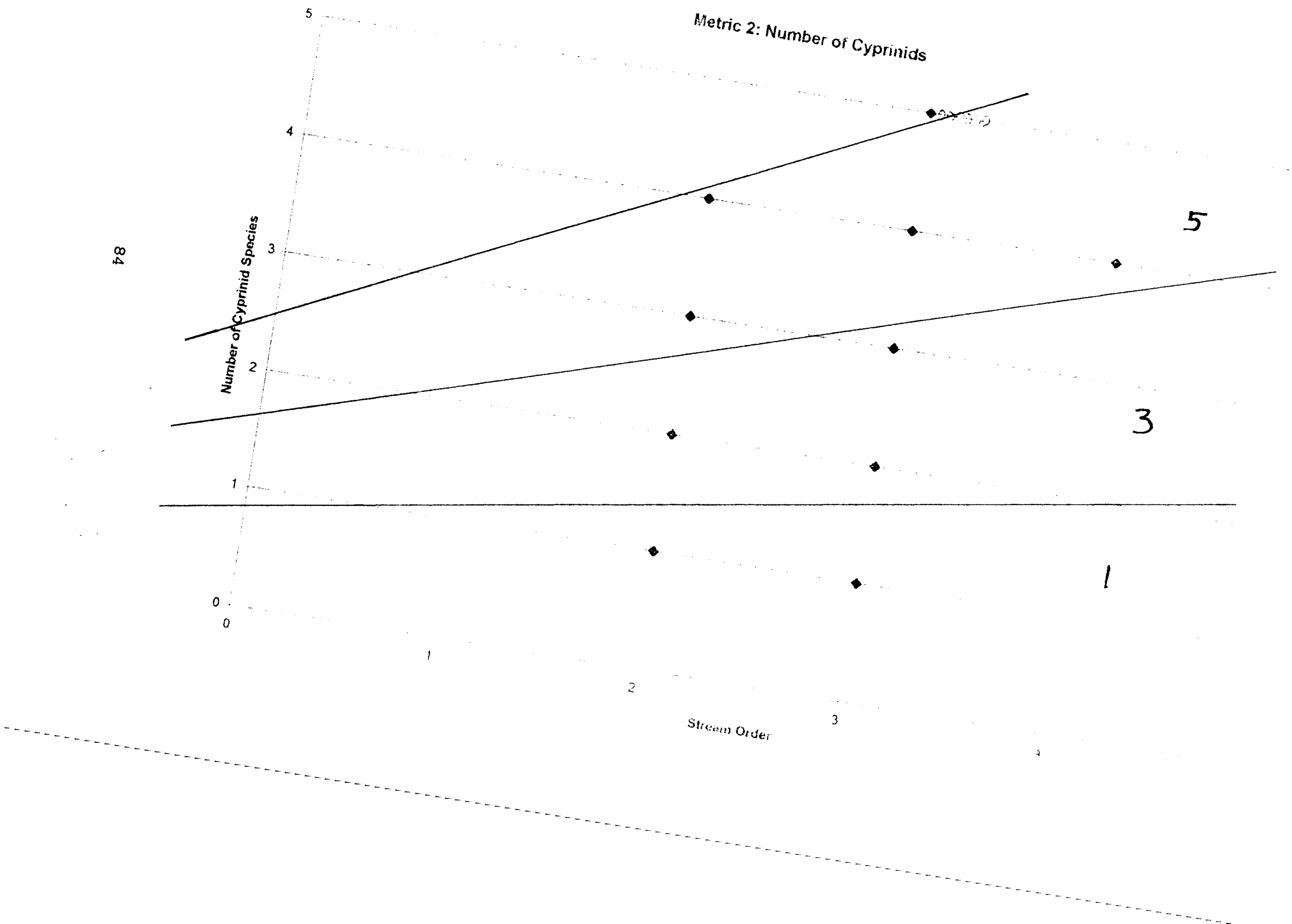
Metric #		Brushy	Harris	Moore	Caney
1	# Dominants in Common	4	3	4	
2	# Common Taxa	27	25	29	
3	Similarity Index	64	64.2	71.5	
4	Taxa Richness % Diff.	5.5	8.2	14	
5	Indicator Assemb. Index	0.667	1.074	0.758	
6	Missing Genera	0	0	0	
7	Funct. Group % Simil.	76	92	79	
	Total Number Taxa	35	34	43	37
	No. individuals/minute	17	28	24	22
	% EPT*	50	62	60	57
	% CA*	7	3	7	3

* EPT - Ephemeroptera, Plecoptera, Tricoptera
CA - Chironomidae, Annelida

APPENDIX D

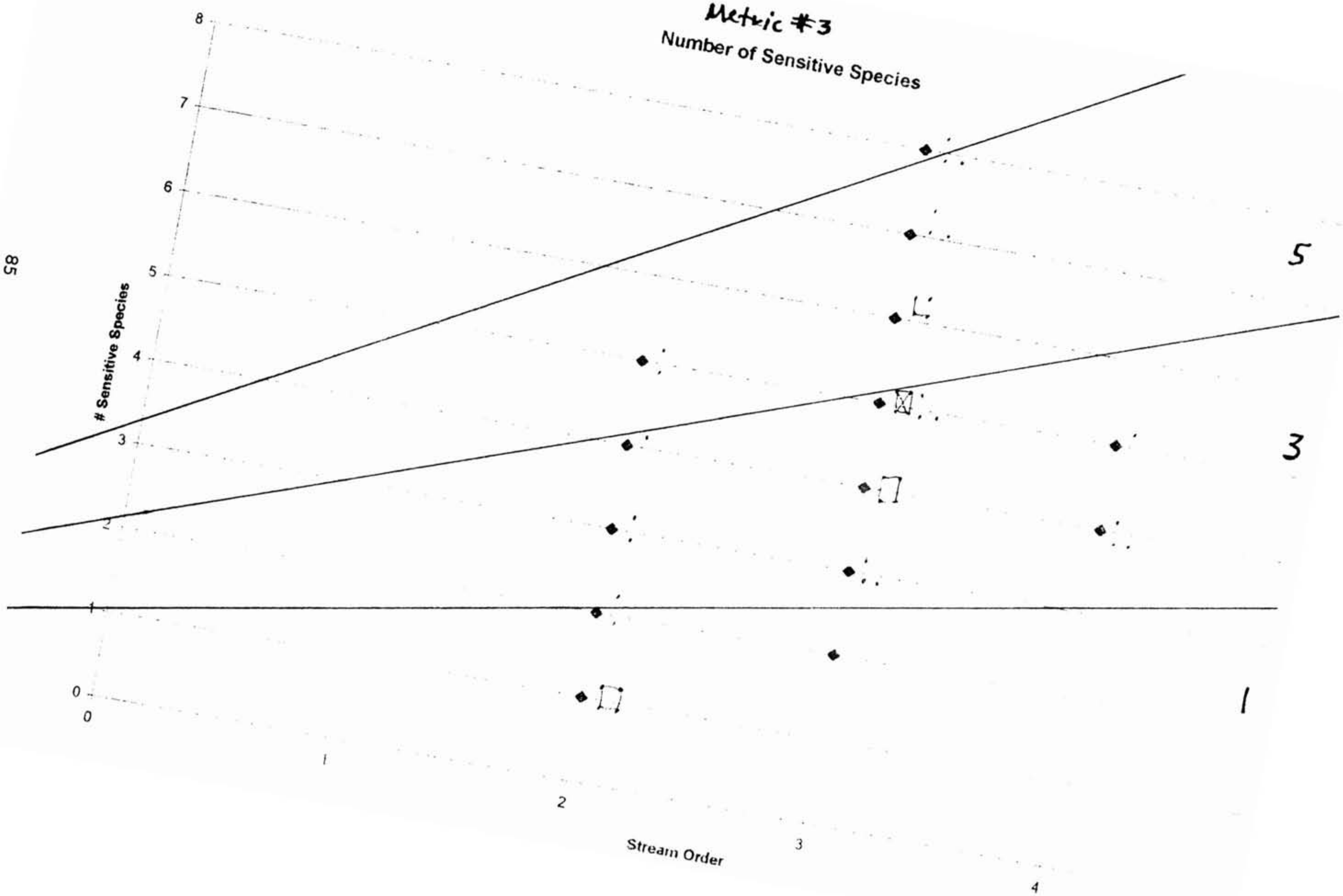
SCORING CRITERIA FOR IBI METRICS


Metric 1: Species Richness
Number of Species
Stream Order
0
2
4
6
8
10
12
14
16
18
0
0.5
1
1.5
2
2.5
3
3.5
4
4.5
5
3
1

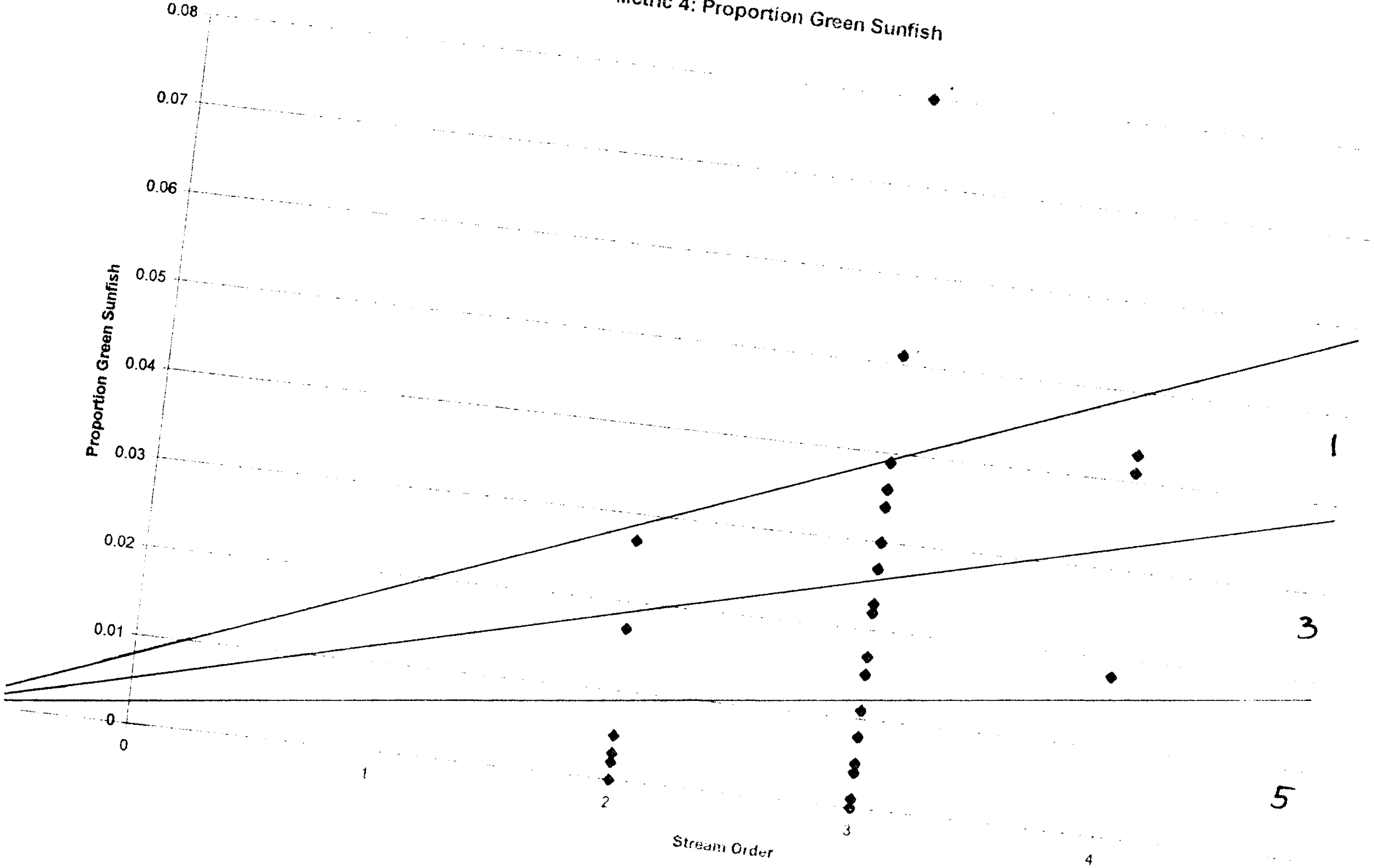

Metric 2: Number of Cyprinids
Number of Cyprinid Species
Stream Order
0
1
2
3
4
5
5
3
1

Metric #3

Number of Sensitive Species

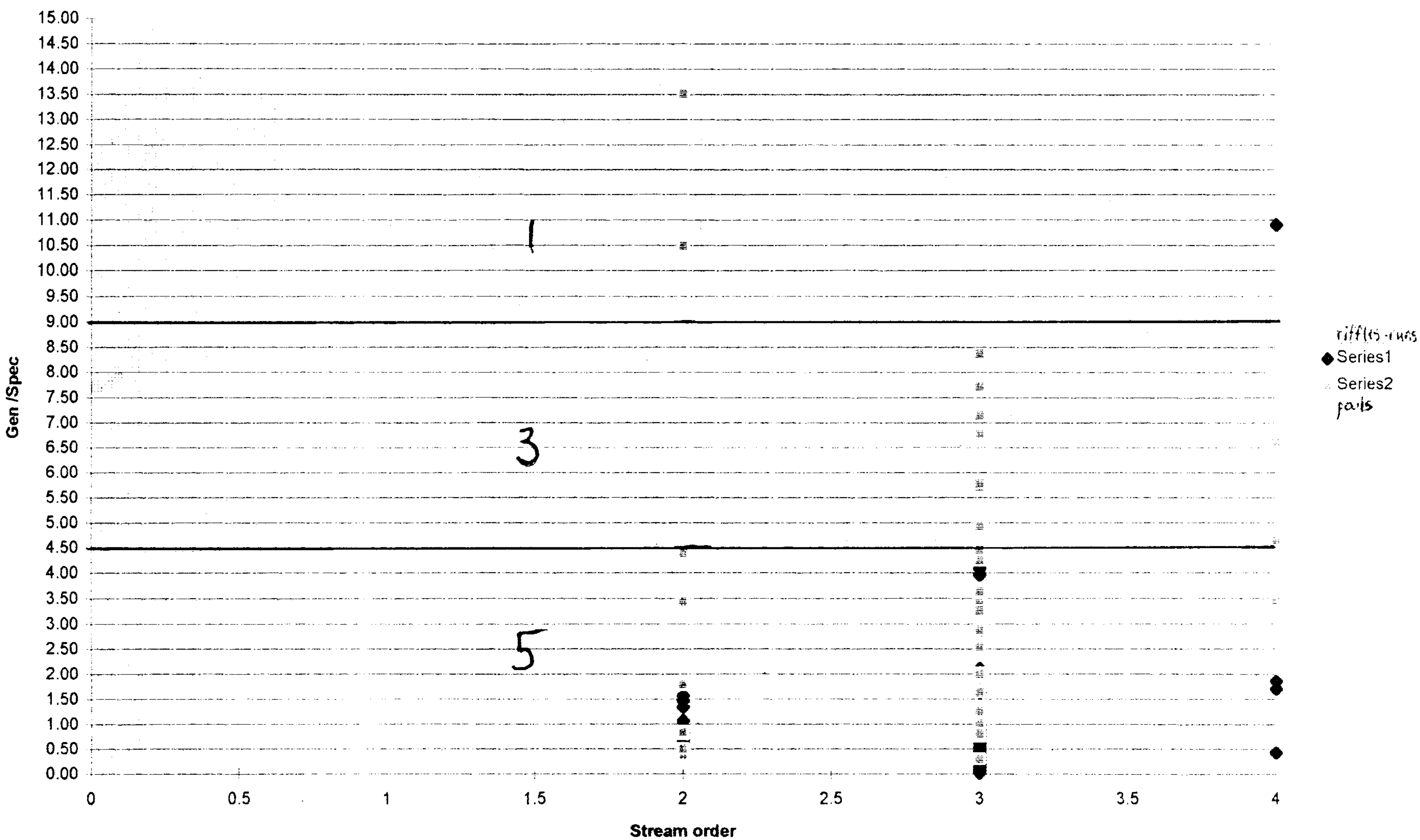


Metric 4: Proportion Green Sunfish

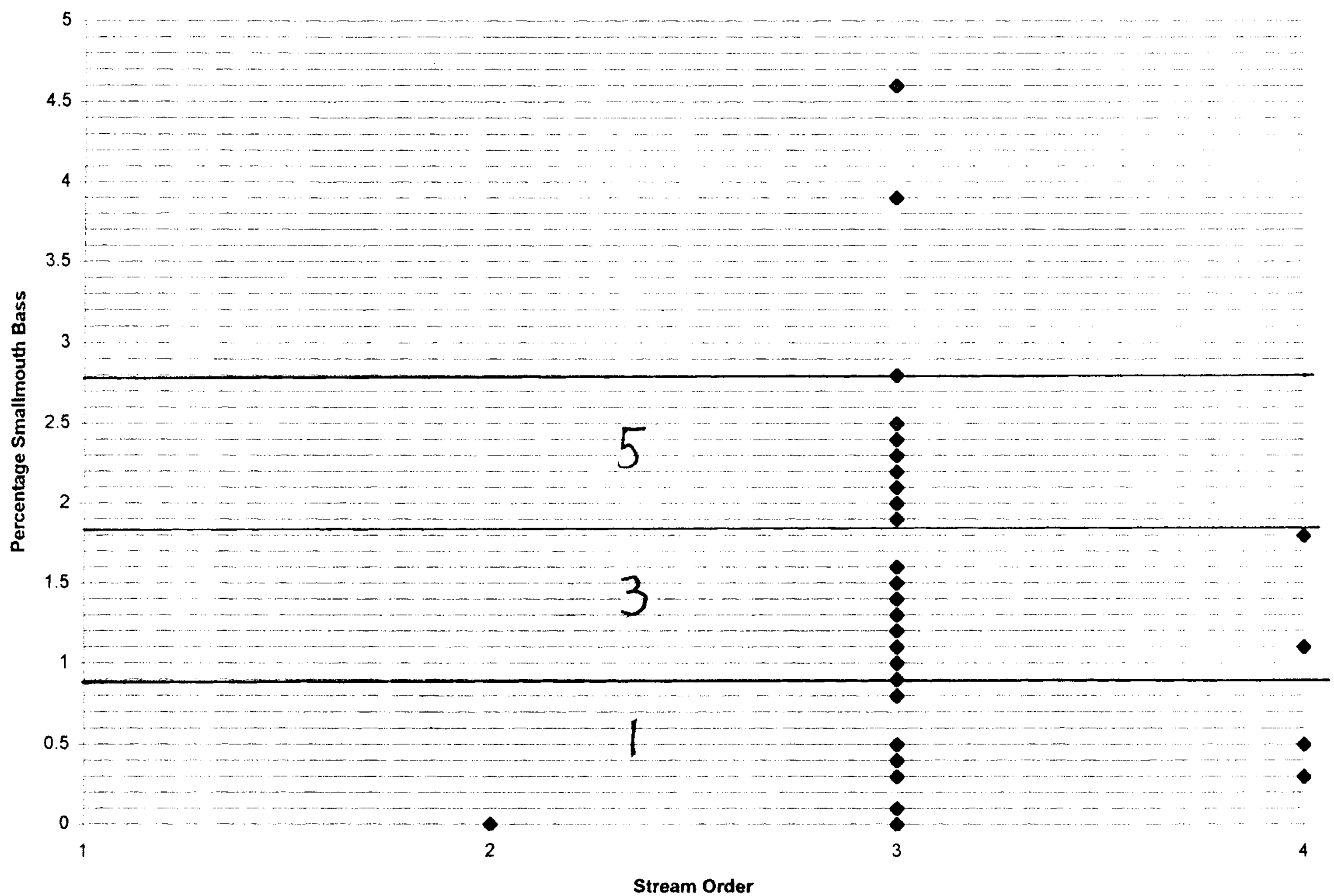


Metric #5

Generalists/Specialists



Metric 6: Percentage Top Carnivores



VITA

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