CAN RIPARIAN FORESTS MEDIATE IMPACTS OF URBANIZATION ON STREAM FISH ASSEMBLAGES?

by

ALLISON HUNT ROY

(Under the Direction of Mary C. Freeman)

ABSTRACT

Streamside riparian forests have been used to minimize degradation of water quality and biotic integrity in watersheds impacted by agricultural and silvicultural activities; however, little is known about the function of riparian forests in urban and suburban areas. This study tested the importance of riparian areas at (a) the local, reach scale; (b) the catchment scale; and (c) given varying levels of hydrologic and sediment disturbance in urbanizing streams in the Etowah River basin, Georgia, USA. In a study with paired open and forest canopies, riparian forests were not necessary for maintaining reach-scale habitat quality or biotic integrity (as measured by aquatic macroinvertebrates, salamanders, and fishes). Open reaches exhibited higher amounts of algae and herbivorous fishes, suggesting that openings in riparian canopies may provide patches of increased productivity leading to shifts in the trophic basis of production. At the catchment scale and based on data from 66 streams, fish species expected to be sensitive to disturbance were reduced in streams with lower % riparian forests along the entire upstream network. Catchment-wide development acted as a filter on fish assemblage integrity, with losses of sensitive species occurring at levels of riparian deforestation exceeding *ca.* 30 %. Local scale riparian deforestation secondarily affected tolerant species, which consistently had high abundances in streams with < *ca.* 60 % local riparian forest cover. I then examined relations between hydrologic alteration and fish assemblages, and asked whether the importance of riparian forests was contingent on levels of sediment and hydrologic disturbance. Increased frequency, magnitude, and duration of storm events and prolonged baseflow duration explained 20-66 % of the variation in fish assemblage richness and abundance, and these hydrologic disturbance variables were associated with increased percent impervious surface cover. Hydrologic alteration overwhelmed any influence of riparian forests on stream biota. However, for sensitive fish species, % riparian forest cover was predictive of richness and abundance if an interaction effect between riparian forests and level of sediment disturbance was included. Watershed management strategies must, at a minimum, simultaneously address hydrologic, sediment, and riparian disturbance in order to protect fish assemblage integrity.

INDEX WORDS: Riparian buffers, Urban, Land use/land cover, Impervious surface cover, Stream, Fish, Biotic integrity, Hydrologic alteration, Sedimentation, Stormwater, Etowah River, Habitat Conservation Plan

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CHAPTER 1

INTRODUCTION AND LITERATURE REVIEW

Urban development is an increasing and dominant disturbance to natural ecosystems worldwide. Humans have transformed one-third to one-half of the land area to maximize goods and services provided by the land (Vitousek et al. 1997). Approximately half the world population now lives in cities (vs 12% in 1900), thus concentrating impacts on a smaller land area, but necessitating reliance on resources from outside cities (Cohen 2003). There is a growing interest in understanding urban environments, in terms of the impacts of urban development on native ecosystems (ecology *in* cities), and the dynamics of urban environments as ecosystems (ecology *of* cities; Grimm et al. 2000). In fact, two National Science Foundation U. S. Long Term Ecological Research sites, Baltimore Ecosystem Study and Central Arizona-Phoenix, have been established to study the social, economic, and ecological elements of human-dominated systems. Although scientists have formed and tested many predictions about urban environments, the ecology of urban systems is largely in its infancy. Ultimately this research will be critical for determining how natural communities and ecosystem processes can persist in human-dominated environments.

Effects of urbanization on stream fishes

Urban development and associated increases in impervious surface cover can drastically alter geomorphology, hydrology, water quality, and function of stream ecosystems, resulting in subsequent biotic impairment (see reviews by Schueler 1994, Paul and Meyer 2001). Comparisons among streams along gradients of urban land cover suggest that urbanization can alter richness, diversity, density and biotic integrity of fish assemblages (Steedman 1988, May et al. 1997, Wang et al. 1997 & 2001, Snyder et al. 2003). In the southeastern United States, homogenization of fish assemblages, as indicated by changes in the ratio of highland endemic to widespread cosmopolitan species, has been documented with increased urbanization (Scott and Helfman 2001, Walters et al. 2003a). Landscape development has also been associated with diet shifts of generalist fishes with foraging flexibility (Weaver and Garman 1994; M.J. Paul, Howard University, *pers. comm.*). These community- and species-level changes in fish assemblages appear to occur at relatively low levels of urbanization (e.g. 10-15% impervious surface cover; Schueler 1994, Wang et al. 2001, Miltner et al. 2004).

The importance of catchment land use (e.g. % urban and % agriculture) for water quality and biotic integrity has become widely accepted in the literature in the last decade (Richards et al. 1996, Roth et al. 1996, Allan and Johnson 1997, Allan et al. 1997). However, we know little about whether location of developments and protected areas within a catchment can effectively minimize the detrimental effects on stream ecosystems within a human-dominated landscape. For example, areas of land adjacent to streams may have a disproportional influence on stream ecosystems relative to upland counterparts. If locations of development relative to stream channels are important, planners can locate developments in areas of the landscape to minimize impacts on stream ecosystems.

Riparian forests as potential mitigating tools

Over the past two decades research has highlighted the importance of undisturbed riparian zones, or land adjacent to flowing water, as critical areas of the landscape for influencing stream ecosystem processes and stream biotic integrity (Correll 2000). These riparian areas have been labeled “buffers” because they function to take up or retain nutrients (Peterjohn and Correll

1984, Osborne and Kovacic 1993), sediments (Davies and Nelson 1994, Waters 1995), metals, and other contaminants (also see reviews Lowrance 1988, Gregory et al. 1991, Sweeney 1992). For streams that have evolved in forested landscapes, forested riparian areas also provide bank and geomorphic stability, large woody debris and other organic matter inputs (Trotter 1990), as well as terrestrial arthropod inputs (Mason and MacDonald 1982, Nakano et al. 1999). Removal of forested riparian zones decreases shading, increases temperature (Brazier and Brown 1973), and increases primary productivity (Murphy et al. 1981, Noel et al. 1986), which influences biotic communities by favoring certain species and altering population dynamics (Barton et al. 1985, Feminella et al. 1989). Many studies have demonstrated how structural habitat provided by riparian buffers indirectly affects higher trophic levels such as macroinvertebrates (Benke et al.1988, Whiles et al. 2000) and fishes (Karr and Schlosser 1978). Essentially, forested riparian areas supply streams with high diversity and quality of instream habitat for aquatic biota (Culp and Davies 1983, Frissell et al. 1986, Richards and Host 1994).

Because they have the potential to provide numerous services to society at relatively low costs, riparian buffers have been recognized as a “conservation bargain” by many state and local governments (Wenger 1999). In the U.S., small streams (< 50 km^2) have mean buffer width requirements (mandatory or recommended by best management practices) of 19.9 m (Lee et al. 2004). In Georgia, the Erosion and Sedimentation Control Act requires primary and secondary trout streams to maintain an undisturbed 50 ft (~ 15 m) riparian buffer, and all other streams to maintain 25 ft (~ 8 m) buffers (O.C.G.A. 12-7). In the last decade, many local governments have adopted more stringent riparian buffer protection programs (England and Roy 2001). Although there is an overwhelming acceptance of the importance of riparian buffers for protecting streams, local forested riparian areas are limited in their ability to influence upstream and downstream disturbances (Allan et al. 1997). Furthermore, while riparian buffers have been shown to protect water quality and biotic integrity in watersheds with agricultural land uses (Peterjohn and Correll 1984, Stevens and Cummins 1999), little is known about the function of riparian buffers in urban and suburban areas (Wenger 1999).

Etowah River basin history & biodiversity

This study took place in the Etowah River basin, a 4823 km^2 catchment in north-central Georgia, USA that is one of the most biologically diverse river systems in the United States. The Etowah River system boasts 76 extant native fish species, including 4 locally endemic species (Burkhead et al. 1997). Of these, 7 fish species are state-listed as imperiled by the Georgia Wildlife Protection Act and 3 fish species are federally protected under the U.S. Endangered Species Act, including the Cherokee darter, *Etheostoma scotti*, which is commonly found in small streams where our sampling took place.

Recent declines of fishes and other aquatic biota in the Etowah River basin have been attributed to changes in land cover in the region. For most of the 20th century, the region was dominated by forest (secondary growth) and agricultural (row-crop and pasture) land uses; however, the area has undergone rapid changes in land cover in the last two decades. Over the course of the 1990s, the city of Atlanta added more people than any other region in the U.S. except Los Angeles, and Forsyth County (part of Etowah basin) was ranked as the fastest growing county in the nation in 1998 (U.S. Census Bureau 1999). Thus, the sub-catchments within the Etowah River basin exhibit a range in urban, agriculture, and forest land cover. The rate of habitat loss related to upland development is such that species extirpations and even extinctions appear possible in coming decades, if growth is not managed properly.

Two of the primary stressors to stream ecosystem health in the Etowah River basin are sedimentation and hydrologic alteration associated with past agricultural practices and recent urban development in the basin. Previous research in the Etowah River basin linking land cover, geomorphology, chemistry, and biotic integrity has shown relations between recent percent urban land cover within the catchment and impaired biotic assemblages coincident with geomorphic alteration (primarily bed sediment) and decreased water quality (Roy et al. 2003, Walters et al. 2003a,b). Sedimentation is known to be a problem because it degrades the gravel and cobble habitat on which the imperiled fish species rely (Waters 1995). Some of this sediment may come from active construction sites, but a significant portion may also originate from the stream channel itself, possibly as a result of increased stormflows from poorly managed runoff. In addition, hydrologic alteration could impact fish assemblages by reducing baseflows and changing the natural hydrograph during storm events, potentially reducing habitat needed by fishes. Although hydrologic alteration is a primary mechanism of urbanization impacts on stream ecosystems (Booth and Jackson 1997), no published study has examined effects of altered hydrology on fish assemblages in small streams with exceptional native species diversity, as in the Etowah. Quantification of these stressors (e.g., sediment, hydrology), their relationships with impervious surface cover, and thresholds where fish assemblage health declines is necessary to answer critical questions about effectiveness of forested riparian areas in urbanizing streams.

Etowah Regional Aquatic Habitat Conservation Plan

In 2002, the U.S. Fish and Wildlife Service awarded a grant to an interdisciplinary team of scientists, lawyers, educators and policy analysts from the University of Georgia, Kennesaw State University, and the Georgia Conservancy to develop a regional aquatic habitat conservation plan (HCP) for the imperiled aquatic species of the Etowah River basin. Under the U.S.

Endangered Species Act, it is illegal to “take” (i.e. kill, harm, capture, otherwise hurt, or significantly modify or degrade their habitat) any species listed as endangered or threatened. Developers and local governments who propose projects that may result in any incidental “take” of listed animals can be required to develop individual HCPs to protect species (Moser 2000). Individual HCP development, consultation, and approval is a costly and time-consuming process which may not effectively protect imperiled species from continued growth and development. The idea behind a *regional* HCP is that local governments would adopt policies and amend zoning codes and development regulations so that any new activities would have minimal impact on imperiled species (McKinney and Murphy 1996). For example, counties and city jurisdictions that fall within the basin might adopt stricter stormwater regulations, erosion and sedimentation control ordinances, or impervious surface limits as part of the regional HCP. All local governments that sign on to the HCP would hold incidental take permits and oversee all development in their jurisdictions. (See web page for more information: http://www.etowahhcp.org/)

In order to ensure that local governments adopt policies and management guidelines that will adequately protect the imperiled fish species in the Etowah River basin, research has been developed to identify critical stressors and mechanisms of impact on these fish species. To that end, projects in this dissertation are aimed at a) investigating the impacts of hydrologic alteration on stream fishes, and b) determining the importance of riparian forests at mitigating impacts of urbanization on stream fishes. Results from these projects will be directly applied to inform local ordinances and policies within the HCP. The challenge for scientists, however, is not only to identify important stressors to the system, but also to ensure that regulations are sufficient to protect imperiled species. This process will necessarily include continued assessment of the effectiveness of regulations and adaptive management to adjust for new findings (Wilhere 2002).

Research objectives

My dissertation research involves a series of four projects evaluating the ability of riparian forests to mediate biotic impacts of urbanization on stream fish assemblages. I anticipate that riparian buffers are less important in urban settings because (1) buffers are often by-passed, and (2) the impacts of catchment urban land use overwhelm capacity of riparian zones to buffer the stream ecosystems. Discontinuous riparian buffers may not solve sediment and water quality problems, because transport through gaps dominates landscape discharge (Forman 1997, Scarsbrook and Halliday 1999, Weller et al.1998). Thus, I expect that riparian buffers will have limited benefits in urban areas where storm runoff is often diverted into conveyances and passes directly into the streams.

For this dissertation, I asked a) to what extent are local- and regional-scale riparian forests protective of stream fishes in urbanizing areas, and b) if the function of riparian forest is dependent on the extent of other stressors such as sediment and hydrology. Chapter 2 compares adjacent open and forested stream reaches in urban settings (15-50 % urban land cover) to identify local, reach-scale physical and biotic differences related to riparian deforestation. The idea is to ask whether, all other things equal, the presence of riparian forests influences reach-scale stream quality. In Chapter 3, I examine relationships between landscape scale (vs. reach scale) riparian condition and fish assemblages using a large, existing database of fish assemblages. The intent of this chapter is to determine the extent of riparian forests necessary to maintain healthy fish assemblages. Hydrologic alteration was expected to be an important stressor on fishes in the Etowah River basin; however, responses of small stream fishes to altered stormflows and baseflows have not been empirically tested. Thus, the fourth chapter investigates relations between hydrologic alteration and fish assemblages across 30 streams with a range in percent impervious surface cover. That project also sets the groundwork to ask whether the importance of riparian forests is contingent on levels of sediment and hydrologic disturbance (Chapter 5). Altogether, these projects will examine the effectiveness of forested riparian areas as a tool for protecting stream habitat and fish assemblage integrity in urbanizing catchments, and the conditions necessary to maintain healthy stream fauna. Results will be used to inform policies within the Etowah Regional Aquatic HCP, determining the necessary components of effective riparian buffer ordinances, and other concurrent policies that must be in place for riparian forests to be effective at mitigating impacts of urban disturbance on stream fish assemblages.

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CHAPTER 2

REACH-SCALE EFFECTS OF RIPARIAN FOREST COVER ON URBAN STREAM ECOSYSTEMS[1]

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Abstract

Forested riparian areas are critical features of the landscape for nutrient and sediment retention and for habitat protection in streams impacted by agriculture and forestry practices. However, little is known about the effectiveness of forested riparian zones for maintaining stream quality and function in urbanizing areas. We compared paired open and forested reaches within 5 small streams (10-20 km^2) in suburban catchments (9-49 % urban) in the Piedmont of Georgia. Stream reaches with open canopies were narrower than forested reaches (4.1 vs. 5.0 m, respectively), supporting results of previously published studies. Interestingly, there were no differences in habitat diversity (variation in velocity, depth, or bed particle size) between open and forested reaches, suggesting that habitat quality is driven by catchment factors in these urbanizing streams. However, absence of riparian forest cover at the reach scale corresponded with decreased large woody debris and increased algal chlorophyll *a* standing crop biomass. These differences in basal food resources translated into higher densities of fishes in open (9.0 ind. m^{-2}) vs. forested (4.9 ind. m^{-2}) reaches, primarily attributed to higher densities of the herbivorous fish, *Campostoma oligolepis*. Reach-scale openings in riparian canopies may provide local patches of increased primary productivity and shifts in the trophic basis of production. Densities of terrestrial invertebrate inputs were higher in open reaches; however, trends suggested higher biomass of terrestrial inputs in forested reaches, and a corresponding higher density of terrestrial prey consumed by water-column feeding fishes. Reach-scale biotic integrity (macroinvertebrates, salamanders, fishes) was largely unaffected by differences in canopy cover, except where high densities of *Campostoma* dominated differences in tolerant and habitat generalist species. Overall, our streams had relatively poor biotic assemblages, reflecting influences of urban disturbance. In urbanizing areas where catchment land cover drives habitat

and biotic quality, management practices that rely exclusively on forested riparian areas for stream protection are unlikely to be effective at maintaining ecosystem integrity.

Introduction

Over the past two decades, research has highlighted the importance of streamside riparian zones as critical areas of the landscape for influencing stream ecosystem processes and stream biotic integrity (Correll 2000). These riparian areas have been labeled "buffers" because they function to take up or retain nutrients (Peterjohn and Correll 1984, Osborne and Kovacic 1993), sediments (Davies and Nelson 1994, Waters 1995), metals, and other contaminants (also see reviews Lowrance 1988, Gregory et al. 1991, Sweeney 1992). Forested riparian areas also provide bank and geomorphic stability, large woody debris and other organic matter inputs (Trotter 1990), and terrestrial arthropod inputs (Mason and MacDonald 1982, Nakano et al. 1999). For streams that have evolved in forested landscapes, removal of forested riparian zones decreases shading, increases temperature (Brazier and Brown 1973), and increases primary productivity (Murphy et al. 1981, Noel et al. 1986), which influences the biotic communities by favoring certain species and altering population dynamics (Barton et al. 1985, Feminella et al. 1989). Many studies have demonstrated how structural habitat provided by riparian buffers indirectly affects higher trophic levels such as macroinvertbrates (Benke et al.1988, Whiles et al. 2000) and fishes (Karr and Schlosser 1978). Essentially, forested riparian areas supply streams with high diversity and quality of instream habitat for aquatic biota (Culp and Davies 1983, Frissell et al. 1986, Richards and Host 1994).

Although intact riparian areas have been shown to minimize degradation of water quality and biotic integrity, this research has primarily been conducted in watersheds with agricultural (e.g. Peterjohn and Correll 1984, Osborne and Kovacic 1993) and silvicultural (e.g. Davies and Nelson 1994, Stevens and Cummins 1999) land uses. Very little is known about the function of riparian buffers in urban and suburban areas (but see Hession et al. 2002). We anticipate that riparian buffers are less important in urban systems because (1) buffers are often by-passed by stormwater conveyances, and (2) the impacts of urban land cover in the catchment are overwhelming. However, at the reach-scale, forested riparian areas may still provide essential functions of shading, bank stability, and organic matter inputs for streams.

This study directly addresses the role of reach-scale riparian forests in urbanizing streams. Based on paired open and forest reaches within five urban streams, we compared water quality, physical habitat, food resources (e.g. algae), benthic and terrestrial invertebrates, salamanders, and fishes between reaches. Figure 2.1 illustrates the expected changes associated with reach-scale riparian deforestation. We expect that riparian deforestation will increase mean annual temperature and temperature variability (Sweeney 1992), resulting in loss of fish species sensitive to higher temperatures and subsequent decreases in fish assemblage integrity. Riparian deforestation is hypothesized to decrease habitat diversity (e.g. variation in depth, velocity, bed sediment size; Jones et al. 1999), which, in turn, will decrease stream biotic integrity (benthic macroinvertebrates, salamanders, fishes). Stream channels without forested canopies are also expected to be narrower than those with forested canopies, as found in previous studies (Sweeney 1992, Davies-Colley 1997, Scarsbrook and Halliday 1999). We expect lower organic matter (e.g. large woody debris, organic matter inputs) in open reaches, which would decrease the density of shredding invertebrates dependent on leaf material (Wallace et al. 1999). Increased light associated with the lack of canopy cover in open reaches is expected to increase primary productivity (e.g. algae, chlorophyll *a*; McIntyre and Phinney 1965) in the stream, and, subsequently, increase densities of herbivorous invertebrates (e.g. scrapers) and fishes. Streams with lower canopy cover are also hypothesized to have lower terrestrial invertebrate inputs, as observed by Mason and MacDonald (1982), Edwards and Huryn (1996), and Kawaguchi and Nakano (2001). The relative changes in benthic and terrestrial invertebrates associated with riparian deforestation are expected to translate into diet shifts of insectivorous salamanders and fishes (Fig. 2.1). For this study, we tested these predicted differences between open and forested reaches to determine the function of reach-scale riparian forests in urban streams.

Methods

Study sites

This study took place in small streams within the Etowah River basin, a 4823 km^2 catchment in north-central Georgia. The region was originally dominated by deciduous forests; however, human practices in the last century have transformed the landscape to a mosaic of secondary forest, agriculture, and urban uses. In the last two decades, suburban development spreading from metropolitan Atlanta (population > 4 million) has led to concerns of habitat degradation and species loss within the watershed (Burkhead et al. 1997). Specifically, small streams (10-20 km^2, approximately 2nd-3rd order) within the basin exhibited an average 13% decrease in forest cover and 11% increase in urban land cover between 1973 and 1997 (Roy et al. 2003a). Although forest land cover is higher in 100 m riparian areas alongside streams relative to the entire catchment, riparian areas have exhibited similar trends of decreased forest and increased urban land cover in the last 3 decades (Roy et al. 2003a).

Five streams with paired sites in open and forested reaches (for a total of 10 sites) in the Piedmont physiographic region of the Etowah River basin were selected for the study (Table

2.1). Streams were selected to be of similar size (drainage area from 10-20 km^2) and similar slope. We selected streams that had > 15 % urban land cover (i.e. above level expected to detect effects of urban disturbance on biota, Schueler 1994) and < 30 % agriculture land cover based on 1997 *Landsat* Thematic Mapper (TM) satellite imagery. Open reaches constituted of at least 200 m of stream that had an open canopy due to lack of forest on at least one side of the stream. Forested reaches had at least 30 m of forest on each side of the stream for the entire 200 m reach. Less than 1 km of stream separated open and forested reaches and no significant tributaries that would drastically change drainage area or upstream land cover entered between the reaches.

Landscape assessment.

Subsequent to site selection, we recalculated land cover for each catchment with 2001 *Landsat* TM satellite imagery because this was the most recent and applicable coverage available. We characterized land cover for the sites based on percent urban (low and high density), agriculture (cultivated/exposed land, cropland/grassland), forest (evergreen, deciduous, mixed), golf courses, and open water land cover. Percent land use/cover categories were determined for the entire upstream catchment and within a 30 m buffer for the upstream drainage network. We also determined percent impervious cover for each sub-catchment from a classified dataset created by the Georgia Land Use Trends Project (GLUT, Natural Resources Spatial Analysis Laboratory, Institute of Ecology, University of Georgia, Athens, GA, USA). Department of Transportation road data (1999) were used to calculate number of road crossings per stream length for each site. 1999 U.S. Geological Survey NAPP ortho-corrected air photos were used to count the number of impoundments per stream length.

At 10 random transects (corresponding to terrestrial invertebrate input transects), we measured distance from stream bank and diameter at breast height (DBH) for all trees within a 5 m wide corridor extending 30 m from the stream bank to determine tree density and mean tree diameter. We also converted measurements to basal area (ft^2) per acre to compare tree coverage to Georgia's Best Management Practices (BMPs) for Forestry (GA EPD 1999).

Water chemistry

Stream water chemistry was sampled 5 times from November 2001-2002. Specific conductance, pH, and dissolved oxygen were measured with a portable Hydrolab Datasonde© 4 (Hydrolab Corporation, Austin, TX, USA), which was calibrated before each use. Turbidity was measured in the field with a Hach© Turbidimeter (HACH Company, Loveland, CO, USA). Grab water samples were filtered (Gelman® glass fiber, 0.45 μm, Gelman Sciences, Ann Arbor, MI, USA) into acid-washed bottles, placed on ice, and transported to the laboratory for ammonium-nitrogen (NH_4-N), nitrate/nitrite-nitrogen (NO_3/NO_2-N) and soluble reactive phosphorus (SRP) analysis. Dissolved nutrients were analyzed using an Alpkem autoanalyzer (Alpkem Corporation, Wilsonville, OR, USA) by the University of Georgia Institute of Ecology Chemical Analysis Laboratory following Standard Methods protocol (American Public Health Association 1989). We also calculated the molar ratio of dissolved N:P as DIN (NH_4-N + NO_2/NO_3-N) to SRP. An additional unfiltered grab sample was brought back to the lab, filtered, dried, and weighed to determine total suspended sediments. Hobo© temperature data loggers were placed at each site in November 2001 and stream water temperature was recorded every 15 minutes for one year.

Physical habitat

Stream slope was measured at tops of riffles with a TopCon© survey scope and stadia rod for the 200 m reach. Percent riffle, pool, and run habitat were calculated based on habitat units designated in the fish survey. A visual estimate of % fines and % embeddedness was made in random locations within each habitat unit using a view box. Depth, velocity, and modal particle size (*phi*) were measured at 34 random distances (68 for centerline) in the 200-m reach along 5 longitudinal transects (10, 30, 50, 70, 90 percentile) for a total of 204 measurements. These variables were used to calculate the Shannon Index (H', Zar 1974) as a measure of habitat diversity for single variables and all combinations of depth, velocity, and particle size (following Jones et al. 1999). A measure of evenness (J'; relative diversity) was also calculated by dividing Shannon's diversity (H') by the log of the number of categories (Zar 1974). Light (% canopy cover) was measured at the stream centerline at 10 locations along each reach in July 2002 (corresponding to terrestrial invertebrate transects) and September-October 2003 (corresponding to algae transects) using a spherical densiometer. A quantitative estimate of percent fines (< 2 mm) in riffle habitats was obtained by taking three 1-L grab samples of the bed sediment and dry sieving and weighing the material in the laboratory.

Organic inputs

The amount of wetted large woody debris was measured for each 200 m sample reach. Diameter and length were measured for all submerged wood ≥ 10 cm in diameter and ≥ 1 m long, and characteristics pertaining to the stability of the wood (angle, attached, pool-forming) were recorded.

Algae

Biofilm and algae were sampled September-October 2003 during leaf-out at 10 equally-spaced transects along the 200-m reach. Three samples from the dominant substrate (rock or sand) at the transect were combined in a whirl-pack and brought back to the lab on ice. A 60-mL syringe was modified into a Loeb sampler by replacing the end of the plunger with a firm toothbrush and adding a neoprene seal to the base to sample biofilm from rocks (Loeb 1981).

Rocks were scrubbed with the sampler and water was removed and placed into the bag. For sandy habitats, the tip of a 60-mL plastic syringe was removed to create a sediment core, and the top ~10 cm of sand was removed using the core and a spatula. In the laboratory, sand samples were sonicated for 1 min. All samples were stirred for ~1 min. and subsamples of a known volume of slurry were filtered onto Whatman® GF/F filters (Whatman Inc., Clifton, NJ, USA) for chlorophyll *a* and biofilm biomass. Biofilm biomass samples were dried, ashed, and weighed to determine biofilm ash free dry mass (AFDM). Chlorophyll *a* was extracted from frozen filters by first cutting and grinding in 90% acetone to break apart any blue-green cells (adapted from Wetzel and Likens 1991). Samples were decanted into a centrifuge tube and placed in a refrigerator for ~24 hours. Samples were then centrifuged for ~5 min, subsampled (if necessary), and analyzed for chlorophyll *a* with a fluorometer (TD-700, Turner Designs, Sunnyvale, CA).

Invertebrates

Benthic invertebrates were sampled in 5 replicate riffle habitats (1 sample per riffle) at each of the 10 study reaches in 8-15 April 2002. Invertebrates were collected using a 0.09-m^2 Surber sampler (500-µm mesh) and scrubbing rocks to a depth of 10 cm for three minutes. Samples were elutriated in the field, separated into small (0.5-1 mm) and large (>1 mm) size fractions, and preserved in 10% formalin. In the laboratory, the small fraction was subsampled to 100 invertebrates using a wheel sample splitter when necessary. All invertebrates in the large fraction and the sub-sampled small fraction were counted and identified to genus except for Chironomidae (classified as predatory Tanypodinae or collector-gatherers) and non-insects (identified to order) using standard keys (Merritt and Cummins 1996). Richness, density, and other macroinvertebrate composition measures (Ephemeroptera, Plecoptera, and Trichoptera (EPT) richness; % in functional feeding groups; % chironomids) were calculated.

Terrestrial invertebrate inputs into the stream were sampled 8-16 July 2002. Invertebrates were collected by placing 10 floating, clear containers (surface area = 0.344 m^2 each) filled with water and unscented soap (surfactant) at random transects along the stream reach (based on methods similar to Mason and MacDonald 1982). Invertebrates were removed after 24 hours and preserved in formalin. In the lab, invertebrates were identified to order, counted, and dried to determine biomass. Diversity was calculated using the Shannon-Weiner diversity index (H').

Salamanders

Stream reaches were surveyed for stream salamanders at the three streams with riffle habitat in both open and forested reaches (Lawrence, Picketts Mill, and Clark Creeks) on 30-31 March 2003. We used a backpack electroshocker (Model 12-B; Smith-Root®, Inc., Vancouver, WA) in leaf packs and shallow riffle areas while kicking rocks and organic debris and holding a 500-µm mesh kick net downstream. We searched for equal amounts of time in open and forested reaches within each stream and recorded shock time. Salamanders were euthanized using an overdose of MS-222, preserved in 10% formalin, and brought back to the laboratory for identification and gut analysis. Area (m^2) of riffle habitat sampled for salamanders was measured to estimate salamander densities.

In the laboratory, salamanders were transferred to 70% ethanol. Each individual was counted and measured for snout-vent length (SVL). Salamander stomach contents were analyzed for all individuals in Lawrence Creek and Picketts Mill Creek, and a randomly selected subset of 12 individuals within each sample at Clark Creek. We removed the stomachs and intestines of individuals under a dissecting microscope (15x). Gut contents were identified to lowest possible taxonomic unit (typically order or family) and counted. Salamander abundance, catch per unit effort (CPUE), and diversity of prey items (Shannon-Weiner, H') were calculated for each reach.

Fishes

In fall 2001, fish assemblages were sampled using a backpack electroshocker (Model 12-B; Smith-Root®, Inc., Vancouver, WA), dip nets, and seines in the most efficient manner for the habitat present. Block nets were used to separate habitat units (riffle, pool, run) and each segment was fished for 3 passes. We sampled a minimum of 3 replicate riffle and pool habitats for a total stream length of up to 200 m. Fish were identified in the field, measured (standard length), and released. Individual fish that could not be identified (including young of the year) were euthanized using MS-222, preserved in 8% formalin, and brought back to the lab for identification.

Fish data were used to calculate richness, density, and relative abundance of various composition metrics. Measures of tolerant species, habitat generalists, highland endemic species, and cosmopolitan species were calculated to assess fish assemblage integrity. Tolerant species were widespread, ubiquitous species found in over 80% of small streams in the Etowah based on previous research (Walters et al. 2003) and are expected to be tolerant of multiple stressors. Habitat generalists are species that do not show preference for pool, pool-run, or riffle-run habitats, and were classified based on Etnier and Starnes (1993) and Mettee et al. (1996). Cosmopolitan species were defined as those fishes native to at least 10 major drainages, whereas endemic species were primarily limited to the Coosa River drainage (Walters et al. 2003). We also examined the ratio of endemics to cosmopolitans (based on species richness and abundance), which should reflect a homogenization of fish assemblages coincident with a loss of endemic species (Scott and Helfman 2001, Walters et al. 2003).

Individuals of two invertivores (*Percina nigrofasciata*, the blackbanded darter and *Cyprinella callistia*, the Alabama shiner) and two trophic generalists (*Lepomis auritus*, the redbreast sunfish and *Hypentelium etowanum*, the Alabama hogsucker) collected during sampling in fall 2001 were brought back to the lab for gut content analysis. *Notropis xaenocephalus* (Coosa shiner, an invertivore) was collected instead of *C. callistia* at one site where the latter species was not present. In the laboratory, all individuals were measured, weighed, and contents of the stomachs and intestines were removed from the fishes. We identified the gut contents of a maximum of 10 individuals; if more than 10 individuals were collected we selected individuals most similar in size between open and forested reaches at a stream. Gut contents were identified to the lowest possible taxonomic unit (typically order or family) and counted. Percent abundance food types (at the level of order) and % terrestrial (vs aquatic) food items were compared between open and forested reaches.

Data analysis

This study was designed to compare reaches with different amounts of riparian cover, with all other land cover factors being equal. Thus, paired *t*-tests were used to compare mean differences between open and forested reaches in water quality, physical habitat, organic matter, algae, invertebrate, salamander, and fish variables. Fish lengths were compared using two-sample *t*-tests (assuming unequal variances) for each species that had > 4 individuals in open and forested reaches. All percentage data were transformed using arcsin squareroot, and abundance data were transformed using log (x+1) or $x^{0.25}$, where necessary to reduce skewed distributions.

Results

Differences in environmental variables and basal food resources

Canopy cover at forested reaches (77.6%) was about twice that at open reaches (32.3%; Table 1.2). Land cover adjacent to open reaches included golf courses (2 sites), pasture (2 sites), and ball fields (1 site; Table 2.1). Streams varied in % urban land cover in catchment (8.8 to 49.3), % forest in 100 m riparian area (40.5 to 73.9), and density of road crossings (0.19 to 0.61 m^{-1} stream length, Table 2.1). Riparian vegetation differed between reaches, with higher densities of trees in forested reaches, although trees in open reaches tended to be larger in diameter than those in forested reaches. Overall, the basal area of trees per acre was higher in forested (2097 ± 922) vs open (469 ± 350) reaches (Table 2.2).

In general, water quality was similar between open and forested reaches, supporting our hypothesis that water quality is driven by catchment-scale parameters rather than reach-scale cover. One notable exception was dissolved oxygen, with higher concentrations in open (9.4 mg L^{-1}) vs. forested (8.9 mg L^{-1}) reaches. Additionally, NO_2/NO_3-N concentrations were higher in forested reaches vs. open reaches (Table 2.2). Differences in canopy cover between reaches also affected stream temperature, with higher summer daily minimum temperature and trends toward higher diel temperatures in open reaches (Table 2.2).

We measured numerous variables related to channel morphology and physical habitat at each site. There were no significant differences in riffle area, pool/run area, or % habitat types between open and forested reaches (Table 2.2). Although there were no differences in depth between reaches, our data indicate that streams were on average about 20% wider in forested vs. open reaches (5.0 m vs. 4.1 m). We hypothesized that riparian deforestation would alter in-stream habitat diversity; however, we found no evidence of differences in diversity (H') or evenness (J') of depth, velocity, or bed sediment size, or any combination of those variables (Appendix B.4).

As expected, the amount of large woody debris (LWD) was higher in forested reaches compared to open reaches (Table 2.2). Although the diameter of LWD was the same between reaches, pieces of LWD in open reaches were longer than those in forested reaches, and were more critical in forming pool habitats in open reaches. However, our data indicate that LWD in forested reaches is more stable, with a higher percent of pieces attached to the bank in forested (56.5%) vs. open (38.1%) reaches (Table 2.2).

Higher light associated with open canopies was expected to result in higher amounts of benthic algae. Although there was no significant difference in biofilm biomass between reaches, biomass on rock substrates were marginally higher in open vs. forested reaches. We found significantly higher concentrations of chlorophyll *a* in open (32.7 mg m^{-2}) vs. forested (13.7 mg m^{-2}) reaches (Table 2.3).

Differences in biotic assemblages

Benthic invertebrates in riffle habitats did not differ in total taxa richness, or richness of insect or EPT taxa (Table 2.4). In addition, there were no significant differences in benthic invertebrate densities between reaches, although most streams had higher insect densities in open vs. forested reaches (Table 2.4, Fig 2.2). The variation in densities among replicate samples (an indicator of patchiness) was significantly higher in open reaches compared to forested reaches. Reaches had similar % chironomids (36% and 33% in open and forested reaches, respectively). Scraper density (i.e. those insects that primarily feed on benthic algae) was higher in open reaches compared to forested reaches in all sites except Picketts Mill; however, the overall differences did not result in a significant effect of canopy cover. Shredders (i.e. insects that feed primarily on leaf material) accounted for a very small portion of the invertebrate density, and there were no differences in densities between reaches (Table 2.4; Fig 2.2).

Terrestrial invertebrate inputs were numerically dominated by Diptera and Collembola at all sites (Fig. 2.3). We expected overall inputs to be higher in forested reaches, where insects were more likely to fall from the canopy into stream reaches. Contrary to our expectation, there was a higher density of terrestrial invertebrates in open vs. forested reaches (Table 2.4), primarily driven by higher densities of Diptera in open reaches (Fig 2.3). In contrast, forested reaches had higher mean terrestrial invertebrate biomass (0.092 g m^{-2}) compared to open reaches (0.044 g m^{-2}), although the difference was not significant. Forested reaches also exhibited a higher diversity of terrestrial invertebrate inputs compared to open reaches (Table 2.4).

Eurycea cirrigera, the Southern two-lined salamander, was the only salamander found at any of the three streams sampled, and all but 3 individuals were larvae. The abundance and CPUE varied more across streams than between reaches, and there were also no differences in salamander size between open and forested reaches (Table 2.4). Notably, at Clark Creek, a site with abundant cobble/pebble riffle habitat and greatest salamander abundances, we captured 100 individuals in the open reach compared to 18 individuals in the forested reach. Larval salamander gut contents revealed similar diversity of diets between open and forested reaches (Table 2.4).

There were higher densities of fishes in open (9.0 m^{-2}) vs. forested (4.9 m^{-2}) reaches (Table 2.5). These higher overall densities in open reaches were driven by higher densities of *C. oligolepis*, the largescale stoneroller (an herbivore), and *Fundulus stellifer*, the southern studfish (Fig. 2.4). There were no differences in insectivorous cyprinids, centrarchids, or benthic invertivores (Percidae and Cottidae) between open and forested reaches. These comparisons were consistent across habitat type; riffle habitats had higher *C. oligolepis* densities in open reaches, pool habitats had higher *F. stellifer* densities in open reaches, and no other fish group showed significant differences between reaches within habitat types (Fig 2.4).

There were no differences in fish assemblage integrity terms of species richness or richness of endemics to cosmopolitans (E/C) (Table 2.5). Trends suggested a higher abundance of endemics relative to cosmopolitan species in forested reaches, and lower dominance by 1 or 2 taxa in forested reaches; however, none of these paired comparisons were significant. There were significantly higher densities of tolerant species and habitat generalists in the open reaches relative to the forested reaches (Table 2.5). These differences were driven primarily by higher densities of *C. oligolepis*, which is considered a tolerant and habitat generalist species.

Four cyprinid fish species were significantly larger in forested reaches, and these differences were driven by young of the year cyprinids collected in the open reach of Westbrook Creek (Table 2.6). *C. oligolepis*, *Moxostoma duquesnei*, *Lepomis macrochirus*, and *L. microlophus* were significantly larger in open reaches, potentially reflecting differences in food resources. Conversely, *Noturus leptacanthus*, *F. stellifer*, *Cottus carolinae zopherus*, *L. auritus*, and *P. nigrofasciata* were significantly larger in forested reaches (Table 2.6).

Based on gut content analysis, we found no significant differences in the diversity of prey items in the guts of the invertivores (*P. nigrofasciata*, *C. callistia*) or the trophic generalists (*L. auritus* and *H. etowanum*; Table 2.5). *Lepomis auritus* had higher proportional densities of dipteran pupae in open reaches, and higher arachnids in forested reaches (Appendix 2.1). Both water-column feeding fishes, *L. auritus* and *C. callistia* showed trends of higher proportions of terrestrial prey items in forested reaches; however, the differences were not significant. Although we hypothesized that fishes would consume a higher proportion of scraping insects in open reaches, none of the four fishes had differences in aquatic mayflies (Ephemeroptera, primarily scrapers) in their guts between open and forested reaches (Appendix 2.1).

Discussion

Changes in stream quality/integrity

Habitat diversity, a measure commonly used to evaluate stream quality, was remarkably unaffected by reach-scale riparian deforestation in our study. In fact, of 27 morphometric, bed texture, and habitat diversity (depth, velocity, particle size) variables measured, the only difference between reaches was in wetted width (Appendix B.4). Narrower channels in open (pasture) vs. forested reaches have been found in numerous studies (Sweeney 1993, Davies-Colley 1997, Scarsbrook and Halliday 1999, Hession et al. 2003); this is likely due to higher floodplain accretion rates and higher sediment trapping in streams with grassy bank vegetation (Allmendinger et al. 1999). Because mean depth and baseflow discharge were not different between reaches (Appendices B.2 and B.4), mean velocities were likely lower in forested reaches. LWD is also important in structuring stream habitat, and we found higher amounts of stable (attached) wood in forested reaches, but longer, pool-forming LWD in open reaches. Although local riparian deforestation may not eliminate LWD from stream systems, long-term reductions in supply might have an important influence on LWD volume and habitat complexity in urban streams.

Water quality was not expected to differ between open and forested reaches, since chemistry was assumed to be driven by catchment-scale factors. Although most of the water quality variables did not differ between reaches, we measured higher amounts of dissolved oxygen in open reaches. This difference is likely due to increased photosynthesis in open reaches that have higher amounts of primary producers. We also found higher concentrations of NO_2/NO_3-N in forested reaches. Although riparian zones are typically sinks for nitrate, Groffman et al. (2003) found that nitrification is increased in urban systems due to lowered water tables. Uptake of nitrogen by algae in open reaches may also be important in driving lower ambient NO_2/NO_3-N concentrations in open reaches. Local riparian deforestation increased summer minimum water temperatures by about 1°C; however, we did not observe significant annual responses of temperature and temperature fluctuations to riparian deforestation. Longer open reaches are likely to alter stream temperatures more significantly, which may have large impacts on fishes that cannot tolerate warmer temperatures (Scott and Helfman 2001).

Reach-scale biotic integrity was largely unaffected by local canopy cover. Although these streams had lower macroinvertebrate and EPT richness relative to other streams in the area, there were no differences between open and forest reaches. This supports results from Hession et al. (2002) and suggests that macroinvertebrates are driven more by catchment-scale factors than reach-scale forest cover. Salamander abundances did not differ based on forest cover, supporting the results of Hawkins et al. (1983) and Wilson and Dorcas (2003). Again, the reduced richness in these urban streams (only 1 species observed) suggests that salamanders are related to catchment disturbance rather than riparian disturbance. Total and endemic fish species richness did not indicate differences in fish assemblage integrity between reaches. Although fish assemblages had higher densities of tolerant and habitat generalist species in open reaches, these were driven by the abundance of two species (*C. oligolepis* and *F. stellifer*) and do not reflect an overall community shift in biotic integrity between reaches. The lack of response in biotic assemblages may be a consequence of minimal differences in habitat quality with reach-scale riparian deforestation. It is also possible that urban streams are already so depauperate of fishes that further changes in land cover would not affect the assemblage. However, these streams still have 10 to 18 fish species, which is common for streams of similar size in this basin (Leigh et al. 2002). It is more likely that the high mobility of fishes and the presence of sufficient habitat is preventing any local depletion of fish species. Given these factors, the open reaches could also be areas of "sink" fish populations, supported by upstream or downstream forested areas (Gotelli and Taylor 1999, Jones et al. 1999).

Changes in the trophic basis of production

Reduced tree density in deforested reaches resulted in reduced organic inputs into the stream. On average, the open reaches had one-fifth the amount of trees compared to forested reaches, which is expected to translate to a significant reduction in leaf inputs. For example, in a study looking at the effects of forest clearcutting on a headwater stream in North Carolina, tree densities were reduced from 103.1 ft^2 $acre^{-1}$ to 34.4 ft^2 $acre^{-1}$ (Elliot et al. 1997), and there was a concomitant reduction in litterfall (259.2 g m^{-2} to 4.2 g m^{-2}) and blow-in (174.8 g m^{-1} to 38.6 g m^{-1}) inputs to the stream (Webster and Waide 1982). There was also less LWD in open reaches, thereby reducing the amount of allochthonous material and stable surface area for autochthonous production. Georgia Forestry BMPs recommend leaving 50 ft^2 basal area of trees per acre evenly distributed throughout riparian areas. This is lower than the mean basal area for both our forested (2097 ft^2 $acre^{-1}$) and open (469 ft^2 $acre^{-1}$) reaches (although two sites had open reaches on one bank that were lower than this level, see Appendix B.1, GA EPD 1999). Since the differences in tree densities in our study translated to differences in LWD and expected leaf inputs to the streams, this recommendation appears inadequate for maintaining natural levels of organic inputs to streams.

As expected, we found a higher amount of chlorophyll *a* in open vs. forested reaches, suggesting that light is limiting primary production in the forested reaches. In terms of biofilm biomass, we only found differences on rock substrates, emphasizing the importance of stable substrate for biofilm production in these streams. These significant differences in chlorophyll *a* and biofilm biomass between reaches create an important difference in the trophic basis of production, which was expected to translate to higher trophic levels. There were trends toward higher overall densities of benthic invertebrates in open reaches, and benthic invertebrates were more variable in open reaches, suggesting that the invertebrates may be patchy. Contrary to our hypotheses, we did not observe any differences in the densities of scraping or shredding invertebrates between reaches. Because we only sampled invertebrates in one season (spring), we were unable to determine whether differences existed at other times of the year when food resources may differ more between open and forested reaches. The lack of differences in functional feeding groups may also be due to invertebrate consumption by higher trophic levels. However, there were no differences in the proportion of scraping insects (Ephemeroptera) in the guts of insectivorous fishes or salamanders. Nonetheless, there were significantly higher densities of herbivorous fishes (*C. oligolepis*) in open reaches, presumably driven by higher amounts of algae.

Because terrestrial invertebrates are also important food resources for some insectivorous fishes, we examined differences in terrestrial invertebrate inputs and subsequent differences in fish diets between reaches. Contradicting our expectations, we found higher densities of terrestrial invertebrate inputs in open reaches; however, similar results were also seen in Romaniczyn (2000). These higher densities were primarily attributed to increased inputs of small-bodied Diptera and Collembola in open reaches. Thus, there were trends toward larger animals and higher terrestrial invertebrate biomass in forested reaches. Since drift-eating fish are visual predators and tend to select larger invertebrates for consumption, these differences in terrestrial inputs were expected to translate into differences in fish diets. The water column fishes, *L. auritus* and *C, callistia* showed trends of higher % terrestrial prey in forested reaches;

however, the differences were not significant. While some studies have found reduced dependence on terrestrial organic matter associated with riparian deforestation (Hicks 1997, Hession et al. 2002, England and Rosemond 2004), other results suggest that annual prey consumption by fishes can be similar even with drastically different inputs of terrestrial invertebrates (Kawaguchi and Nakano 2001). This may be due to the mobility of fishes and/or the importance of invertebrate drift from upstream patches. Because stomach contents do not account for variable digestion of prey or the relative incorporation of terrestrial vs. aquatic-derived carbon in the fish tissue, we could not make any definitive conclusions about the relative contribution of food items into fish biomass. Further analysis using stable isotopes in conjunction with additional fish diet analysis would give a better picture of trophic shifts associated with riparian deforestation in urban streams.

Four fish species were significantly larger in open reaches, and 9 fish species were larger in forested reaches, suggesting that multiple mechanisms likely contribute to differences in lengths of fish species between reaches. Two species of *Cyprinella* and *Notropis* exhibited higher average lengths in forested reaches due to young of year (YOY) fishes in open reaches of Westbrook Creek. These high abundances of YOY fishes are likely a result of patchy distributions of these schooling fish in shallow, run habitat, rather than limitations based on riparian cover. Differences in *F. stellifer* between reaches was also driven by higher abundances of YOY fishes in open reaches; however, YOY were present at multiple sites, so this might reflect a difference in habitat preference between cover types. *C. carolinae zopherus*, *N. leptacanthus*, and *P. nigrofasciata*, all benthic invertivores, were larger in forested reaches, suggesting possible differences in biomass of benthic invertebrates between reaches. Finally, *C. oligolepis* and two species of generalist predators were larger in open reaches, potentially reflecting higher nutritional quality and assimilation efficiency of algal-derived resources (Cummins 1974, Rosenfeld and Roff 1992). Food, habitat, and spawning preferences may all play important roles in explaining the distribution of different size individuals in certain reaches, and these differences, if large enough to affect fish life cycles (e.g. by inhibiting reproduction), may have important implications for species persistence.

Putting the results in context

For our study, sites were selected in urbanizing areas to determine the influence of reach-scale forested canopies for urban stream integrity. Because this design did not allow us to assess sites with higher forest cover (vs. urban cover) in the catchment, we compared our data to a previous study that selected sites randomly within the Etowah River basin and had a gradient of urban land cover (Leigh et al. 2002). Compared to the earlier Etowah study, the sites in our study had higher mean % urban cover (24.7 vs 15.0) and lower % agriculture land cover (7.5 vs 22.1), but had similar % forest in catchments and 30 m riparian corridors (Table 1.7). In terms of water quality, the sites in the current study had higher mean dissolved oxygen, conductivity, pH, and turbidity; however, they were on the lower ends of the ranges in nutrient concentrations. Mean algal biomass and chlorophyll *a* concentrations were similar for the sites in the two studies. The sites in the current study had lower richness, EPT richness, and densities of benthic invertebrates sampled in the riffle habitats, suggesting lower macroinvertebrate integrity for sites in our study relative to other streams in the Etowah basin (Roy et al. 2003b). Although fish richness was similar in the two studies, our study had much lower relative abundances of endemic species, with abundances of endemics equal to half or less of the abundance of endemics and cosmopolitans (i.e. $E/(E+C) \leq 0.5$ vs ≤ 4.1 in earlier Etowah study, Table 7, Walters et al. 2003, Leigh et al. 2002). The lower water quality and biotic integrity in this study reflects the influence of higher urban land cover, and suggests that local riparian forest land cover may not be adequate to maintain healthy fish assemblages in streams with already reduced assemblage integrity.

Because there were only 5 small streams within the Etowah River basin that fit our design criteria, we compared our results to those from a study by Hession et al. (2002) in the Piedmont of southeast Pennsylvania and northern Delaware to evaluate the generality and significance of our results. The Hession et al. (2002) study included 12 pairs of sites with forested and non-forested (open) canopies that ranged in land cover from 1-91 % urban (mean = 56.3 % urban), and had similar criteria for site selection as our study. Where data were comparable, there were no conflicting results between the two studies (Table 8). Both indicated that open reaches had lower tree density, but trees in open reaches were larger in diameter. In addition, both studies showed that streams were narrower in open reaches, and there were no differences in depth between reaches. Lower amounts of LWD were found in open reaches in both studies, and Hession et al. (2002) demonstrated that bank stability was reduced in open reaches relative to forested reaches (not measured in this study). Other measures of physical habitat, including variability in velocity, depth, and particle size were not different between reaches in either study, suggesting that reach-scale canopies do not control local habitat diversity. Hession et al. (2002) found reduced SRP, NH_4, and DIN:SRP in open reaches. Although our study showed reduced NO_2/NO_3 in open reaches, there was not strong support for differences in other nutrient concentrations between reaches. Both studies found higher chlorophyll *a* concentrations in open reaches; however, there were no clear differences in benthic invertebrates between reaches. In terms of fishes, both studies found significantly higher densities in open reaches, including higher densities of tolerant fish species. Although the studies measured trophic composition in different ways, both demonstrated a trophic response to increased algal resources in open reaches, as indicated by increased herbivore densities in this study and increased $\partial^{13}C_{\mathrm{periphyton}}$ in consumers in Hession et al. study (Table 8). The consistent results in these two studies strengthen the conclusion that reach-scale riparian deforestation may result in trophic responses but has minimal effects on habitat quality and biotic integrity.

In this study, we analyzed all biotic data in terms of densities (i.e. m^{-2} area) rather than per stream length because we felt that these were the most appropriate units for comparing adjacent stream reaches. Given the reach conditions, we found higher *densities* of fishes, terrestrial invertebrate inputs, algal chlorophyll *a*, etc. in open reaches, suggesting that this reach could support more primary producers and consumers on per m^2 basis. However, because streams were ~20% wider in forested reaches compared to open reaches, there may be differences in overall production and nutrient uptake potential per m stream length.

Implications for urban riparian reforestation

Many land management efforts incorporate riparian protection into plans and assume that these are sufficient to protect stream ecosystems from disturbance in the landscape (Newbold et al. 1980, Growns and Davis 1991). However, there is limited research on whether forested riparian areas are effective at protecting streams against urban disturbance, and, if so, what amount of urbanization can be mediated with forested riparian areas. In Georgia, the Erosion and Sedimentation Control Act of 1975 (O.C.G.A. 12-7) requires 50 foot buffers on all trout streams and 25 foot buffers on all other streams throughout the state. Although many counties have adopted stricter buffer requirements, the exclusive use of riparian buffers as stream protection tools seems unjustified without further exploration of conditions necessary for forested areas to be protective. For example, although continuity of riparian forest may be essential for stream protection (Jones et al. 1999), current regulations do not address retrofit of gaps in riparian forests. Jackson et al. (2001) observed many small bank failures associated with clearcutting to the edge of streams, and these contributed to high amounts of sediment loading. Such gaps in riparian forests and stormwater drains leading directly into streams may be driving stream ecosystem integrity regardless of riparian regulations (Scarsbrook and Halliday 1999). These upstream and catchment-scale factors need to be addressed before assuming that local-scale riparian protection is effective and sufficient at protecting stream ecosystems.

Although riparian "buffers" imply a goal of protecting stream ecosystems, local forested riparian areas are limited by their ability to influence upstream and downstream disturbances (Allan and Johnson 1997). Many studies have demonstrated the importance of catchment-scale land cover relative to local-scale land cover in driving habitat quality and biotic integrity (Richards et al. 1996, Roth et al. 1996, Allan et al. 1997). Although Roth et al. (1996) found that land cover within riparian areas was correlated with fish and habitat, these relationships only existed for riparian areas for the entire upstream network, not local riparian cover. This study directly tested changes associated with local riparian cover, and also found that habitat quality and biotic integrity were not affected by local riparian deforestation. These results suggest that preservation of forested riparian fragments is not adequate to protect stream ecosystems (Harding et al. 1998).

Our results demonstrate that reach-scale openings in riparian canopies can create local patches of increased food resources. Algal resources may be important for supporting food webs in urban systems where organic inputs are minimized. Although we observed trends of higher benthic invertebrate densities and significantly higher fish densities in open reaches, biomass and productivity may be similar between reaches, especially when calculated per linear length of stream reach (i.e. accounting for wider forested reaches). Further, the switch from allochthonous- to autochthonous-supported food webs, particularly in smaller, headwater streams, may have important implications in terms of altered ecosystem function in these naturally forested landscapes (Vannote et al. 1980).

We caution against concluding that forested canopies are not important, since the role of extent and continuity of riparian forests was not tested in this study. These sites had > 50 % forest in the 30 m riparian zone in the upstream drainage network, so we do not know how these urbanizing streams would respond to a complete loss of riparian cover and associated organic matter inputs. Although we did not have enough streams in this study to correlate upstream % riparian forest cover with measures of stream quality, Clark Creek had notably less % riparian forest (53%) compared to the other sites (70-81%), so we would expect reduced organic inputs in this site. However, Clark Creek did not have remarkably lower LWD or other measures of organic material compared to other streams. For fish assemblages, other research shows that riparian cover in the upstream network is important for maintaining sensitive fishes, while local reductions in riparian forest cover increase tolerant fishes (A.H. Roy, unpublished data). Further studies addressing effects of local riparian forest cover with a range of network deforestation would be useful to understand the relative importance of upstream land cover on reach-scale stream conditions.

Conclusions

Previous studies have shown that long-term, low levels of urbanization can negatively impact fish assemblages, structuring them toward more tolerant species that are habitat and trophic generalists (Weaver and Garman 1994, Wang et al. 2000, Walters et al. 2003). Although maintenance of riparian forests have been successfully used to mediate the negative effects of agricultural and forestry practices on aquatic ecosystems, there are few studies addressing the effectiveness of riparian forests at protecting stream ecosystems from urban disturbance. Hession et al. (2002) proposed a framework suggesting that riparian reforestation improves stream ecosystem health; however, the relative benefits may be reduced in watersheds with increasing urbanization. Although we did not test the function of reach-scale riparian forests along a gradient of urbanization, our study suggests that at low levels of urbanization, reach-scale riparian forests provided minimal benefits to reach-scale stream condition. The combined results of this study and the Hession et al. (2002) study provide strong evidence that although riparian forests may be important regulators of stream width, temperature, and the food/energy base at the reach-scale, they do not dictate stream habitat quality or biotic integrity. Thus, reach-scale riparian reforestation is necessary but not sufficient to protect aquatic resources in urbanizing landscapes.

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Table 2.1. Site characteristics for each study stream within the Etowah River basin. Land cover is based on 2001 *LandSat* TM imagery.

	Clark	Copper Sandy	Lawrence	Picketts Mill	Westbrook
Location					
UTM Northing (NAD 1983)	717590	746302	701124	708236	707590
UTM Easting (NAD 1983)	3773997	3777695	3758149	3761223	3770001
County	Cobb/Cherokee	Fulton	Paulding	Cobb/Paulding	Paulding/Cobb
Open reach					
Type of landuse	golf course	horse/cow pasture	ball fields	horse pasture	golf course
Location (relative to forest reach)	upstream	downstream	downstream	upstream	downstream
Length (m)	554	350	348	368	400
Basin and stream morphometry					
Drainage area (km^2)	16.2	16.0	11.8	15.1	18.9
Stream length (km)	36.9	34.2	24.4	34.1	46.2
Drainage density ($km\ km^{-2}$)	2.3	2.1	2.1	2.3	2.4
Elevation (m)	860	980	880	900	740
Stream slope	0.0034	0.0054	0.0035	0.0039	0.0034
Catchment land cover					
% Impervious	17.3	14.5	9.7	13.2	8.8
% Urban	49.3	20.5	8.8[a]	25.6	19.5
% Forest	40.5	55.1	73.9	61.0	69.3
% Agriculture	3.5	13.0	9.1	7.8	4.2
% Open water	0.7	1.9	1.1	0.4	0.7
Riparian land cover (network)					
% Impervious	13.1	10.7	9.2	9.1	7.0
% Urban	39.4	10.0	4.8	14.0	9.2
% Forest	52.9	70.3	81.4	76.0	79.9
% Agriculture	1.5	5.1	7.6	3.2	3.3
% Open water	2.7	8.8	3.3	1.8	3.3
Road crossings (no. km^{-1} stream)	0.51	0.26	0.61	0.47	0.19
Impoundments (no. km^{-1} stream)	0.46	0.55	0.29	0.53	0.28

[a] Reclassification of land cover subsequent to site selection resulted in < 15% urban land cover

Table 2.2. Comparison of riparian cover, water quality, and selected physical habitat variables between open and forested reaches (paired *t*-test for means, df = 4). Bold indicates $p \leq 0.05$.

	OPEN		FOREST			
	mean	st. dev.	mean	st. dev.	*t*	*p*
Cover						
% Canopy cover (July 2002)	**32.3**	**21.0**	**77.6**	**12.5**	**4.73**	**0.005**
% Canopy cover (Sept. 2003)	**39.9**	**20.3**	**90.4**	**5.8**	**6.00**	**0.002**
Trees density (no. m^{-2})	**0.038**	**0.037**	**0.179**	**0.055**	**4.53**	**0.010**
Tree diameter (cm)	22.2	12.3	11.6	2.6	1.68	0.096
Basal area (ft^{2}) per acre	**469**	**350**	**2097**	**922**	**3.66**	**0.018**
Water quality						
Dissolved oxygen (mg L^{-1})	**9.4**	**1.0**	**8.9**	**1.0**	**2.51**	**0.033**
Specific conductance (µs cm^{-1})	99.2	19.3	90.0	23.6	1.47	0.108
pH	8.2	0.4	8.0	0.5	0.87	0.216
Turbidity (NTU)	12.8	9.2	10.8	4.7	0.82	0.229
SRP (µg L^{-1})	0.008	0.010	0.005	0.003	0.52	0.315
NH_4-N (µg L^{-1})	0.005	0.007	0.003	0.004	0.92	0.206
NO_2/NO_3-N (µg L^{-1})	**0.114**	**0.033**	**0.126**	**0.038**	**5.71**	**0.002**
DIN:SRP	92.1	158.9	18.3	11.8	0.91	0.215
Total suspended solids (mg L^{-1})	24.3	6.9	22.7	3.1	0.75	0.247
Temperature						
Annual mean (°C)[a]	15.60	2.20	15.24	2.01	1.78	0.075
Annual standard deviation (°C)[a]	6.87	0.69	6.60	0.47	0.97	0.194
Annual daily diel (°C)[a]	4.34	0.98	3.55	0.70	1.43	0.113
Annual degrees days[a, b]	5995	239	5934	200	0.51	0.324
Summer daily minimum (°C)[b]	**21.06**	**0.88**	**20.13**	**0.17**	**2.34**	**0.051**
Summer daily maximim (°C)[b]	27.13	1.85	25.86	1.35	1.15	0.166
Summer daily diel (°C)[b]	4.76	1.68	2.84	0.85	1.53	0.112
Channel morphology						
Riffle area (m^{2})	101.1	61.3	213.8	176.3	1.37	0.121
Pool/run area (m^{2})	533.4	212.3	542.8	161.4	0.17	0.436
% riffle area	17.2	10.1	27.1	18.4	1.12	0.148
Width mean (m)	**4.1**	**1.2**	**5.0**	**1.1**	**2.12**	**0.051**
Depth mean (m)	0.22	0.08	0.25	0.12	0.46	0.334
Large woody debris						
Total # pieces	**10.0**	**5.1**	**31.8**	**17.6**	**2.92**	**0.022**
% pool-forming	**21.2**	**13.3**	**8.7**	**5.4**	**2.28**	**0.042**
% attached	**38.1**	**23.5**	**56.5**	**13.8**	**2.62**	**0.029**
Mean length (cm)	**167.2**	**68.5**	**132.4**	**56.5**	**4.20**	**0.007**
Mean diameter (cm)	16.2	3.4	15.5	3.1	0.37	0.365
Total volume (m^{3})	0.4	0.4	1.3	1.3	1.51	0.103

[a] Clark open 4/1 to 6/26 missing, replaced with forest data; Westbrook data ends 6/11

[b] No data for Westbrook Creek

Table 2.3. Comparison of biofilm and algal chlorophyll *a* concentrations on sand and rock surfaces between open and forested reaches (paired *t*-test for means, df = 4). AFDM = ash free dry mass. Bold indicates $p \leq 0.05$.

	OPEN		FOREST			
	mean	st. dev.	mean	st. dev.	*t*	*p*
Biofilm						
Total AFDM (g m^{-2})	32.90	9.69	100.87	150.63	1.04	0.179
Sand AFDM (g m^{-2})	66.50	45.10	130.63	158.75	1.12	0.172
Rock AFDM (g m^{-2})	**13.86**	**6.60**	**8.13**	**5.49**	**2.32**	**0.051**
Algae						
Total chlorophyll *a* (mg m^{-2})	**32.73**	**16.86**	**13.67**	**11.30**	**2.17**	**0.048**
Sand chlorophyll *a* (mg m^{-2})	25.76	10.24	15.30	12.40	1.69	0.095
Rock chlorophyll *a* (mg m^{-2})	28.30	23.70	8.13	4.18	1.76	0.089
Algae/Biofilm						
Total	**1.082**	**0.92**	**0.4532**	**0.5406**	**2.31**	**0.041**
Sand	**1.15**	**1.645**	**0.1012**	**0.0775**	**3.61**	**0.018**
Rock	1.223	0.52	0.7588	0.6216	1.16	0.166

Table 2.4. Mean values for benthic invertebrates (5 replicates) and terrestrial invertebrates (10 replicates) samples, and total salamander measures for open and forested study reaches. Open and forested reaches were compared using a paired *t*-test for means, df = 4. Bold indicates $p < 0.05$.

	Clark		Copper Sandy		Lawrence		Picketts Mill		Westbrook		OPEN		FOREST			
	open	forest	open	forest	open	forest	open	forest	open	forest	mean	st. dev.	mean	st. dev.	*t*	*p*
Benthic invertebrates																
Richness																
Total richness	20	20	29	19	18	22	24	23	23	18	22.8	4.2	20.4	2.1	1.01	0.185
Insect richness	17	14	19	10	13	18	22	21	18	14	17.8	3.3	15.4	4.2	1.06	0.075
EPT richness	6	6	6	3	3	8	9	8	9	5	6.6	2.5	6.0	2.1	0.38	0.361
Density																
Total density (no. m^{-2})[a]	1131.1	1093.3	3268.9	1235.6	551.1	420.0	1286.7	1435.6	1846.7	1120.0	1616.9	1032.3	1060.9	382.8	1.75	0.077
St. dev. density (no. m^{-2})[a]	**1403.6**	**452.6**	**1601.9**	**442.5**	**299.6**	**194.7**	**1527.5**	**1728.8**	**629.1**	**297.0**	**1092.3**	**589.2**	**623.1**	**627.3**	**2.74**	**0.026**
Insect density (no. m^{-2})[a]	962.2	651.1	1453.3	353.3	415.6	295.6	526.7	655.6	1648.9	911.1	1001.3	546.3	573.3	251.2	1.91	0.065
St. dev. insect density (no. m^{-2})[a]	1332.4	199.5	668.1	140.2	283.0	140.7	236.0	661.9	677.4	226.2	639.4	439.4	273.7	220.2	1.65	0.087
EPT density (no. m^{-2})[a]	562.2	102.2	168.9	37.8	91.1	97.8	133.3	206.7	306.7	106.7	252.4	191.1	110.2	60.8	1.76	0.077
St. dev. EPT density (no. m^{-2})[a]	1025.6	65.9	99.9	33.0	65.5	36.9	28.3	248.7	175.2	56.4	278.9	420.9	88.2	90.7	0.85	0.222
Chironomid density (no. m^{-2})[a]	273.3	468.9	986.7	251.1	242.2	102.2	320.0	328.9	1062.2	635.6	576.9	410.4	357.3	204.3	0.45	0.338
% Chironomids[b]	24.2	42.9	30.2	20.3	44.0	24.3	24.9	22.9	57.5	56.7	36.1	14.4	33.4	15.8	1.31	0.131
FFG Density																
Shredders	0.0	2.2	6.7	2.2	6.7	13.3	11.1	4.4	6.7	2.2	6.2	4.0	4.9	4.8	1.35	0.124
% Shredders[b]	0.0	0.3	0.5	0.6	1.6	4.5	2.1	0.7	0.4	0.2	0.9	0.9	1.3	1.8	0.60	0.291
Scrapers	91.1	62.2	82.2	37.8	95.6	55.6	55.6	117.8	275.6	91.1	120.0	88.3	72.9	31.6	1.31	0.131
% Scrapers[b]	9.5	9.6	5.7	10.7	23.0	18.8	10.5	18.0	16.7	10.0	13.1	6.8	13.4	4.6	0.25	0.407
Filterers	533.3	71.1	175.6	20.0	31.1	66.7	51.1	84.4	86.7	20.0	175.6	207.5	52.4	30.3	1.40	0.117
% Filterers[b]	55.4	10.9	12.1	5.7	7.5	22.6	9.7	12.9	5.3	2.2	18.0	21.1	10.8	7.8	0.72	0.255
Gatherers	277.8	471.1	991.1	251.1	253.3	124.4	353.3	360.0	1075.6	646.7	590.2	407.3	370.7	200.8	1.27	0.137
% Gatherers[b]	28.9	72.4	68.2	71.1	61.0	42.1	67.1	54.9	65.2	71.0	58.1	16.6	62.3	13.4	0.41	0.352
Predator	11.1	13.3	8.9	8.9	11.1	6.7	22.2	22.2	6.7	11.1	12.0	6.0	12.4	6.0	0.22	0.418
% Predator[b]	1.2	2.0	0.6	2.5	2.7	2.3	4.2	3.4	0.4	1.2	1.8	1.6	2.3	0.8	1.34	0.126
Terrestrial invertebrates																
Total density (no. m^{-2})[a]	**90.7**	**73.5**	**409.6**	**79.9**	**518.0**	**156.7**	**212.2**	**104.3**	**142.4**	**123.8**	**274.6**	**182.1**	**107.7**	**33.9**	**2.23**	**0.045**
Total biomass (g m^{-2})	0.027	0.203	0.076	0.051	0.034	0.035	0.028	0.041	0.057	0.127	0.044	0.022	0.092	0.073	1.32	0.129
Diversity (H')	**1.31**	**1.67**	**1.09**	**1.96**	**1.23**	**1.33**	**1.41**	**1.78**	**1.38**	**1.66**	**1.28**	**0.13**	**1.68**	**0.23**	**3.07**	**0.019**
Salamanders (Eurycea cirrigera)																
Abundance (no. individuals)	100	18	--	--	1	3	13	14	--	--	38.0	54.0	11.7	7.8	0.95	0.222
Catch per unit effort (CPUE)	1.25	0.23	--	--	0.02	0.05	0.19	0.20	--	--	0.5	0.7	0.2	0.1	0.93	0.225
Snout-vent length (mm)	2.97	2.75	--	--	3.50	3.13	2.45	2.70	--	--	3.0	0.5	2.9	0.2	0.59	0.308
Stomach content diversity (H')	2.06	1.86	--	--	0.64	1.08	0.81	0.91	--	--	1.2	0.8	1.3	0.5	0.64	0.294

[a] Log (x+1) transformed for analysis

[b] Arcsin squareroot (%/100) transformed for analysis

Table 2.5. Comparison between measures of fish assemblage composition and integrity between open and forested reaches (paired *t*-test for means, df = 4). Invertebrate prey diversity was assessed in four species of fish. Bold indicates $p < 0.05$.

	OPEN		FOREST			
	mean	st. dev.	mean	st. dev.	*t*	*p*
Overall measures						
Species richness	16	2.9	14.8	3.0	1.63	0.089
Density (no. m^{-2})	**9.0**	**2.4**	**4.9**	**1.7**	**4.78**	**0.004**
Abundance[a]	2.8	0.2	2.6	0.1	2.01	0.058
Rel. abundance 1 dominant spp.	0.37	0.10	0.29	0.10	1.27	0.136
Rel. abundance 2 dominant spp.	0.56	0.09	0.43	0.14	1.88	0.067
Endemic (E) & cosmopolitan (C) species						
Endemic richness	2	1	1.6	1.1	1.63	0.187
Endemic density (no. m^{-2})	1.05	0.74	0.85	0.70	0.88	0.215
Cosmopolitan density	**5.22**	**2.77**	**2.83**	**1.94**	**3.81**	**0.009**
Cosmopolitan density (no. m^{-2})	8.6	1.1	8.2	1.8	1.00	0.187
E/C (richness)	0.23	0.12	0.21	0.15	0.68	0.268
E/C (abundance)[b]	0.21	0.16	0.41	0.41	1.72	0.080
E/E+C (abundance)[b]	0.16	0.11	0.24	0.21	1.64	0.088
Tolerants						
Relative abundance	0.57	0.17	0.61	0.18	1.15	0.159
Density (no. m^{-2})	**5.38**	**2.04**	**2.99**	**1.64**	**4.43**	**0.006**
Habitat generalists						
Relative abundance	0.43	0.12	0.37	0.05	0.97	0.193
Density (no. m^{-2})	**4.30**	**1.35**	**1.74**	**0.75**	**3.98**	**0.008**
Gut content diversity (H')						
Lepomis auritus	0.19	0.01	0.20	0.01	1.56	0.096
Cyprinella callistia	0.07	0.07	0.17	0.02	1.82	0.105
Percina nigrofasciata	0.11	0.01	0.11	0.01	0.37	0.370
Hypentelium etowanum	0.11	0.02	0.11	0.01	0.04	0.484

[a] Log (x+1) transformed

[b] Abundance $x^{0.25}$ transformed prior to calculating ratio

Table 2.6. List of all fish species collected in study and their sensitivity categories (E = endemic, C = cosmopolitan, T = tolerant, G = habitat generalist). Comparison of mean fish standard lengths between open and forested reaches (two-sample *t*-test, assuming unequal variances). Bold numbers indicate significantly longer lengths ($p < 0.05$). Species with < 4 individuals in one or both reaches were not analyzed (indicated by --).

			Open		Forest			
Family Name *Scientific name*	Common name	Sensitivity	n	Mean length (mm)	n	Mean length (mm)	*t*	*p*
Petromyzontidae								
Ichthyomyzon sp. cf. gagei	southern brook lamprey		2	115	9	141	--	--
Cyprinidae								
Campostoma oligolepis	largescale stoneroller	C, T, G	1060	**59**	440	53	7.35	<0.001
Cyprinella callistia[a]	Alabama shiner	T, G	166	34	59	**55**	7.36	<0.001
Cyprinella venusta[a]	blacktail shiner		47	24	22	**76**	14.04	<0.001
Hybopsis sp. cf. winchelli	clear chub		5	56	4	56	0.54	0.310
Notropis longirostris	longnose shiner		107	26	8	33	1.69	0.065
Notropis stilbius[a]	silverstripe shiner		107	28	17	**47**	5.24	<0.001
Notropis xaenocephalus[a]	Coosa shiner	E	257	42	230	**47**	4.86	<0.001
Phenacobius catostomus	riffle minnow	E	1	80	0	--	--	--
Semotilus atromaculatus	creek chub	C, T	43	87	54	86	0.11	0.458
Catostomidae								
Hypentelium etowanum	Alabama hog sucker	T, G	210	83	194	81	0.72	0.236
Moxostoma duquesnei	black redhorse	C	52	**149**	18	113	3.00	0.002
Moxostoma poecilurum	blacktail redhorse	C	4	166	0	--	--	--
Ictaluridae								
Ameiurus brunneus	snail bullhead	C, G	1	145	1	56	--	--
Ameiurus natalis	yellow bullhead	C	1	127	0	--	--	--
Noturus leptacanthus	speckled madtom	C	69	42	17	**47**	1.87	0.036
Fundulidae								
Fundulus stellifer[b]	southern studfish		768	36	201	**54**	12.29	<0.001
Poecilidae								
Gambusia affinis	eastern mosquitofish	C, T	0	--	1	27	--	--
Cottidae								
Cottus carolinae zopherus	Coosa banded sculpin	E	13	47	25	**52**	1.74	0.048
Centrarchidae								
Lepomis auritus	redbreast sunfish	C, T	169	62	168	**70**	2.93	0.002
Lepomis cyanellus	green sunfish	C, T	41	62	108	67	1.30	0.099
Lepomis gulosus	warmouth	C		--	1	80	--	--
Lepomis macrochirus	bluegill sunfish	C, T	251	**63**	365	58	4.04	<0.001
Lepomis microlophus	redear sunfish	C	10	**68**	6	49	2.01	0.036
Micropterus coosae	Coosa bass		10	78	14	96	1.27	0.108
Micropterus punctulatus	spotted bass	C	15	60	29	57	0.75	0.229
Micropterus salmoides	largemouth bass	C	19	96	15	69	1.49	0.073
Pomoxis nigromaculatus	black crappie	C	0	--	1	110	--	--
Percidae								
Etheostoma scotti	Cherokee darter	E	157	42	128	41	0.49	0.313
Etheostoma stigmaeum	speckled darter	C	20	42	22	40	1.33	0.096
Perca flavescens	yellow perch	C	1	166	0	--	--	--
Percina kathae	Mobile logperch		11	85	2	80	--	--
Percina nigrofasciata	blackbanded darter	C, T	99	48	63	**59**	4.36	<0.001

[a] Difference in length due to young of year fishes in open reach of Westbrook Creek

[b] Difference in length due to young of year fishes in open reaches of multiple streams

Table 2.7. Mean and ranges for selected land cover, water quality, algal, benthic invertebrate, and fish variables for earlier Etowah (n = 29-31 sites) and this study (n = 10 sites). The earlier Etowah project involved sampling streams (8 – 121 km^2 catchments) within the Piedmont of the Etowah River basin in randomly selected catchments in 1999 (Leigh et al. 2002). Thus, the comparison largely reflects a difference in urban land cover, with sites from this study having higher % urban land cover (24.7%) compared to the larger population of streams within the basin reflected in the randomly selected catchments (15.0%). n/a indicates data not available.

	Randomly selected catchments		This study	
	Mean	Range	Mean	Range
Land cover				
% urban	15.0	(4.9 to 60.7)	24.7	(8.8 to 49.3)
% forest	61.7	(27.2 to 87.0)	60.0	(40.5 to 73.9)
% agriculture	22.1	(6.5 to 38.4)	7.5	(3.5 to 13.0)
Riparian forest	67.4	(33.7 to 94.6)	72.1	(52.9 to 79.9)
Water quality				
Dissolved oxygen (mg L^{-1})	8.5	(6.8 to 9.3)	9.1	(7.3 to 10.2)
Specific conductance (μs cm^{-1})	71.7	(21.3 to 171.6)	94.6	(51.6 to 115.2)
pH	7.1	(6.7 to 7.6)	8.1	(7.2 to 8.7)
SRP (μg L^{-1})	68.9	(7.5 to 117.2)	3.9	(0 to 15.9)
NH_4-N (μg L^{-1})	28.1	(4.8 to 90.6)	4.3	(0 to 17.5)
NO_2/NO_3-N (μg L^{-1})	345.3	(33.3 to 878.3)	115.3	(64.3 to 166.0)
Turbidity (NTU)	7.1	(2.6 to 16.0)	11.8	(3.8 to 27.2)
Total suspended solids (mg L^{-1})	11.2	(1.5 to 49.9)	16.8	(0 to 30.4)
Annual mean °C	15.2	(13.1 to 17.3)	15.4	(11.8 to 17.2)
Algae				
Biofilm biomass (g m^{-2})	n/a	n/a	66.9	(6.8 to 368.5)
Chlorophyll *a* (mg m^{-2})	26.2	(0.7 to 152.0)	23.2	(6.9 to 56.5)
Benthic invertebrates[a]				
Total richness	23.2	(8 to 40)	21.6	(18 to 29)
Insect richness	19.1	(5 to 36)	16.6	(10 to 22)
EPT richness	9.4	(0 to 23)	6.3	(3 to 9)
Density (no. m^{-2})	2045.8	(174.1 to 4777.8)	1338.9	(420 to 3268.9)
Fishes[b]				
Species richness	13.5	(10 to 23)	15.4	(10 to 18)
E/E+C (abundance)	2.24	(0 to 4.07)	0.37	(0 to 0.50)

[a] Comparison with riffle samples (vs. all habitats) from earlier Etowah study (Roy et al. 2003b)

[b] Comparison with small streams (15 km^2 ± 25%) from earlier Etowah study (Walters et al. 2003)

Table 2.8. Responses to reach-scale riparian deforestation in Piedmont streams of southeastern Pennsylvania and northern Delaware (n = 12 pairs, Hession et al. 2002) and northcentral Georgia (n = 5 pairs, this study). Up arrow (↑) indicates significantly higher in open reaches ($p < 0.05$), down arrow (↓) indicates significantly lower in open reaches ($p < 0.05$), “trend” indicates $0.05 < p < 0.20$, and -- indicates no difference ($p > 0.20$). n/a indicates variable not measured.

Responses to riparian deforestation					
Response variable	Hession et al.	This study	Response variable	Hession et al.	This study
Riparian vegetation			Organic matter		
tree density	↓	↓	leaf inputs	↓	n/a
tree size	↑	trend ↑	large woody debris	↓	↓
litter quality	--	n/a	Algae		
Channel morphology			biofilm biomass	↑	trend ↑
width	↓	trend ↓	chlorophyll *a*	↑	↑
depth	--	--	diatom composition	--	n/a
bed slope	--	--	Benthic invertebrates[a]		
Physical habitat			richness	n/a	--
rootwads	↓	n/a	density	n/a	trend ↑
bank erosion rate	↑	n/a	Terrestrial invertebrates		
temperature	n/a	trend ↑	density	n/a	↑
velocity variability	--	--	biomass	n/a	trend ↓
depth variability	--	--	diversity	n/a	↓
particle size	--	--	Fishes		
embeddedness	--	--	density	↑	↑
Nutrient concentrations			biomass	↑	n/a
SRP	↓	--	tolerant	↑	↑
NH_4-N	trend ↓	--	herbivores	n/a	↑
NO_2/NO_3-N	--	↓	Trophic composition		
DIN:SRP	↑	--	$\partial^{13}C_{periphyton}$	↑	n/a
			% terrestrial in guts[b]	n/a	trend ↓

[a] Hession et al. (2002) tested used a dissimilarity index to compare benthic invertebrate composition between forested and non-forested reaches and found no difference.

[b] *Lepomis auritus* and *Cyprinella callistia*

Figure legends

Fig. 2.1. Conceptual model of relations between reach-scale riparian deforestation and responses of basal food resources and consumers in urban streams. Riparian deforestation is expected to result in increases in temperature variability and algae, and reductions in habitat diversity, organic matter inputs, and terrestrial invertebrate inputs. The changes in temperature and habitat diversity are expected to negatively impact benthic invertebrate, salamander, and fish integrity. Reductions in terrestrial-derived carbon and increases in autochthonous carbon are expected to increase densities of algae-eating invertebrates and fishes, and decrease densities of shredding insects. We hypothesize that these changes in basal food resources will be reflected in the proportions of prey types consumed by salamanders and fishes. Responses in bold indicate significant effects observed in this study.

Fig. 2.2. Mean densities of benthic invertebrate functional feeding groups in open and forested reaches of each stream. Comparison in overall mean densities between open and forested reaches within streams (two-sample t-test assuming unequal variances, df = 4) and among all stream pairs (paired *t*-test for means, df = 4) is reported. Lines indicate standard error for overall mean densities. * $p < 0.05$.

Fig. 2.3. Mean densities of common terrestrial invertebrate groups (mean density > 1 m^{-2}) in open and forested reaches of each stream. Comparisons in overall densities between open and forested reaches within streams (two-sample *t*-test assuming unequal variances, df = 9) and among all stream pairs (paired *t*-test for means, df = 4) are reported. Lines indicate standard error for overall mean densities. * $p < 0.05$, ** $p < 0.01$.

Fig. 2.4. Densities (mean ± 1 SE) of selected fish families and species in riffle habitats, pool habitats, and all habitats combined. Open bars represent open reaches, solid bars represent forested reaches. Comparisons between mean densities for open (open bars) and forested (solid bars) reaches are reported (paired *t*-test for means, df = 4). * $p < 0.05$, ** $p < 0.01$.

Fig. 2.1

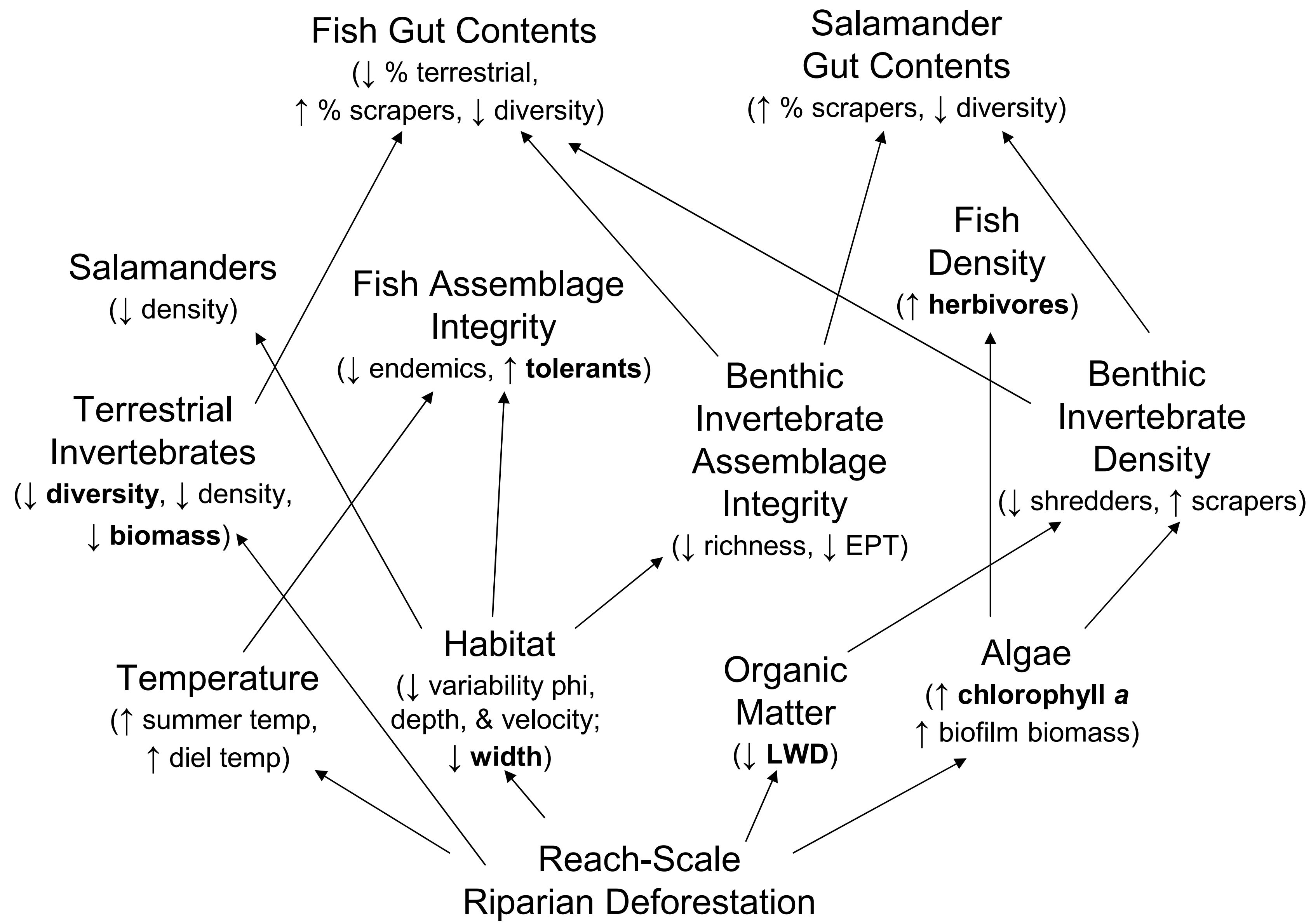
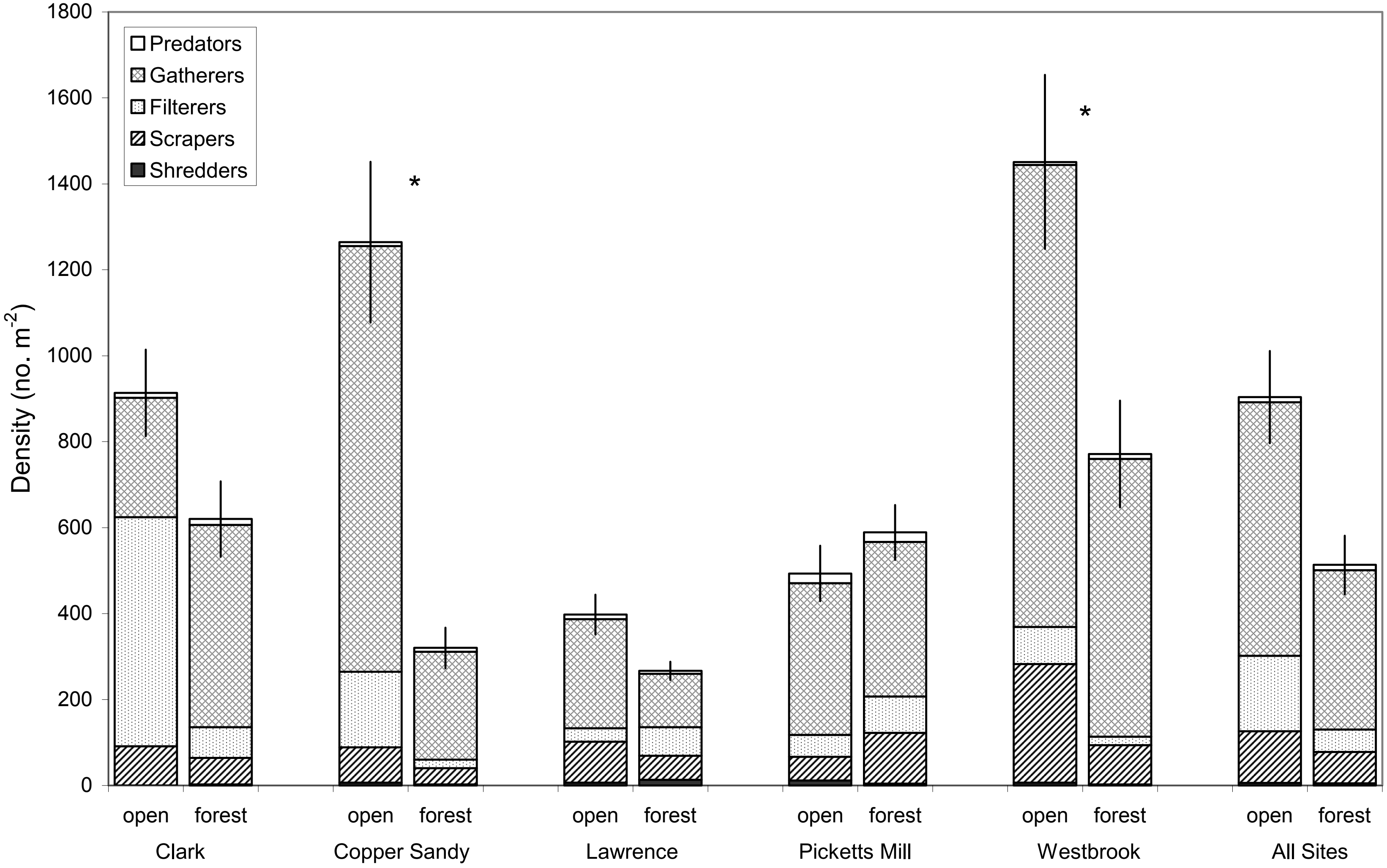

Fish Gut Contents
(↓ % terrestrial,
↑ % scrapers, ↓ diversity)
Salamander
Gut Contents
(↑ % scrapers, ↓ diversity)
Fish
Density
(↑ **herbivores**)
Salamanders
(↓ density)
Fish Assemblage
Integrity
(↓ endemics, ↑ **tolerants**)
Benthic
Invertebrate
Assemblage
Integrity
(↓ richness, ↓ EPT)
Benthic
Invertebrate
Density
(↓ shredders, ↑ scrapers)
Terrestrial
Invertebrates
(↓ **diversity**, ↓ density,
↓ **biomass**)
Habitat
(↓ variability phi,
depth, & velocity;
↓ **width**)
Organic
Matter
(↓ **LWD**)
Algae
(↑ **chlorophyll *a***
↑ biofilm biomass)
Temperature
(↑ summer temp,
↑ diel temp)
Reach-Scale
Riparian Deforestation

Fig. 2.2



Density (no. m-2)
0
200
400
600
800
1000
1200
1400
1600
1800
Predators
Gatherers
Filterers
Scrapers
Shredders
*
*
open
forest
open
forest
open
forest
open
forest
open
forest
open
forest
Clark
Copper Sandy
Lawrence
Picketts Mill
Westbrook
All Sites

Fig. 2.3

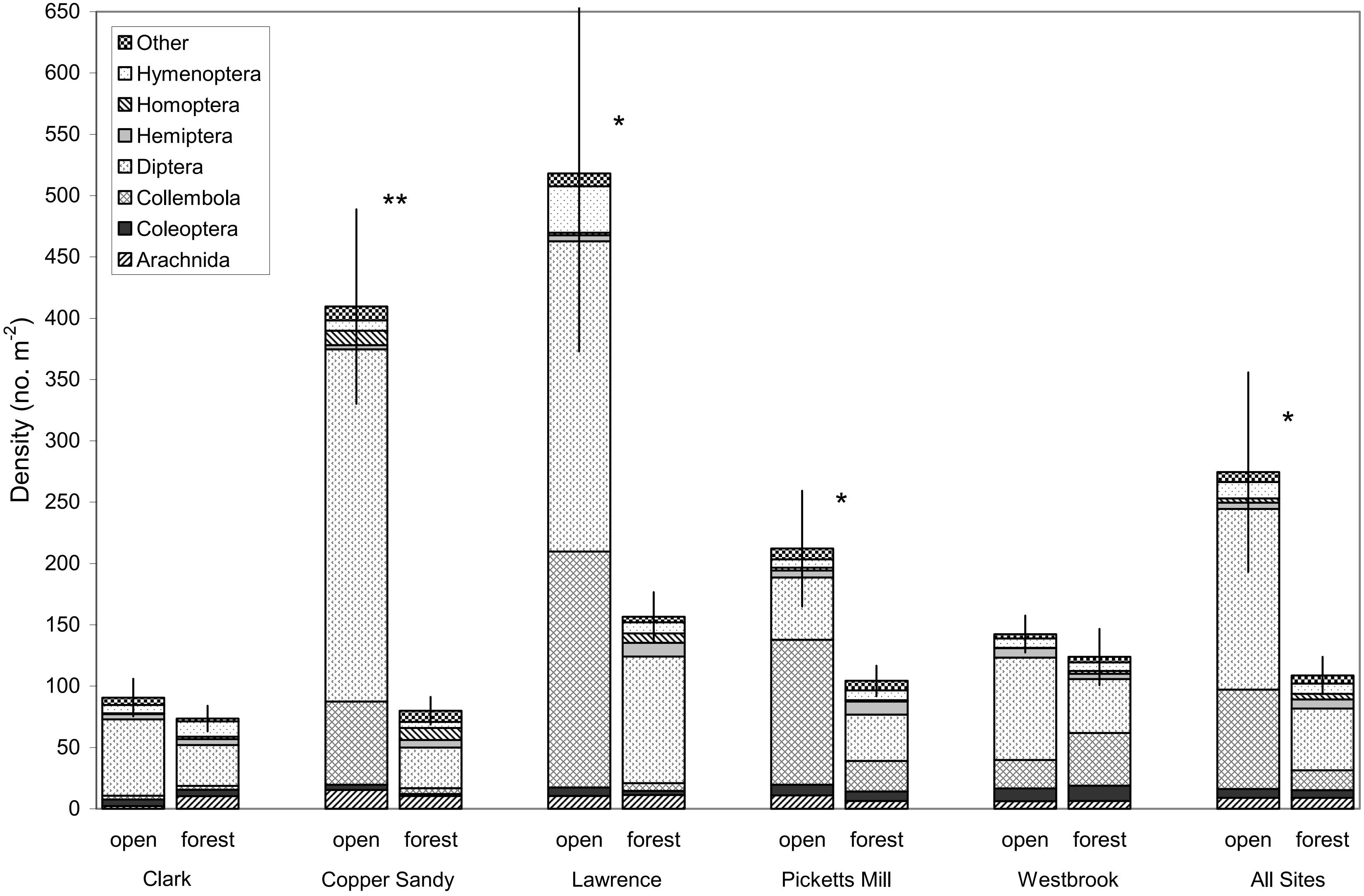
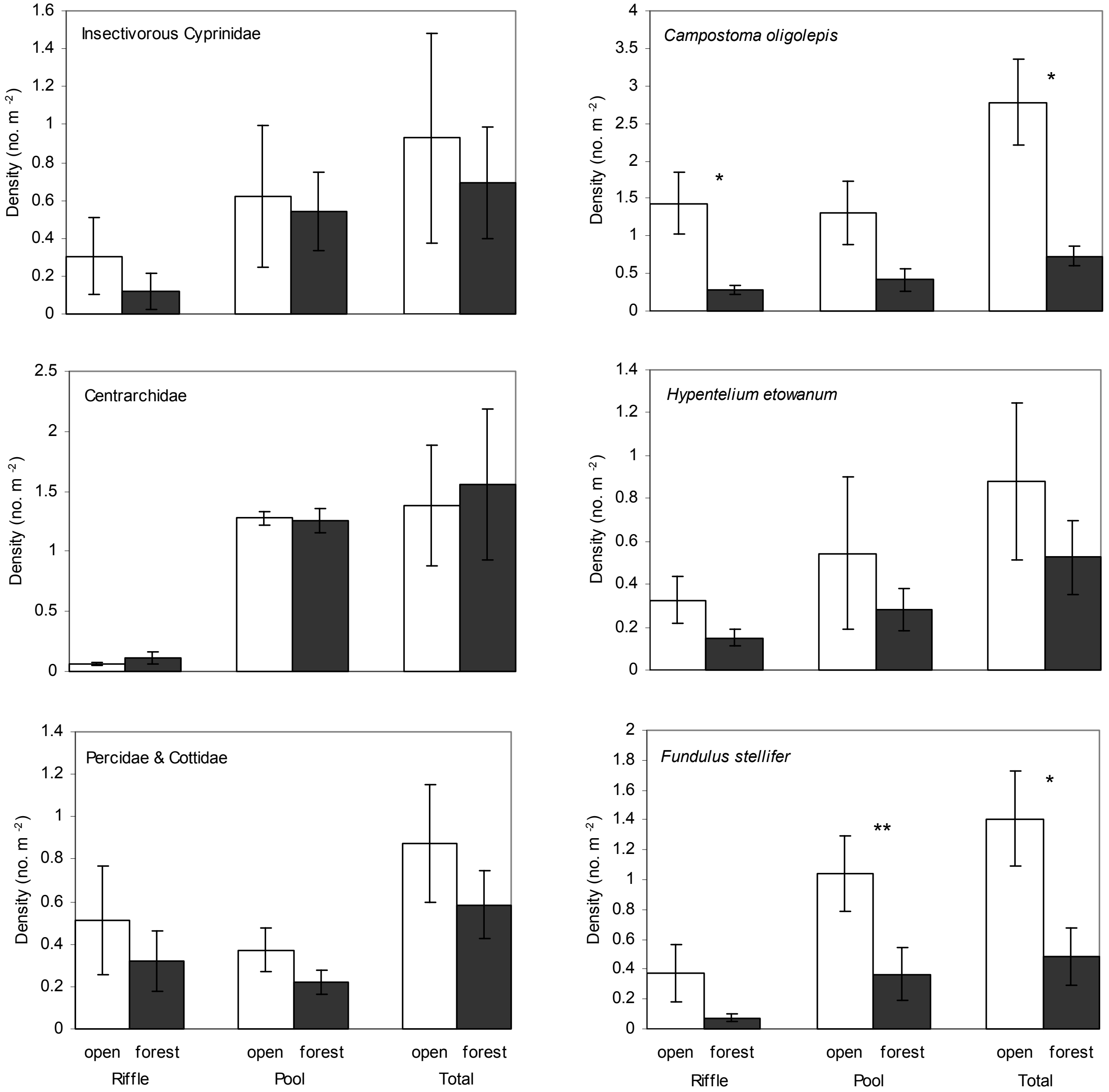

Other
Hymenoptera
Homoptera
Hemiptera
Diptera
Collembola
Coleoptera
Arachnida
Density (no. m-2)
650
600
550
500
450
400
350
300
250
200
150
100
50
0
**
*
*
*
open
forest
Clark
Copper Sandy
Lawrence
Picketts Mill
Westbrook
All Sites

Fig. 2.4


Insectivorous Cyprinidae
Campostoma oligolepis
Centrarchidae
Hypentelium etowanum
Percidae & Cottidae
Fundulus stellifer
Density (no. m -2)
1.6
1.4
1.2
1
0.8
0.6
0.4
0.2
0
4
3.5
3
2.5
2
1.5
0.5
*
**
open
forest
Riffle
Pool
Total

Appendix 2.1. Mean and standard deviations of percent composition of aquatic and terrestrial prey items for fish species, exclusive of sand and plant items. Percent empty is the percent of individuals that had ≥ 95% of their guts empty. Percent sand and percent plant are means of visual estimates of the amount within fish guts. Results for comparison between open and forested reaches (paired *t*-test for means, df = 4). Bold indicates < 0.05.

	Lepomis auritus (n = 5 sites)						*Cyprinella callistia*[a] (n = 3 sites)					
	Open		Forest				Open		Forest			
	Mean	StDev	Mean	StDev	t	*p*	Mean	StDev	Mean	StDev	t	*p*
Length (mm)	**91.02**	**3.75**	**96.69**	**2.52**	**4.98**	**0.004**	58.07	4.61	63.80	11.43	1.26	0.168
Weight (g)	35.17	22.22	28.78	4.21	0.75	0.247	2.97	0.76	4.23	2.13	1.50	0.137
Gut contents:												
% empty	0.11	0.14	0.13	0.14	0.30	0.391	**0.33**	**0.58**	**0.12**	**0.04**	**2.00**	**0.037**
% sand	3.46	4.46	1.53	1.69	1.79	0.074	--	--	8.33	14.43	1.00	0.211
% plant	8.67	8.87	3.43	1.84	1.61	0.091	0.57	0.98	1.00	1.00	0.44	0.352
Aquatic	55.82	20.61	45.60	6.59	1.19	0.150	44.29	41.72	53.16	32.63	0.47	0.342
Limpet	--	--	--	--	--	--	--	--	--	--	--	--
Copepod	--	--	--	--	--	--	--	--	--	--	--	--
Gastropod	1.16	2.17	--	--	1.43	0.113	--	--	--	--	--	--
Corbicula	6.74	8.42	0.39	0.88	1.49	0.105	--	--	--	--	--	--
Fingernail clam	--	--	--	--	--	--	--	--	--	--	--	--
Oligochaeta	0.67	1.49	--	--	1.00	0.187	--	--	--	--	--	--
Crayfish	2.59	3.22	2.31	2.52	0.06	0.479	--	--	--	--	--	--
Odonata	5.78	4.30	2.69	2.41	1.21	0.146	--	--	3.03	5.25	1.00	0.211
Ephemeroptera	8.72	15.76	6.43	5.70	0.10	0.462	2.86	4.95	19.39	20.03	1.32	0.158
Coleoptera	4.84	4.02	1.33	2.07	1.37	0.121	--	--	1.67	2.89	1.00	0.211
Hemiptera	--	--	0.89	1.99	1.00	0.187	--	--	--	--	--	--
Plecoptera	0.16	0.36	1.33	2.98	0.58	0.297	--	--	--	--	--	--
Trichoptera	5.85	8.64	7.05	3.75	0.93	0.202	33.81	29.29	24.37	18.66	0.15	0.446
Diptera	15.76	13.61	22.94	10.41	1.11	0.165	4.76	8.25	4.70	4.55	0.52	0.326
Dipteran pupae	**3.56**	**2.16**	**0.24**	**0.53**	**3.30**	**0.015**	2.86	4.95	--	--	1.00	0.211
Terrestrial	44.18	20.61	54.40	6.59	1.19	0.150	22.38	25.41	46.84	32.63	1.81	0.106
Arachnida	**3.38**	**3.00**	**5.14**	**2.89**	**2.59**	**0.030**	--	--	--	--	--	--
Diplopoda	--	--	--	--	--	--	--	--	1.67	2.89	1.00	0.211
Mite	--	--	--	--	--	--	--	--	1.96	3.40	1.00	0.211
Orthoptera	1.89	2.29	0.71	1.58	1.02	0.182	--	--	--	--	--	--
Thysanoptera	0.36	0.81	--	--	1.00	0.187	--	--	--	--	--	--
Odonata	0.67	1.49	2.22	4.97	0.37	0.366	--	--	--	--	--	--
Ephemeroptera	--	--	1.18	1.75	1.59	0.093	--	--	--	--	--	--
Coleoptera	6.13	3.21	7.80	5.71	0.48	0.330	--	--	1.96	3.40	1.00	0.211
Neuroptera	1.00	2.24	0.74	1.66	1.00	0.187	--	--	--	--	--	--
Hemiptera	2.09	3.21	4.06	2.34	1.62	0.090	--	--	--	--	--	--
Homoptera	2.50	4.75	4.37	3.68	1.36	0.123	--	--	--	--	--	--
Trichoptera	1.21	1.42	1.07	1.05	0.05	0.481	1.90	3.30	3.33	5.77	0.16	0.442
Hymenoptera	13.81	10.24	11.80	6.98	0.20	0.426	**0.95**	**1.65**	**19.20**	**9.75**	**4.16**	**0.027**
Diptera	11.13	9.27	12.15	4.25	0.64	0.278	19.52	26.74	18.73	20.84	0.02	0.493
Lepidoptera	--	--	3.16	4.89	1.57	0.095	--	--	--	--		

[a] *Notropis xaenocephalus* used instead of *Cyprinella callistia* at one site

Appendix 2.1. Continued.

Percina nigrofasciata (n = 4 sites)						*Hypentelium etowanum* (n = 5 sites)					
Open		Forest				Open		Forest			
Mean	StDev	Mean	StDev	t	*p*	Mean	StDev	Mean	StDev	t	*p*
64.34	10.17	63.96	6.59	0.10	0.463	114.70	3.95	130.43	16.74	2.10	0.052
3.79	1.70	3.04	1.00	1.27	0.148	26.97	4.00	38.40	13.55	1.91	0.064
0.11	0.13	--	--	1.73	0.091	0.10	0.09	0.20	0.18	0.64	0.279
--	--	--	--	--	--	17.50	14.58	6.83	3.35	1.30	0.131
--	--	--	--	--	--	--	--	--	--	--	--
--	--	--	--	--	--	98.12	1.90	99.64	0.34	1.29	0.133
5.29	6.89	23.36	22.40	1.43	0.124	--	--	--	--	--	--
--	--	0.47	0.94	1.00	0.196	--	--	--	--	--	--
--	--	--	--	--	--	--	--	--	--	--	--
--	--	--	--	--	--	2.82	3.60	6.98	13.39	0.44	0.340
--	--	--	--	--	--	17.00	31.45	12.24	27.37	1.37	0.121
--	--	--	--	--	--	--	--	0.11	0.25	1.00	0.187
1.79	3.57	--	--	1.00	0.196	--	--	--	--	--	--
1.15	2.31	--	--	1.00	0.196	0.90	1.39	0.22	0.50	1.57	0.096
31.50	31.28	15.33	10.99	0.72	0.260	0.72	1.38	4.76	8.96	0.80	0.235
0.42	0.85	0.53	1.06	0.07	0.474	0.14	0.32	0.07	0.17	0.21	0.424
--	--	--	--	--	--	0.04	0.08	--	--	1.00	0.187
--	--	--	--	--	--	--	--	--	--	--	--
3.81	7.63	1.52	3.03	0.28	0.398	0.68	0.55	7.70	8.58	2.23	0.045
48.58	24.80	51.03	19.02	0.23	0.416	61.47	24.22	53.64	27.94	0.53	0.313
7.46	4.41	7.77	4.56	0.21	0.424	14.36	11.71	13.91	5.57	0.19	0.430
--	--	--	--	--	--	1.88	1.90	0.36	0.34	1.29	0.133
--	--	--	--	--	--	--	--	--	--	--	--
--	--	--	--	--	--	--	--	--	--	--	--
--	--	--	--	--	--	1.20	1.35	0.32	0.32	1.15	0.157
--	--	--	--	--	--	--	--	--	--		
--	--	--	--	--	--	--	--	--	--		
--	--	--	--	--	--	--	--	--	--		
--	--	--	--	--	--	--	--	--	--		
--	--	--	--	--	--	--	--	--	--		
--	--	--	--	--	--	--	--	--	--		
--	--	--	--	--	--	--	--	--	--		
--	--	--	--	--	--	0.07	0.15	--	--	1.00	0.187
--	--	--	--	--	--	--	--	--	--	--	--
--	--	--	--	--	--	0.22	0.35	--	--	1.57	0.096
--	--	--	--	--	--	0.39	0.88	0.04	0.08	0.62	0.284
--	--	--	--	--	--	--	--	--	--	--	--

CHAPTER 3

RIPARIAN INFLUENCES ON STREAM FISH ASSEMBLAGE STRUCTURE IN URBANIZING LANDSCAPES

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[1] Roy, A. H., B. J. Freeman, and M. C. Freeman. To be submitted to *Landscape Ecology*.

Abstract

Upstream urbanization can have an overwhelming effect on downstream aquatic biota. We assessed the degree to which riparian forests at a reach-scale and along the upstream drainage network can mitigate the negative effects of catchment urbanization on stream fish assemblages. Data from 66 small tributaries of the Etowah River, Georgia, sampled for fishes between 1992 and 2003 were analyzed to evaluate relations between fish assemblage integrity and land cover assessed at multiple spatial scales. Land use/cover from satellite imagery was used to calculate % forest, urban, and agriculture within 30 m and 100 m buffers at 200 m, 1 km and the entire network upstream of the sampling location. Catchment and riparian land cover within the upstream network were highly correlated, so we were unable to distinguish between these variables. Endemic and sensitive species richness and abundance were best predicted by variables reflecting forest and urban land uses within the catchment. Cosmopolitan and tolerant species were positively related to % agriculture and negatively related to % forest at the local (200 m and 1 km) riparian scale. Sites with < 70 % forest in the riparian area along the upstream network had virtually no sensitive species, whereas streams with < 60% forest within the 1 km riparian area had consistently high abundances of tolerant species. After accounting for variation in catchment-scale land cover, local forest buffers were still important in explaining abundances of cosmopolitan and tolerant fishes. Catchment-wide development seemed to be an initial filter of fish species, with losses of sensitive species at high levels of urbanization, whereas local scale % forest cover secondarily affected abundances of tolerant species. These results support other studies with basin-wide ranges of urban and agriculture land cover, suggesting that catchment land cover is an important driver of biotic assemblages, and riparian forests are not sufficient to protect stream ecosystems. Maintenance of forest land cover within catchments and the riparian

network via regional land use planning and enforcement of riparian buffer ordinances are necessary to maintain high-quality fish assemblages.

Introduction

Landscapes are being developed and managed to meet human needs, subsequently altering stream hydrology, water quality, geomorphology, and biotic integrity. Changes in stream biotic integrity have been documented for agricultural (Roth et al. 1996, Lammert and Allan 1999), silvicultural (Davies and Nelson 1994, Stevens and Cummins 1999), and urban (Wang et al. 1997 & 2001, Walters et al. 2003) land cover disturbances in the catchment. The location of development within the landscape, such as the relative distance from stream channels, should play an important role in determining the in-stream effects of differences in land cover on stream ecosystems. For example, many studies have suggested that forested riparian areas adjacent to streams have a disproportional influence on stream biota (Steedman 1988, May et al. 1997, Lee et al. 2001). Further, the scale of disturbance (e.g. local vs. upstream) may also influence relations between land cover and in-stream communities (Roth et al. 1996, Lammert and Allan 1999, Wang et al. 2001).

Several previous studies have examined relations between biota and land cover assessed at multiple scales; however, results are conflicting in terms of the relative importance of catchment vs. riparian areas for driving differences in fish assemblages (Table 3.1). For example, some studies have found that basin land cover variables are the best indicator of fish assemblages (Roth et al. 1996, Snyder et al. 2003), while others have indicated that riparian and local scale land cover are most correlated with fish assemblages (Lammert and Allan 1999, Van Sickle et al. 2004). Many have proposed reasons why these results differ, including: a) resolution and age of land cover data, b) riparian scales (width and length) measured, and c) poor statistical resolution of intercorrelated variables. Further, some studies have been specifically designed to vary local riparian conditions (e.g. Jones et al. 1999, Lammert and Allan 1999, Stauffer et al. 2000, Lee et al. 2001), while others studies include a range in catchment land cover (Roth et al. 1996, Wang et al. 2001), creating statistical differences in ability to find significant effects at various spatial scales (Allan et al. 1997).

For studies that investigate the influence of land cover at multiple spatial scales, we expect that the type and range of dominant land cover will influence the strength of relations with fish assemblages. Most of the studies examining the relative influence of catchment and riparian land cover at multiple spatial scales have occurred in agricultural landscapes, but a few have addressed this question across a range of urban land cover. In one study in an urbanizing watershed, Steedman (1988) found that % forest within a 20 m wide riparian area along the entire network was positively related to a fish index of biotic integrity (IBI); however, % forest and % urban land cover in the catchment were also correlated with fish IBI. A study by Wang et al. (2001) emphasized the importance of catchment impervious within a 3.2 km radius of the sample location and within riparian areas, but they also found relations between fish richness, IBI, and diversity with land cover at the basin scale. In an analysis of fish databases within the mid-Atlantic region, Strayer et al. (2003) also examined sites across a range in % urban land cover; however, pasture and cultivated land (rather then urban land) within the basin and riparian corridor were most related to fish richness. Although these studies have touted the importance of forested riparian areas as being critical for protecting stream fish assemblages, the correlation between basin and riparian land cover leads to inconclusive results about the importance of forested riparian areas in urbanizing stream ecosystems.

We hypothesize that streams in urban landscapes are overwhelmed by upstream disturbances, and that forested riparian patches do not drive differences in aquatic fauna. Our previous research has shown that local forest canopies may not strongly affect fish assemblages in suburban streams (Roy et al. *in prep*), so in this study we asked whether cumulative effects of riparian deforestation along stream networks are important in driving fish assemblage structure. First, we asked which catchment and reach-scale riparian variables are related to fish assemblages and what scale of riparian variables are best predictors of fish assemblages. We further analyzed the data to ask whether the ability of forested buffers to mitigate effects of upland disturbance on fish assemblages is dependent on amount of urban development within the landscape.

Methods

Study sites

The Etowah River basin is a 4823 km^2 catchment in north-central Georgia which harbors high species diversity (Burkhead et al. 1997); however, increased urbanization in the basin in the last two decades has impacted biotic integrity within small streams (Roy et al. 2003b, Walters et al. 2003). Sites were selected for this study from a database of 901 fish collections made within the Etowah River basin from 1992 to 2003. From this database, we selected non-nested, small streams (5-36 km^2) that had been quantitatively sampled for fishes. When multiple samples were taken from the same location, we selected the most recent sample for analysis. Finally, we excluded sites with impoundments within the 1 km reach upstream of the sample site to yield a final data set of 66 streams.

Land use/cover

Land use/cover (hereafter referred to as land cover) was quantified within the 30 m and 100 m riparian buffers on each side of the stream (i.e. 60 m and 200 m corridors) and the entire catchment (inclusive of the riparian area) using ArcView© 4.0 Geographic Information Systems (GIS). A drainage network created from Digital Elevation Models (DEMs), which was similar to a 1:24,000 scale stream network, was used to create buffers around a) the lower 200 m reach at the sampling location, b) the 1 km reach upstream of the site, and c) the entire drainage network upstream of the site.

Landsat Thematic Mapper (TM) satellite imagery for 1992, 1998, and 2001 (17 land cover classes) were used to calculate percentages of land cover categories within the catchment and riparian areas upstream of sample sites. Land cover was categorized as urban (high density and low density urban), agriculture (cultivated/exposed land and cropland/grassland), and forest (evergreen, deciduous, mixed, and forested wetlands). The most recent prior imagery date corresponding to fish collections was used (e.g. 1992 for collections 1992-1997, 1998 for collections 1998-2000, and 2001 for collections 2001-2003) for each site. Percent impervious cover was calculated for each sub-catchment from a classified dataset created by the Georgia Land Use Trends Project (Natural Resources Spatial Analysis Laboratory, Institute of Ecology, University of Georgia, Athens, GA, USA).

Road crossings were used as a surrogate for riparian gaps. Crossings were mapped by overlaying 1999 Georgia Department of Transportation road coverage with the drainage network. Number of road crossings upstream of sample sites were divided by stream length (km) of the network.

Fish assemblages

Fishes were collected in late spring through early autumn (May-October) during low flow conditions using a backpack electroshocker. Although protocol differed for fish collections, all samples were complete (i.e. included entire fish assemblage) and quantitative. Rare species (those found only at 1 site, n = 19) were omitted from analyses. Estimated richness was calculated from species counts using a first-order jackknife which uses the number of species with only 1 individual collected to estimate species not observed (Burnham and Overton 1979, Nichols et al. 1998). Fish assemblage structure was evaluated based on estimated richness (no. species) and abundance (no. individuals) of endemic vs. cosmopolitan species, and sensitive vs. tolerant species (Appendix 3.1). Endemic species are fishes primarily limited to the Coosa River drainage (which includes the Etowah River), and these were expected to decrease with urbanization (Walters et al. 2003). Conversely, cosmopolitan species were defined as those fishes native to at least 10 major drainages, and were expected to increase with urbanization (Walters et al. 2003). We also examined the ratio of endemics to cosmopolitans (based on species richness and abundance), which should reflect a homogenization of fish assemblages coincident with loss of endemic species (Scott and Helfman 2001, Walters et al. 2003). Sensitive species were those species found in the Etowah River basin that were expected to be sensitive to disturbance (due to specific life history or habitat requirements), and exhibited a negative response to increased urban land cover (Seth J. Wenger, University of Georgia, *unpublished data*). Cherokee darters, *Etheostoma scotti*, were excluded from the sensitive species group because they did not respond to urban land cover. Tolerant species were habitat generalists, capable of completing their life cycle in lentic environments (Travnichek et al. 1995), and were classified using Etnier and Starnes (1993) and Mettee et al. (1993).

Data analysis

All variables were tested for normality using Shapiro-Wilk goodness of fit test and transformed when necessary. All fish abundance metrics were transformed using log(x+1) and percentage variables were transformed using arcsin(sqrt(%/100)).

We used Pearson's correlation coefficient (*r*) to analyze relations between catchment land cover and riparian land cover variables assessed at three spatial scales. Although these are not independent variables (i.e. smaller spatial units are incorporated within larger spatial units), we used correlation as a descriptive measure of the relative strengths of relationships. Correlation analysis was also used to determine relations between land cover variables and fish assemblages. We further compared slopes of regression models to evaluate what change in fish assemblages results from incremental changes in land cover (following Wang et al. 2001).

Since catchment land cover may provide an overriding influence on fish assemblages, we wanted to evaluate the importance of riparian land cover after accounting for catchment land cover. To do this, we created least-squares linear regression models using catchment land cover alone, catchment land cover plus % riparian forest, and catchment land cover plus an interaction between catchment land cover and % riparian forest to predict fish assemblage variables. We quantified riparian forest cover using % forest in the 30 m buffer for the 1 km reach, because analyses showed that this variable was not highly correlated with catchment land cover variables, and the 1 km length encompassed a reasonable reach experienced by a fish (vs. 200 m reach). For endemic and sensitive species, models with forest and urban land cover were included. Because % agriculture cover was found to be important in explaining cosmopolitan and lentic tolerant fishes, model sets for these fishes included forest, urban, and agriculture.

We used Akaike's Information Criterion, adjusted for small sample size (AIC_c), to assess fit of candidate models, with lowest AIC_c indicating the best-supported model for predicting each fish assemblage metric within model sets (Burnham and Anderson 2002). In contrast to hypothesis testing, this information-theoretic approach avoids overfitting models by identifying scientifically appropriate models *a priori*, and evaluates the relative support for each model within a set of plausible models based on model likelihood and parameters measured (Burnham and Anderson 2002). Akaike weights (w_i) were computed as $w_i = \exp(-½\Delta_i)/\sum\exp(-½\Delta_i)$, where Δ_i equals the difference in AIC_c for each model compared to the best-supported model (i.e.$\Delta_i = 0$ for best-supported model) and the denominator is a sum of $\exp(-½\Delta_i)$ for all models in the set. We used Akaike weights (which vary from 0 to 1 with the best-fitting model having the highest weight) to measure the weight of evidence for each model given the data (Burnham and Anderson 2002). We summed weights across models that included each catchment land cover variable to estimate the relative importance of forest, urban, and agriculture land cover for predicting fish assemblage metrics. Although adjusted R^2 values provide useful information about the variance explained in a model, we feel that AIC_c is the best approach to compare candidate models and determine the best-supporting model relative to the model set. Because AIC_c assumes that a favorable model exists within the model set, we did not compare candidate models within a group if all the models exhibited poor fit (i.e. low adjusted R^2).

We were interested in determining whether sites with low or high impervious cover responded differently to % forest cover in the riparian area. We divided sites into less urbanized (<15 % urban land cover) and more urbanized (>15 % urban land cover) corresponding to literature reported threshold values (Wang et al. 1997, Paul and Meyer 2001) and evaluated differences in correlations with fish variables between the two categories. All analyses were run using JMP Version 4 (SAS Institute Inc., Cary, NC, USA).

Results

Land cover variables

Sites exhibited a range of forest (28.8-96.5%), urban (0.5-65.2%), and agriculture (0.1-31.2%) land cover (Table 3.2). On average % forest land cover was higher within the 30 m riparian buffer compared to the entire catchment for all spatial scales. Mean % urban land cover was lower in riparian areas vs. catchment, while mean % agriculture was higher in the riparian areas vs. catchment at the 200 m and 1 km reach scales (Table 3.2).

There were many significant correlations among land cover variables at the various spatial scales (Table 3.3). The highest correlations were between catchment land cover and riparian land cover for the upstream network (vs. 200 and 1 km reaches), with the 100 m width buffer being most similar to catchment land cover. Because these variables were so tightly correlated (e.g. for % forest in catchment vs. % forest in riparian network for 30 m and 100 m buffers, r = 0.95 and 0.98, respectively), we were unable to distinguish between these catchment and riparian network variables. Thus, throughout this paper we selected either catchment or riparian network variables to relate to fish assemblage variables, and we did not distinguish between these variables in interpretations. Percent forest and % urban were also highly negatively correlated with each other at the largest spatial scales, with r-values ranging from -0.84 to -0.89. Road crossings had weaker relations with % forest and % urban land cover (r = -0.49 and 0.59, respectively), so this variable was considered somewhat independent of overall land cover (Table 3.3).

Correlations between land cover and fish assemblages

Between 6 and 22 fish species were found at each site, with an average abundance of 240 individuals (Table 3.2). *Hypentelium etowanum* (Alabama hogsucker), *Campostoma oligolepis* (largescale stoneroller), *Lepomis macrochirus* (bluegill sunfish), *Lepomis auritus* (redbreast sunfish), and *Percina nigrofasciata* (blackbanded darter) were the most commonly found species, present at > 80% of the sites, and also the species with the highest average abundances. *Onchorynchus mykiss* (rainbow trout) was the only non-native fish found at the sites, but it was eliminated from analyses since it was only found at one site. Sites exhibited a range in endemic (0-7), cosmopolitan (2-18), sensitive (0-8), and tolerant (0-12) species richness, and corresponding ranges in abundances of the various groups of fish species (Table 3.2).

Fish assemblages were related to all land cover variables, with the greatest number of strong relations with % impervious surface in catchment, % forest in riparian network, and % urban in riparian network (Table 3.4). Endemic and sensitive species richness and abundance were best predicted by variables reflecting forest and urban land uses in the catchment. In addition to relationships with forest and urban land uses in the catchment, cosmopolitan and tolerant species richness and abundance were positively related to % agriculture and negatively related to % forest at the local riparian scale (Table 3.4). Regression slopes revealed that unit changes in % riparian forest variables result in greater changes in endemic and sensitive species richness than in cosmopolitan and tolerant species richness (Fig. 3.1A). In terms of abundance, regression slopes were consistently highest with % forest in network (vs. other scales) for all fish assemblage groups, indicating that unit changes in % forest at the largest spatial scale (network) resulted in the biggest changes in abundance (Fig 3.1B).

Importance of riparian forests based on catchment land cover

Endemic and sensitive fish assemblage variables were significantly related to % forest and % urban land cover in the catchment, so we asked whether riparian land cover could explain additional variation in the fish assemblages, as an additive variable or as an interaction with catchment land cover. The best-supported models were consistently single variable models with % forest or % urban land cover (Table 3.5). Combined weights suggested that % forest and % urban land cover were equally good at predicting differences in endemic and sensitive species across sites (Table 3.5). For cosmopolitan and lentic tolerant species, adding % riparian forests (1 km scale) to models with catchment land cover consistently improved model fit, and the best-supported model in each case was % forest in catchment plus % forest in riparian area (Table 3.6). Abundances of cosmopolitan and lentic tolerant species were best predicted by reduced amounts of forest cover (vs. urban or agriculture).

We hypothesized that sites with more urbanized catchments (>15 % urban land cover) would have weaker relations with % forest in riparian compared to less urbanized catchments (< 15 % urban land cover). Plots of abundance of endemic and sensitive species with % forest in riparian network revealed an overall trend of increased abundance with higher % forest in riparian network; however, the response with endemic species was not strong. There were no clear trends when comparing more urbanized vs. less urbanized catchments (Fig. 3.2). Streams with < *ca*. 70 % riparian forest cover, most which had > 15 % urban land cover in the catchment, had low abundances of sensitive fishes. Variables other than riparian forest cover influence abundance of sensitive species above this 70 % riparian forest threshold (Fig. 3.2). Abundances of cosmopolitan and tolerant species were consistently high at sites with < ca. 60 % riparian forest cover, whereas sites with > 60 % riparian forest cover exhibited a range in abundances of these fish species (Fig. 3.3). More urbanized and less urbanized catchments showed similar responses, indicating that catchment urbanization did not appear to affect the response of cosmopolitan and lentic tolerant species to local riparian forest cover (Fig. 3.3).

Discussion

What land cover variables best predict fish assemblages?

The number of endemic and sensitive species found in a stream was best predicted by % forest and % urban in the catchment, % impervious surface in the catchment, and the number of road crossings (endemic richness only). Interestingly, the number of cosmopolitan and tolerant species was not related to catchment urbanization, but was instead related to % forest and agriculture land cover at the local, 1 km scale (Table 3.4). Development in the catchment seems to be an initial filter of fish species richness, with losses of sensitive species associated with increasing amounts of urban disturbance. Local scale land cover created a secondary filter, but only for tolerant species, which increased with loss of riparian forests and increases of agriculture land cover. This concept of "landscape filters" has been proposed for stream ecosystems (Poff 1997), and seems to fit well to these fish assemblage measures.

Abundances of fish assemblage groups also responded to various spatial scales of land cover, and suggest potential land cover thresholds for fish assemblages. Abundances of sensitive species exhibited a striking response to % forest in the riparian network, with virtually no sensitive species found at sites with < *ca.* 70 % forest (Fig 3.2). Since % forest in the riparian network is also correlated with % urban at that scale (r = -0.85), we expect that the loss of forest and a concomitant increase in % urban is reducing abundances of sensitive fishes in these streams. We also found a similar threshold of ~60 % forest in the local riparian buffer for cosmopolitan and tolerant species; at sites below this threshold abundances of cosmopolitan and tolerant species were consistently high (Fig. 3.3). Regardless of upland % urban land cover, streamside riparian forests can significantly influence fish assemblages. Although sites with high amounts of forest cover may have a wide range of fish assemblage integrity, these results support other studies indicating that loss of forest cover can lead to proportional shifts in the abundances of sensitive and tolerant species (Scott and Helfman 2001, Walters et al. 2003).

Studies that incorporate a range of catchment land cover often demonstrate significant relationships between land cover and stream quality (see Table 3.1). This study had the greatest differences in % forest and % urban land cover (vs. smaller ranges in % agriculture) across sites, and these variables were most important in predicting fish assemblages. Conversely, streams with one dominant land cover (and a very small range) are more likely to observe shifts associated with minor changes in riparian land cover if tested at the appropriate scale. For example, Stauffer et al (2000) and Lee et al. (2001) found that small increases in local forest cover within the riparian area resulted in shifts toward higher fish assemblage integrity in catchments that were dominated (88-100%) by agricultural land cover. Similarly, Jones et al. (1999) documented changes in fish assemblages with local riparian deforestation in primarily forested (96-100%) watersheds. In this study we found minimal evidence that local riparian forests were driving fish assemblages, possibly because the streams lie within landscapes that have multiple land uses, and because there were large differences in basin land cover across sites. Taken together, these studies suggest that landscape context is critical to understanding the extent of influence of riparian areas on stream ecosystems (Naiman and Decamps 1997).

Does urbanization influence the relative importance of riparian forests?

Because of the high range of urbanization and forest cover across our study sites, we were able to specifically address whether riparian forests were able to influence fish assemblages after accounting for differences in catchment land cover across sites. Cosmopolitan and tolerant species abundances were the only fish variables for which riparian forests significantly improved their predictive ability, and these were both negatively influenced by the % forest in 1 km riparian area. This suggests that local patches of forest, although not sufficient at maintaining sensitive species, are important for preventing high abundances of tolerant species.

We initially hypothesized that streams in urban settings would not respond to differences in riparian forest cover because catchment-level processes would overwhelm local land cover and reduce assemblage integrity. An alternative hypothesis is that streams in predominantly urban settings may respond *more* to riparian deforestation than those in less urban systems if fish species are not already lost, because these fishes would be more vulnerable to change. In this study, levels of urbanization did not seem to affect responses of fishes to riparian forest cover (Figs. 3.2 and 3.3); however, the tight correlation between urban and forest cover complicates these analyses. Further investigation of this question with specific attention to aspects of urban land cover that might impair streams (e.g. altered hydrology) would better tease apart potential mechanisms of fish response to riparian deforestation and conditions influencing benefits of riparian forests for protecting fish assemblages.

Potential problems with spatial scale analyses

We were unable to detect differences in the relative importance of catchment vs. riparian land cover over the entire upstream network, because these variables were highly correlated within land cover classes. Other studies have also reported significant correlations among land cover variables, especially at the scale of the entire stream network (Lammert and Allan 1999, Wang et al. 2001). We suspect that this problem of multicollinearity exists in most watersheds and equally restricts the ability to interpret differences in predictive models among landscape variables. Although these observational studies are limited in their ability to distinguish among key variables, large-scale manipulative studies in already disturbed landscapes are unlikely to occur. Van Sickle et al. (2004) suggest that modeling alternative land cover scenarios may be useful at distinguishing differences between scales; however, their data-driven modeling approach was not sensitive enough to detect differences between riparian and catchment land cover. The authors suggest that making predictive models based on “expert judgment” may offer discriminatory power between these variables (Van Sickle et al. 2004). We contend that such models may not be useful, since not enough is known about what % riparian forest is necessary to provide sufficient functions (e.g. organic matter inputs, bank stability) to prevent loss of biotic integrity.

Satellite imagery was used to characterize land cover at the multiple spatial scales because these data were available at multiple dates corresponding to fish sampling; however, the results may have been influenced by the poor resolution of satellite imagery (30 m pixels), which may be especially biased at small spatial scales (i.e. narrow buffers and local reaches). Many studies have simultaneously varied the land cover data source in order to best characterize land cover within each scale. For example, field transects are sometimes used to characterize riparian areas at the local (100-200 m) scale; aerial photography is often used to assess reach (1-2 km) scale land cover; and satellite imagery or other digital land cover databases have been used for catchment scale land cover (see Table 1). Lattin et al. (2004) tested whether these differences in sources of land cover (aerial photography vs. satellite imagery) affected the accuracy and strength of relationships between land cover and biota. Although they found slightly stronger relationships with aerial photography, assessment at multiple scales was more important than the source of imagery for detecting associations between land cover and fish assemblage integrity. The authors suggested that incorporating different imagery sources may mask changes in land cover over time and across spatial scales, and choice of imagery should depend on the question being addressed (Lattin et al. 2004).

Management implications

Riparian buffers have been used for managing non-point source disturbances in the United States since the late 1960s (Calhoun 1988), with the primary goal of isolating upland disturbances from aquatic resources (Lee et al. 2004). These regulations have underlying implications that stream conditions in the catchment can be mitigated by protecting land adjacent to streams (Allan et al. 1997, Harding et al. 1998). However, research continues to suggest not only that catchment land cover is an important driver of biotic assemblages, but also that riparian forests are not sufficient for protecting stream ecosystems (Allan and Johnson 1997, Harding et al. 1998).

These results emphasize the importance of regional land use planning in protecting stream ecosystems. Regional planning is often complicated by multiple jurisdictions transecting watershed boundaries. In the Etowah River basin, scientists, lawyers, and stakeholders are in the process of developing a regional aquatic Habitat Conservation Plan (HCP) for protection of imperiled fish species. Developers must obtain an incidental take permit to build in areas that may cause harm to the species or their habitat, and to do so must create an HCP stating that they are minimizing impacts caused by the activity and incorporating some method of mitigation for their incidental take (Moser 2000). One of the primary advantages of a regional HCP is protection within the entire range of the species, rather than restoration of small habitat patches (McKinney and Murphy 1996). Because these fishes are ultimately impaired by catchment-level factors, regional land use planning within this HCP offers promise of successful protection of these fish assemblages.

Although forested riparian buffers may not be sufficient for protecting fish assemblages, these results do not imply that riparian forests are unimportant. Many studies have indicated the important functions of riparian forests in for stream ecosystems (see reviews Gregory et al. 1991, Sweeney 1992, Naiman and Decamps 1997, Lowrance 1998, Pusey and Arthington 2003). Based on the number of potential linkages between riparian alteration and fish assemblages (Pusey and Arthington 2003), it is not surprising that quantification of local, riparian conditions is sometimes found to be related to fish assemblage integrity (Meador and Goldstein 2003). Since riparian forests provide certain functions such as temperature regulation and organic matter input that are essential for maintenance of stream integrity, complete removal of riparian forests would be detrimental to stream ecosystems. In this study, we found that reductions of reach-scale (200 m, 1 km) riparian forests increased abundances of tolerant species, suggesting that forest cover may also control resources (e.g. autochthonous production) in these streams.

Results from this study and other studies suggest that human alteration affects stream processes at multiple spatial scales. In addition to % land cover within catchments and riparian areas, the continuity of riparian forests (Stewart et al. 2001) and historic land use in the catchment (Harding et al. 1998) likely also influence fish assemblages. Regardless of what riparian variables are most important, these results lead to similar recommendations for stream protection. High amounts of private land ownership coupled with the inability to require retrofit of riparian buffers limit complete protection of riparian buffers and challenge policymakers to adapt regulations for the existing mosaic of land cover within basin and riparian areas. Currently, Georgia’s stream buffers are protected by the Erosion and Sedimentation Control Act of 1975 (O.C.G.A. 12-7) which requires a minimum of 25 foot riparian buffers on all streams. However, riparian areas exhibited an average 10.3% decrease in forest cover and 8.5% increase in urban land cover between 1973 and 1997 (Roy et al. 2003b), suggesting that these regulations and/or current enforcement of these regulations have not been effective at protecting stream ecosystems from continued loss of forest cover and subsequent declines in fish assemblage integrity. Efforts to enforce stricter buffer regulations on future developments to prevent cumulative loss of > 70 % riparian forest within a 30 m buffer would offer the best protection of stream fishes if associated with regional planning to minimize basin-wide disturbances.

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Table 3.1. Selected studies relating riparian land cover to fish assemblage integrity (listed in chronological order).

Citation	# Sites (basin area)	Range of land cover in catchments	Land cover data type	Riparian scales[a]	Correlations between basin & riparian land cover and fish assemblage
Steedman 1988	108 (2-895 km^2)	0-~25% forest[b] 0-~75% urban	topo maps	20 m wide riparian, % length	basin & riparain % forest v. IBI (+) basin % urban v. IBI (-)
Roth et al. 1996	23[c] (21-251 km^2)	10-25% forest 1-13% urban 36-84% agriculture	land cover database[d]--basin air photos--reach field transects--local	50, 125, & 250 m width 150 m, 1.5 km, & network length	basin % forest v. IBI (+) basin % urban & agriculture v. IBI (-) 50 m % forest network v. IBI (+)
Lammert & Allan 1999	18[b] (50-76 km^2)	14-29% forest 1-9% urban 21-73% agriculture	land cover database[d]	50 & 125 m width for network length and segment to next site (~1 km)	50 & 125 m forest v. IBI (+) 50 m agriculture v. IBI (-)
Stewart et al. 2001	38 (9-71 km^2)	1-34% forest 0-33% urban 19-88% agriculture	air photos	0-10, 10-20, & 20-30 width for network length forest gaps & length	basin % forest v. diversity (+) & richness (+) < 30 m & basin % forest v. % tolerant (-) mean gap length v. IBI (-) basin % urban & 20-30 m grassland v. fish density (-)
Wang et al. 2001	47 (10-101 km^2)	0-18% forest 3-97% urban 0-89% agriculture	land cover database[e]	<50, 50-100, >100 m width <1.6, 1.6-3.2, >3.2 km radius upstream	< 3.2 km impervious v. richness (-), IBI (-) & diversity (-) > 3.2 km land cover v. richness (-), IBI (-) & diversity (-)
Snyder et al. 2003	20 (3-85 km^2)	22-53% forest 0-28% urban 38-74% agriculture	satellite imagery--basin air photos--reach field transects--local	30 m for length 80x stream width 129 m for length 400x stream width	basin % urban & agriculture v. IBI (-)
Strayer et al. 2003	944 (0.2-22,278 km^2)	2-100% forest 0-93% urban 0-88% pasture	satellite imagery	135 m width for network length 300 m radius	basin & 135 m pasture & cultivated v. richness (-)
Lattin et al. 2004	23 (15-87 km^2)	~46% agriculture	air photos & satellite imagery	25, 50, 100, 150 m width reach, 1 km, 10 km, & network length	10 km & network land cover v. IBI
Roy et al. (this study)	66 (5-36 km^2)	29-97% forest 0-65% urban 0-31% agriculture	satellite imagery	50 & 100 m width 200 m, 1 km, & network length	basin & network % forest v. sensitive (+) & tolerant (-) basin & network % urban v. sensitive (-) & tolerant (+) 1 km % forest v. cosmopolitan (-) & tolerant (-)

[a] Widths reported as distance from centerline of stream; double values to obtain riparian corridor widths

[b] Percentages estimated from figures, unknown amount of agriculture land cover

[c] Some subcatchments spatially nested

[d] Land cover from 1978 Michigan Resource Inventory System updated with 1995 aerial photography; resolution ~ 25 m

[e] Land cover from digital land-use database developed from 1:4800 air photos; resolution 600-4000 m^2

Table 3.2. Land cover and fish assemblage summary data for the 66 study sites. Land cover for catchment and 30 m buffer for three scales of stream length (200 m, 1 km, network) at each site was calculated from *Landsat* TM imagery for closest previous date (1992, 1998, or 2001) prior to fish sampling. Abundance data are number of individuals collected, and richness data are estimated number of species using a first-order jackknife of single-pass collections (Nichols et al. 1998).

Land Cover			Fish Assemblage		
	Mean	Range		Mean	Range
Basin area (km^2)	13.1	(5.4-36.3)	Total richness	13.4	(6-22)
% Impervious (2001)[a]	9.2	(1.7-31.1)	Total abundance[b]	240.4	(46-1115)
Road crossings (no. km^{-1})	0.4	(0-1.39)	Endemic		
Catchment			Richness	2.6	(0-7)
% Forest[a]	65.0	(28.8-96.5)	Abundance[b]	58.4	(0-333)
% Urban[a]	17.4	(0.5-65.2)	Cosmopolitan		
% Agriculture[a]	9.6	(0.1-31.2)	Richness	8.0	(2-18)
Riparian, 200 m			Abundance[b]	110.0	(11-534)
% Forest[a]	69.5	(0-100)	Endemic:Cosmopolitan		
% Urban[a]	7.1	(0-100)	Richness	0.45	(0-3.5)
% Agriculture[a]	13.4	(0-100)	Abundance[b]	1.05	(0-10.5)
Riparian, 1 km			Sensitive		
% Forest[a]	76.9	(1.5-100)	Richness	1.9	(0-8)
% Urban[a]	5.3	(0-98.5)	Abundance[b]	19.7	(0-132)
% Agriculture[a]	10.6	(0-76.1)	Tolerant		
Riparian, network			Richness	4.1	(0-12)
% Forest[a]	76.1	(48.3-98.6)	Abundance[b]	54.0	(0-341)
% Urban[a]	10.7	(0-44.9)			
% Agriculture[a]	5.1	(0-17.8)			

[a] Transformed using arcsin(sqrt(x/100)) for analysis

[b] Transformed using log(x+1) for analysis

Table 3.3. Pearson's correlations (*r*) between catchment variables (road crossings per km stream length, % impervious, and catchment % land cover), and riparian % land cover for two buffer widths (30 m and 100 m) and three longitudinal scales (200 m, 1 km, network). Bold type indicates $r \geq$ 0.50.

	% Impervious	Road Crossings	Catchment Land Cover		
			% Forest	% Urban	% Agriculture
Road crossings	0.49	--	--	--	--
Catchment					
% Forest	**-0.85**	-0.49	--	--	--
% Urban	**0.89**	**0.59**	**-0.86**	--	--
% Agriculture	-0.11	-0.10	-0.20	-0.25	--
Riparian, 200 m					
% Forest, 100 m	-0.23	-0.21	0.36	-0.24	-0.32
% Urban, 100 m	0.46	0.49	**-0.51**	**0.56**	-0.09
% Agriculture, 100 m	-0.11	-0.14	0.00	-0.19	**0.51**
% Forest, 30 m	-0.10	-0.24	0.22	-0.18	-0.20
% Urban, 30 m	0.32	**0.50**	-0.35	0.41	-0.11
% Agriculture, 30 m	-0.11	-0.13	0.04	-0.16	0.38
Riparian, 1 km					
% Forest, 100 m	-0.37	-0.36	**0.51**	-0.36	-0.33
% Urban, 100 m	**0.61**	**0.59**	**-0.64**	**0.70**	-0.11
% Agriculture, 100 m	-0.12	-0.16	-0.01	-0.23	**0.61**
% Forest, 30 m	-0.18	-0.40	0.32	-0.22	-0.33
% Urban, 30 m	0.36	**0.60**	-0.41	0.46	-0.06
% Agriculture, 30 m	-0.15	-0.19	0.03	-0.22	**0.52**
Riparian, network					
% Forest, 100 m	**-0.86**	-0.49	**0.98**	**-0.87**	-0.14
% Urban, 100 m	**0.89**	**0.60**	**-0.85**	**0.99**	-0.26
% Agriculture, 100 m	-0.16	-0.10	-0.14	-0.29	**0.97**
% Forest, 30 m	**-0.83**	-0.48	**0.95**	**-0.84**	-0.17
% Urban, 30 m	**0.87**	**0.62**	**-0.84**	**0.98**	-0.23
% Agriculture, 30 m	-0.16	-0.14	-0.10	-0.30	**0.90**

Table 3.4. Pearson's correlations (*r*) between fish assemblage variables, catchment variables (road crossings per km stream length, % impervious), and 30 m riparian buffer % land cover for three longitudinal scales (200 m, 1 km, network). Bold type indicates $r \geq 0.24$.

	% Impervious	Road Crossings	Riparian Land Cover (30 m buffer)								
			% Forest			% Urban			% Agriculture		
			200 m	1 km	network	200 m	1 km	network	200 m	1 km	network
Endemic (E)											
Richness	**-0.45**	**-0.32**	-0.01	0.07	**0.42**	-0.06	-0.12	**-0.40**	0.10	0.01	0.01
Abundance	**-0.26**	-0.07	0.00	0.07	**0.27**	0.07	-0.01	-0.17	-0.12	-0.15	-0.17
Cosmopolitan (C)											
Richness	0.12	0.01	-0.18	**-0.28**	0.00	-0.04	-0.01	-0.08	**0.30**	**0.37**	0.16
Abundance	**0.36**	**0.25**	**-0.37**	**-0.47**	**-0.37**	0.19	**0.26**	**0.29**	0.20	**0.27**	**0.23**
E:C											
Richness	**-0.36**	**-0.27**	0.08	0.22	0.23	-0.13	-0.19	-0.20	-0.02	-0.10	0.01
Abundance	**-0.37**	-0.16	0.14	0.21	**0.36**	-0.03	-0.10	**-0.25**	-0.17	-0.21	**-0.21**
Sensitive											
Richness	**-0.39**	-0.21	-0.01	-0.04	**0.44**	-0.15	-0.17	**-0.43**	0.08	0.08	0.03
Abundance	**-0.50**	**-0.25**	-0.05	0.00	**0.48**	-0.07	-0.03	**-0.51**	0.12	0.09	0.08
Tolerant											
Richness	**0.34**	0.11	-0.19	**-0.29**	-0.23	0.06	0.14	0.17	0.20	**0.27**	0.12
Abundance	**0.39**	**0.24**	**-0.34**	**-0.42**	**-0.41**	0.22	**0.26**	**0.32**	0.18	**0.29**	0.17

Table 3.5. Multiple linear regression models predicting richness and abundance of endemic and sensitive fish assemblages with % forest ("forest") or % urban cover ("urban") in catchment, and % forest in 30 m riparian area at the 1 km reach ("riparian") based on 66 sites. Abundance of endemic species were excluded because models indicated poor fit. Adjusted R^2, differences in Akaike's Information Criterion from minimum (Δi), Akaike weights (w_i) of each model, and combined weights for models with forest and urban variables are reported. Bold type indicates two best-supported models.

	Adj. R^2	Δ_i	w_i	Combined w_i
Endemic Richness				
Forest	**0.15**	**0.00**	**0.35**	0.60
Forest + Riparian	0.14	2.15	0.12	
Forest + Forest*Riparian	0.15	1.95	0.13	
Urban	**0.13**	**0.74**	**0.24**	0.40
Urban + Riparian	0.11	2.99	0.08	
Urban + Urban*Riparian	0.11	2.99	0.08	
Sensitive Richness				
Forest	**0.20**	**0.00**	**0.24**	0.58
Forest + Riparian	0.24	0.66	0.17	
Forest + Forest*Riparian	0.24	0.61	0.17	
Urban	**0.20**	**0.11**	**0.22**	0.42
Urban + Riparian	0.19	1.67	0.10	
Urban + Urban*Riparian	0.21	1.80	0.10	
Sensitive Abundance				
Forest	**0.26**	**0.49**	**0.22**	0.49
Forest + Riparian	0.29	1.62	0.13	
Forest + Forest*Riparian	0.29	1.33	0.14	
Urban	**0.27**	**0.00**	**0.28**	0.51
Urban + Riparian	0.28	1.72	0.12	
Urban + Urban*Riparian	0.28	1.92	0.11	

Table 3.6. Multiple linear regression models predicting abundance of cosmopolitan and lentic tolerant fish assemblages with % forest ("forest"), % urban cover ("urban"), or % agriculture cover ("agriculture") in catchment, and % forest in 30 m riparian area at the 1 km reach ("riparian") based on 66 sites. Adjusted R^2, differences in Akaike's Information Criterion from minimum (Δi), Akaike weights (w_i) of each model, and combined weights for models with forest, urban, and agriculture variables are reported. Bold type indicates best-supported model within each set of models with forest, urban, and agriculture variables.

	Adj. R^2	Δ_i	w_i	Combined w_i
Cosmopolitan Abundance				
Forest	0.15	2.66	0.08	0.58
Forest + Riparian	**0.27**	**0.00**	**0.28**	
Forest + Forest*Riparian	0.26	0.47	0.22	
Urban	0.07	5.15	0.02	0.24
Urban + Riparian	**0.24**	**1.32**	**0.15**	
Urban + Urban*Riparian	0.20	2.63	0.08	
Agriculture	0.06	5.51	0.02	0.17
Agriculture + Riparian	**0.22**	**2.15**	**0.10**	
Agriculture + Agriculture*Riparian	0.18	3.27	0.06	
Lentic Tolerant Abundance				
Forest	0.16	1.16	0.14	0.60
Forest + Riparian	**0.24**	**0.00**	**0.25**	
Forest + Forest*Riparian	0.23	0.34	0.21	
Urban	0.11	2.92	0.06	0.30
Urban + Riparian	**0.22**	**0.81**	**0.17**	
Urban + Urban*Riparian	0.18	2.41	0.07	
Agriculture	0.02	5.49	0.02	0.11
Agriculture + Riparian	**0.16**	**3.10**	**0.05**	
Agriculture + Agriculture*Riparian	0.13	3.83	0.04	

Figure Legends

Fig 3.1. Slopes of significant linear regression models ($p < 0.05$) for a) richness and b) abundance of fish assemblage variables as predicted by % forest land cover within a 30 m buffer at three upstream spatial scales (200 m, 1 km, and network).

Fig. 3.2. Relationship between endemic and sensitive species abundance (no. individuals, log transformed) and % forest in 30 m riparian buffer for upstream drainage extent. Sites coded as <15 % (○) or > 15 % (●) urban land cover in catchment.

Fig. 3.3. Relationship between cosmopolitan and lentic tolerant species abundance (no. individuals, log transformed) and % forest in 30 m riparian buffer for upstream drainage extent. Sites coded as <15 % (○) or > 15 % (●) urban land cover in catchment.

Fig. 3.1

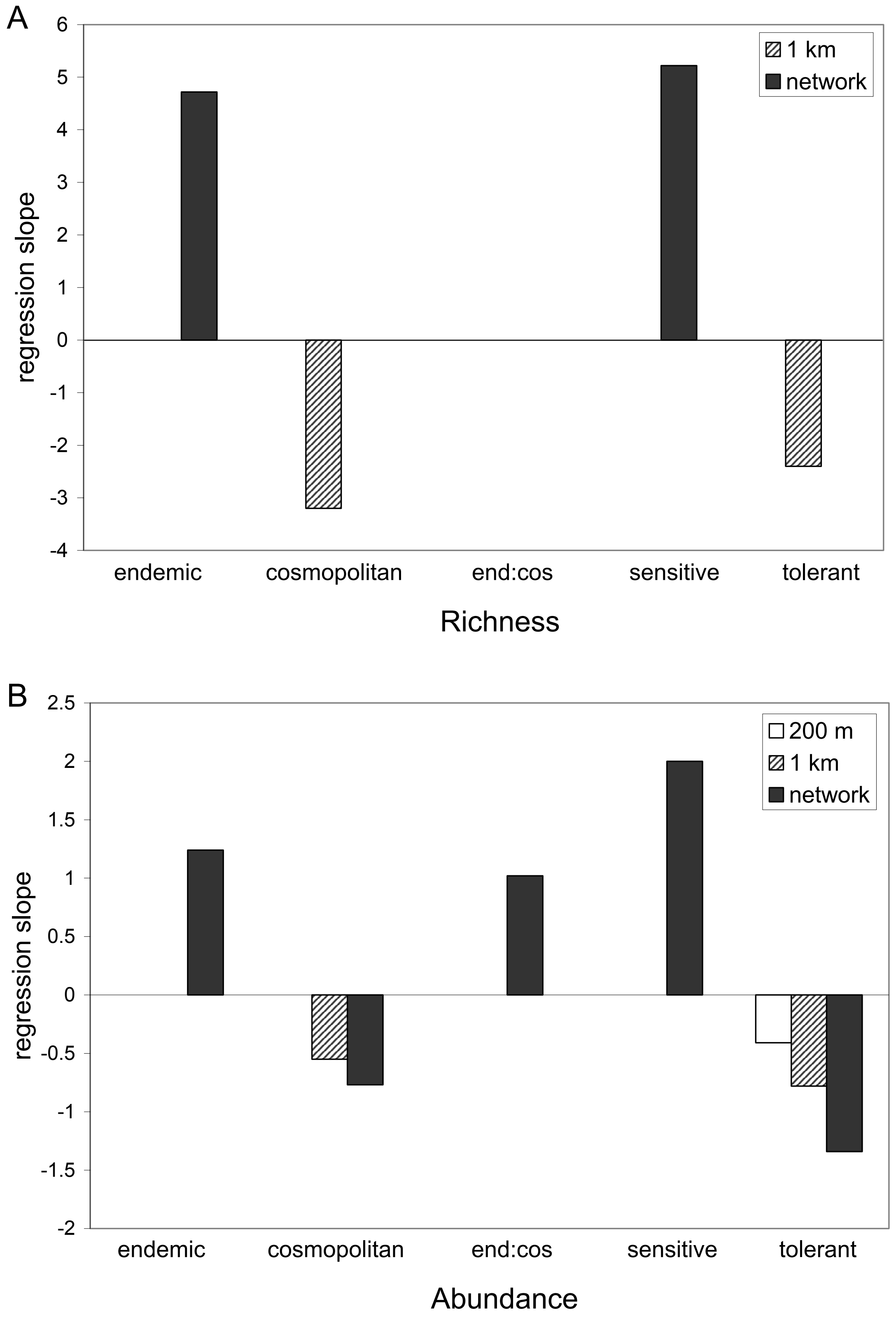

A
6
5
4
3
2
1
0
-1
-2
-3
-4
regression slope
▨ 1 km
■ network
endemic
cosmopolitan
end:cos
sensitive
tolerant
Richness
B
2.5
2
1.5
1
0.5
0
-0.5
-1
-1.5
-2
regression slope
☐ 200 m
▨ 1 km
■ network
endemic
cosmopolitan
end:cos
sensitive
tolerant
Abundance

Fig. 3.2

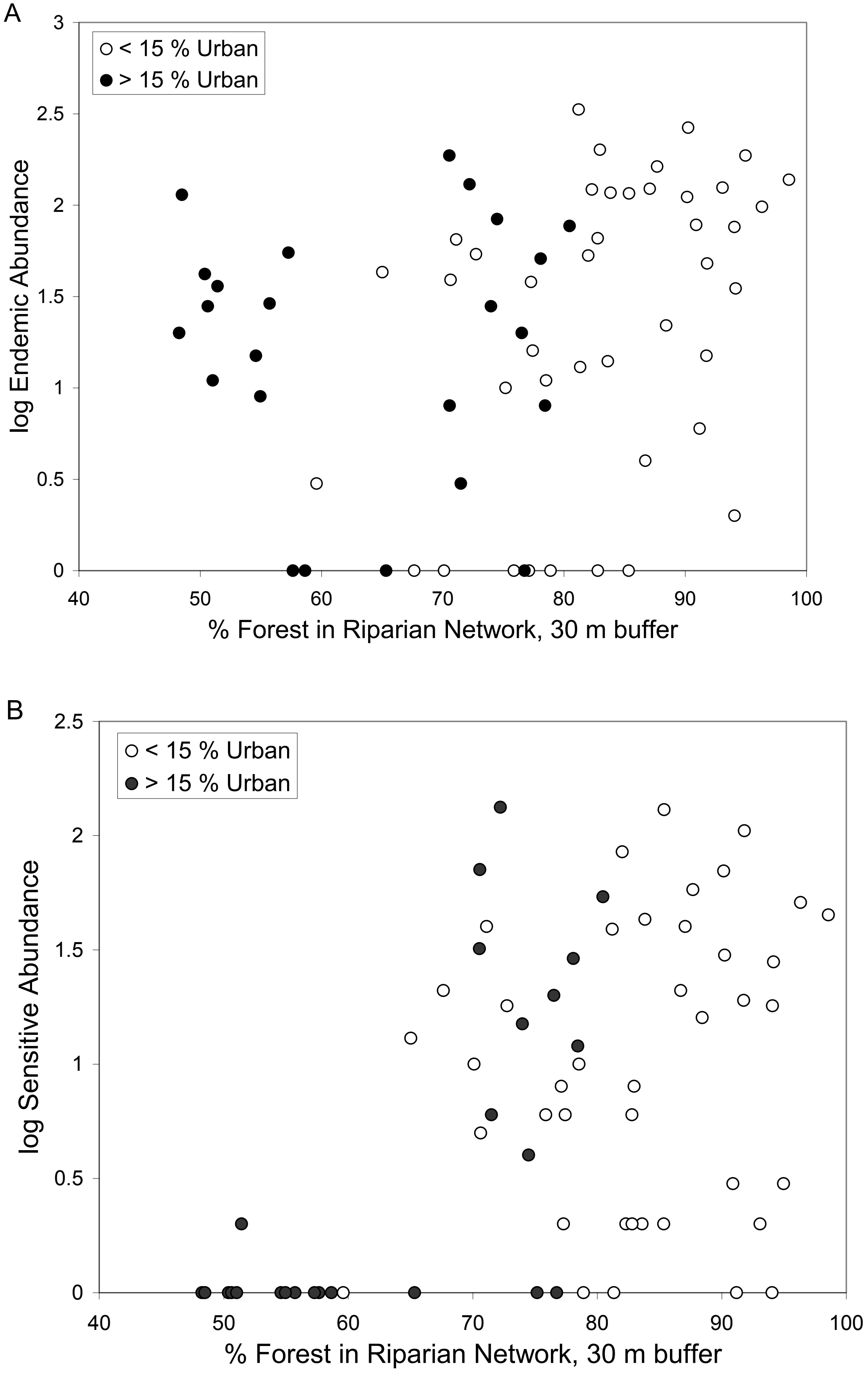
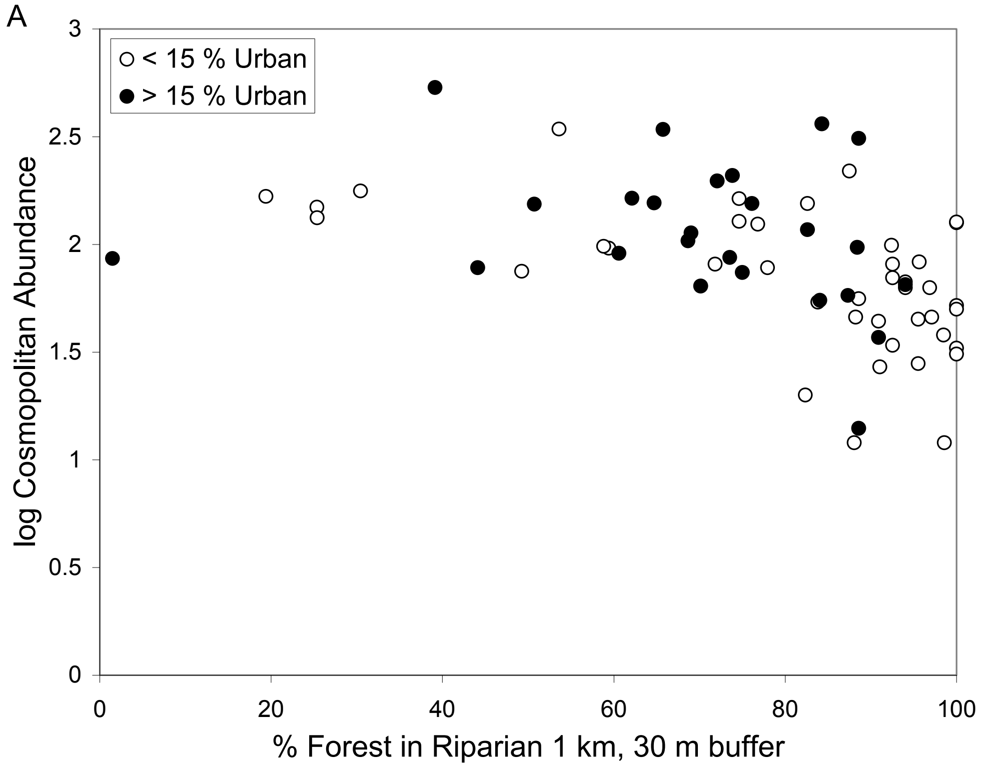
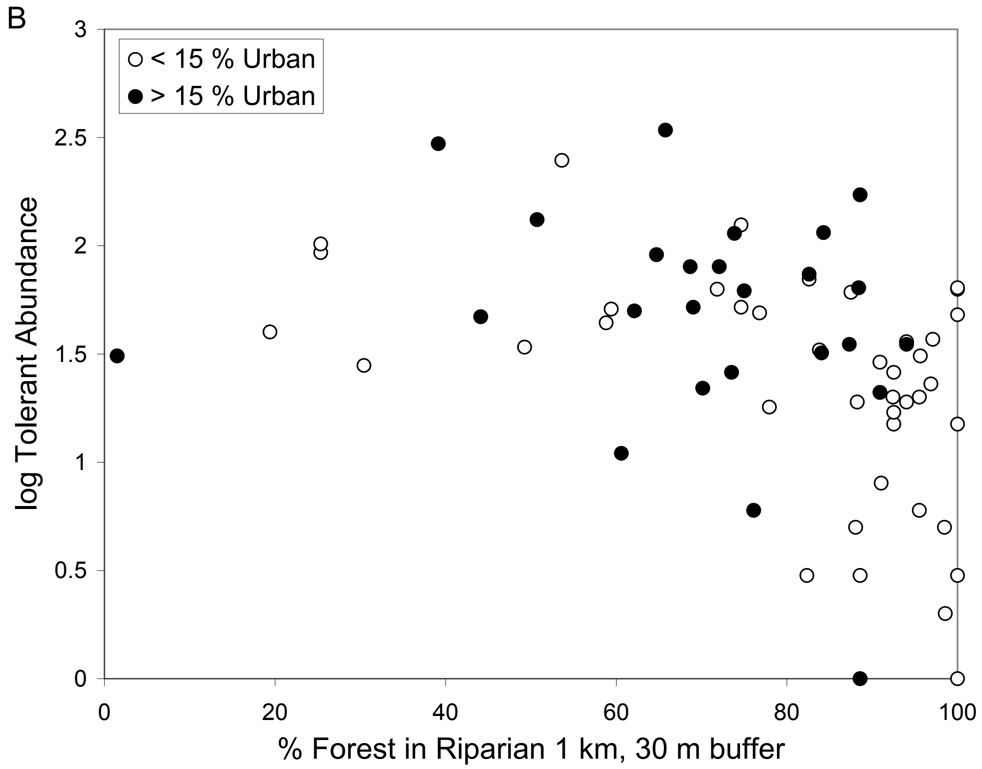

A
3
2.5
2
1.5
1
0.5
0
40
50
60
70
80
90
100
log Endemic Abundance
% Forest in Riparian Network, 30 m buffer
○ < 15 % Urban
● > 15 % Urban
B
2.5
2
1.5
1
0.5
0
40
50
60
70
80
90
100
log Sensitive Abundance
% Forest in Riparian Network, 30 m buffer
○ < 15 % Urban
● > 15 % Urban

Fig. 3.3


A
○ < 15 % Urban
● > 15 % Urban
log Cosmopolitan Abundance
0
0.5
1
1.5
2
2.5
3
0
20
40
60
80
100
% Forest in Riparian 1 km, 30 m buffer


B
○ < 15 % Urban
● > 15 % Urban
log Tolerant Abundance
0
0.5
1
1.5
2
2.5
3
0
20
40
60
80
100
% Forest in Riparian 1 km, 30 m buffer

Appendix 3.1. Fish assemblage integrity categories for species of fish sampled.

Endemic species (limited to Coosa River drainage)[a]

tricolor shiner (*Cyprinella trichroistia*); rainbow shiner (*Notropis chrosomus*); Coosa shiner (*Notropis xaenocephalus*); Coosa banded sculpin (*Cottus carolinae zopherus*); Etowah darter (*Etheostoma etowahae*); Cherokee darter (*Etheostoma scotti*); bronze darter (*Percina palmaris*)

Cosmopolitan species (native to at least 10 major drainages)[a]

largescale stoneroller (*Campostoma oligolepis*); golden shiner (*Notemigonus chrysoleucas*); creek chub (*Semotilus atromaculatus*); spotted sucker (*Minytrema melanops*); black redhorse (*Moxostoma duquesnei*); golden redhorse (*Moxostoma erythrurum*); snail bullhead (*Ameiurus brunneus*); yellow bullhead (*Ameiurus natalis*), brown bullhead (*Amieurus nebulosus*); speckled madtom (*Noturus leptacanthus*); eastern mosquitofish (*Gambusia affinis*); western mosquitofish (*Gambusia holbrooki*); redbreast sunfish (*Lepomis auritus*); green sunfish (*Lepomis cyanellus*); warmouth (*Lepomis gulosus*); bluegill sunfish (*Lepomis macrochirus*); redear sunfish (*Lepomis microlophus*); spotted bass (*Micropterus punctulatus*); largemouth bass (*Micropterus salmoides*); speckled darter (*Etheostoma stigmaeum*); blackbanded darter (*Percina nigrofasciata*)

Sensitive species (negative response to urban land cover)[b]

Alabama shiner (*Cyprinella callistia*); tricolor shiner (*Cyprinella trichroistia*); rainbow shiner (*Notropis chrosomus*); black redhorse (*Moxostoma duquesnei*) or golden redhorse (*Moxostoma erythrurum*); snail bullhead (*Ameiurus brunneus*); speckled madtom (*Noturus leptacanthus*); speckled darter (*Etheostoma stigmaeum*); bronze darter (*Percina palmaris*)

Tolerant species (capable of completing life cycle in lentic environments)[c]

golden shiner (*Notemegonus chrysoleucas*); yellow bullhead (*Ameiurus natalis*), brown bullhead (*Amieurus nebulosus*); eastern mosquitofish (*Gambusia affinis*); western mosquitofish (*Gambusia holbrooki*); redbreast sunfish (*Lepomis auritus*); green sunfish (*Lepomis cyanellus*); warmouth (*Lepomis gulosus*); bluegill sunfish (*Lepomis macrochirus*); redear sunfish (*Lepomis microlophus*); largemouth bass (*Micropterus salmoides*)

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[a] From Walters et al. (2003)

[b] From S. J. Wenger (unpublished data, *Etheostoma scotti* excluded)

[c] Based on Travnichek et al. (1995), classified using Etnier and Starnes (1993) and Mettee et al. (1993)

CHAPTER 4

INVESTIGATING HYDROLOGIC ALTERATION AS A MECHANISM OF FISH ASSEMBLAGE SHIFTS IN URBANIZING STREAMS[1]

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[1] Roy, A.H., M.C. Freeman, B.J. Freeman, S.J. Wenger, W.E. Ensign, and J.L. Meyer. Accepted with major revisions by the *Journal of the North American Benthological Society*. Reprinted with permission of publisher.

Abstract

A primary mechanism of urban effects on stream ecosystems is via altered hydrology; increases in impervious cover result in rapid delivery of stormwater to streams creating flashy flows and reduced baseflows. Although it is presumed that these changes in flow regimes can alter fish assemblages, there is little empirical evidence demonstrating relations between fishes and hydrologic alteration. Further, we do not know what aspects of the hydrograph are particularly important in driving fish assemblage structure and what fishes are most affected by these hydrologic alterations in small streams. We measured hydrologic patterns and sampled fish assemblages in 30 small streams (8--20 km^2) in the Etowah River basin (Georgia, USA) that were stratified by percent impervious cover (<10%, 10-20%, >20%) and the estimated degree of hydrologic alteration (based on synoptic measurements of baseflow yield). Hydrologic variables were derived from stage gauges at each study site for one year (January 2003-2004). Increased % impervious cover was positively correlated with the frequency, magnitude, and duration of storm events during most seasons. Other measures of flashiness such as the rate of the rising and falling limb of the hydrograph were also positively related to impervious cover, except during high flow spring months when soil saturation likely masked any land cover effects. Increased duration of low flows associated with impervious cover only occurred during the autumn low flow period, and this corresponded to significant increases in richness of lentic tolerant species. Altered stormflows in summer and autumn were related to decreased richness of endemic, cosmopolitan, and sensitive fish species, and decreased abundance of lentic tolerant species. Species that were predicted to be sensitive to urbanization based on life history and other attributes were also related to stormflow variables after accounting for variation associated with % fines in riffles. Overall, hydrologic variables explained 20-66% of the variation in fish assemblage richness and abundance. The linkages between hydrologic alteration and stream

fishes were potentially complicated by contrasting effects of elevated flows on sediment delivery and scour, and mediating effects of stream gradient. However, stormwater management practices that promote natural hydrologic patterns are likely to reduce the negative effects of impervious cover on stream fish assemblages.

Introduction

Urban development has become an increasing and dominant disturbance in stream ecosystems worldwide. Approximately half the world population now lives in cities (vs 12% in 1900), thus increasing both the area of land and the number of streams impacted by urban development (Cohen 2003). A primary mechanism by which urbanization impacts stream ecosystems is via altered hydrology. Urban development causes numerous modifications to the land surface such as vegetation clearing, soil compacting, ditching, draining, piping, and ultimately covering land with impermeable roofs and roads (Booth and Jackson 1997). These changes in the landscape result in measurable alterations to in-stream storm flow and baseflow hydrology (see Shaw 1994). Urbanization has been shown to: (1) increase the percent of precipitation that becomes surface runoff; (2) decrease the lag time between precipitation events and elevated stream flows (Graf 1977); (3) increase the magnitude of peak discharges by 2-5 times (Hollis 1974); (4) increase the frequency of high flow events by ≥ 10 times (Booth 1991); and (5) decrease the magnitude of low flows as a result of reduced groundwater recharge (Ferguson and Suckling 1990).

These alterations in stream hydrology, in turn, affect water quality, geomorphology and biotic assemblages. Increases in surface runoff may increase channel erosion (Trimble 1997), alter channel morphology (Doyle et al. 2000, Pizzuto et al. 2000), and increase sediment, nutrient, and contaminant delivery to streams (Wilber and Hunter 1977, Klein 1979, Herlihy et al. 1998, Ometo et al. 2000, Koplin et al. 2002). Fishes and invertebrates respond to urban land cover changes through changes in richness, diversity, density and biotic integrity (see reviews Schueler 1994, Paul and Meyer 2001). Studies have also noted changes in community composition (Scott and Helfman 2001, Walters et al. 2003) and feeding ecology of selected fishes (Weaver and Garman 1994, Poff and Allan 1995). These changes in fish assemblages appear to occur at relatively low levels of urbanization (e.g. 10-15% impervious; Schueler 1994, Wang et al. 2000).

Much of the evidence that altered hydrology impacts fish assemblages is from studies assessing hydrologic alteration downstream of hydropower dams (Power et al. 1996, Pringle et al. 2000, Freeman et al. 2001). However, many theoretical relations suggest that altered hydrology due to urbanization can impact fishes both directly and indirectly. As depicted in Fig. 4.1, changes in stormflow can directly impact fish assemblages via washout of eggs and young of year fishes, and subsequent disruption of life-cycles (Power et al. 1996, Poff et al. 1997, Freeman et al. 2001). Increased stormflows could also reduce water quality by increasing suspended sediment, contaminants, and nutrients to streams (Burkhead et al. 1997). Geomorphically, increased stormflows could increase channel erosion, affecting the frequency of pool/riffle sequences, and reduce habitat quality by decreasing bed texture in unstable stream channels, both of which should have negative impacts on fish assemblages (Meade et al. 1990, Waters 1995, Burkhead et al. 1997, Sutherland et al. 2002). Reduced magnitude and increased duration of low flows can lead to loss of habitat availability and quality (e.g. temperature) and subsequent changes in food web dynamics (Power et al 1996, Poff et al. 1997). Fish responses are expected to vary based on the timing of altered flows in relation to life cycles and life histories (Power et al 1996, Poff et al. 1997, Freeman et al. 2001, Bunn and Arthington 2002).

The goals of this study were to determine how and to what extent hydrologic alteration accounts for the negative relation between impervious cover and fish assemblage integrity.

Although impervious surface is an integrative measure of cumulative impacts to water resources, various factors in the landscape (e.g. storm water connection, impoundments, etc.) can result in variable and non-linear relations between impervious cover and stream hydrology. Thus, for this study we used continuous stream stage monitoring to quantify hydrologic alteration in streams representing an urban gradient and to determine relations between: (1) impervious surface cover and hydrologic alteration; and (2) hydrologic alterations and measures of fish assemblage integrity.

Methods

Study Sites

The study took place in the Etowah River system, a 4823 km^2 catchment in north-central Georgia, USA on the outskirts of metropolitan Atlanta (Fig. 2). The southeastern United States is a hotspot of stream fish diversity and endemism; there are approximately 76 extant fish species native to the Etowah River system, including 4 locally endemic species (Burkhead et al. 1997). Of these, 7 fish species are state protected and 3 fish species are federally protected under the U.S. Endangered Species Act, including the Cherokee darter, *Etheostoma scotti*, which is commonly found in small streams where our sampling took place. For most of the 20th century, the region was dominated by forest (primary and secondary growth) and agricultural (row-crop and pasture) land uses; however, the area has undergone rapid changes in land cover in the last two decades. Suburban development spreading north of the metropolitan Atlanta area (population > 4 million) has increased residential and commercial land uses along corridors of population growth. Thus, the sub-catchments within the Etowah River basin exhibit a range in urban, agriculture, and forest land cover.

Thirty sites were selected to encompass an expected range of hydrologic alteration across streams of similar size and with similar potential fish assemblages. All were small streams (8-20 km^2) within the Piedmont physiographic province of the Etowah River basin (Fig. 4.2). Most streams were located >1 km upstream of the juncture with a large river (e.g. mainstem of Etowah River) or reservoir. Streams that were impounded across all upstream tributaries or had a major chemical alteration (e.g. quarry) were excluded. These criteria reduced the initial set of 87 potential sites to 54 potential sites, which were then classified according to hydrologic alteration.

The thirty selected streams represented a presumed gradient of hydrologic alteration, which was approximated for the purpose of site selection by estimated impervious cover and baseflow yield (Table 1). Sites were stratified by three classes of percent impervious cover in the watershed, < 10%, 10-20% and >20%. Impervious cover was estimated by the following method: using ArcView© Geographic Information System (GIS), we assigned random points across the entire Etowah River basin. Each point that fell within low-density (410 points) and high-density (130 points) urban categories (according to *LandSat* TM 1998 land cover) was classified as pervious or impervious using 1999 U.S. Geological Survey ortho-corrected color infrared photos. Impervious surface percentages (25.37% for low-intensity urban, 48.46% for high-intensity urban) were then multiplied by area of that land use/cover category within each catchment and added to transportation area (assuming 100% impervious) to determine total impervious surface area.

Baseflow yield (defined as discharge/catchment area) was measured at one time at 54 sites on 18-19 June 2002 during baseflow conditions > 1 week after a rain event. Results of the survey of stream discharges revealed a geographic pattern of higher yields in the northeast portion and lower yields at sites in the southwest portion of the Etowah basin (dividing the basin at the upper end of Allatoona reservoir). Since these differences in baseflow are likely due to differences in soils or geology, we selected sites across the natural gradient of underlying flow conditions (i.e. evenly distributed in the northeast and southwest portions of the basin). This ensured that we would be able to detect a baseflow response to anthropogenic stress, if an effect existed. Sites were classified as either above or below the median baseflow yield within each class of percent impervious cover. In the <10% impervious class, we randomly selected 5 sites located in the northeast (most of which were above the median) and 5 sites located in the southwest (mostly below the median) portions of the basin. In the 10-20% and >20% impervious cover classes, sites in the northeast and southwest portions of the basin were more evenly distributed above and below the median yield. We therefore randomly selected 5 sites with yields above and 5 sites with yields below the median yield value for each of these impervious cover classes.

The final set of 30 sites thus represented three levels of impervious surface coverage, and a gradient of relative baseflow yield within each level (Table 1). The thirty sites encompassed most of the range of baseflow yield across the basin (0.0006-0.0062 $m^3s^{-1}km^{-2}$ for the 30 sites versus 0.0005-0.0096 $m^3s^{-1}km^{-2}$ for all 54 potential sites), but averaged higher mean percent impervious cover (15.7%) compared to all 87 small streams in the basin (11.6%).

Hydrologic monitoring

Streams were gauged at the base of each watershed using 2-m AquaRod® water level sensors (Advanced Measurements & Controls, Inc., Woodinville, WA, USA), which use capacitance to measure stage height. AquaRods® have internal dataloggers that were set to record water level every hour and with every 6 mm change in stage height. The data were downloaded seasonally from 16 January 2003 to 28 January 2004.

Steady-flow analysis in HEC-RAS® (Version 2.2, Hydrologic Engineering Center, U.S. Army Corps of Engineers) was used to determine the mean hydraulic depth for the 0.5-year recurrence interval (RI) flood at each site. Discharges for the 0.5-yr RI flood were calculated based on basin area at each site using flood-frequency formulas derived for rural streams in the Georgia piedmont (Stamey and Hess 1993). Manning’s n, stream slope, and cross-sectional area at the water level sensor were determined from field measurements and incorporated into the HEC-RAS® model for each stream. A Topcon® AT-F6 level and stadia rod were used to obtain elevations for a channel cross-section at the location of the AquaRod® and for calculating the energy grade line slope between riffle-tops for a 150-m reach. In addition, an aspect of habitat quality, % fines in riffle habitats, was measured since it was a mechanism by which hydrologic alteration could indirectly impact fishes, and because it may alter fish assemblages independently of hydrologic alteration. Three liters of bed sediment were collected from three riffles within each stream reach and brought back to the lab. Sediments were dried, sieved, and weighed to determine mean % fines (< 2 mm) for each stream.

We calculated 9 baseflow and 18 stormflow variables that we expected to respond to impervious cover and that could affect fish assemblages (Table 4.2). Because we did not have accurate stage-discharge rating curves for each site, we calculated hydrologic variables based on stage, which has been shown to appropriately quantify hydrologic alteration (McMahon et al. 2003). Baseflow variables included daily low stage variables, and magnitude and duration of low stage events below 25%, 10% and 5% of the median stage. Storm flow variables included frequency (i.e. number of flow excursions above a certain stage), magnitude, duration, and volume (stage height * hour) during events above a certain stage, and rate of change associated with the ascending and descending limbs of storms. Proportions (100%, 75%, and 50%) of the mean stage height of the 0.5-year RI flood were used to calculate storm flow variables. All magnitude variables were divided by the mean daily stage to adjust for differences in stream size.

Fish sampling

Fishes were sampled in August-October 2003 during baseflow conditions. Block nets were set every 50 m for a 150 m reach in each stream. The three adjacent 50 m reaches were sampled in a single pass, and one randomly selected reach of the three was sampled with three consecutive passes. During each pass, all habitats were thoroughly sampled using a backpack electroshocker (Model 12-B; Smith-Root® Inc): areas with sufficient flow were sampled by kicking with an 8-ft seine held downstream; pool habitats were sampled with dip nets; and sandy, shallow runs were sampled using seine hauls. Fishes were identified, measured, and released in the field or euthanized with buffered MS-222 and preserved in ~8% formalin for identification in the laboratory.

The program CAPTURE® (White et al. 1978) was used to calculate richness estimates using species detectability based on species caught in single-pass samples in three consecutive 50 m reaches. We used model M(h), which assumes heterogeneity of capture probabilities among species, to estimate species richness (Williams et al. 2002). The removal function in CAPTURE® was used to calculate capture probabilities for each species that declined in abundance among the three passes conducted in one 50 m reach. These capture probabilities were used to estimate fish abundance for each species at each site. For species that did not have depletion among passes we were unable to estimate abundance, so we used the total number of individuals captured instead.

Fish assemblage structure was evaluated based on estimated overall species richness, fish abundance, and richness and abundance of fish assemblage subsets: endemic species, cosmopolitan species, fluvial specialists, lentic tolerants, and sensitive species (see Appendix 4.1). Cosmopolitan species were defined as those fishes native to at least 10 major drainages, and were expected to increase with urbanization (Walters et al. 2003). Endemics, species primarily limited to the Coosa River drainage (which includes the Etowah River), were expected to decrease with urbanization (Walters et al. 2003). We also examined the ratio of endemics to cosmopolitans (based on species richness and abundance), which should reflect a homogenization of fish assemblages coincident with loss of endemic species (Scott and Helfman 2001, Walters et al. 2003). Fluvial specialists are species that require lotic environments for at least part of their life cycle (Travnichek et al. 1995); we classified these species using Etnier and Starnes (1993) and Mettee et al. (1993). Lentic tolerants were fishes that are habitat generalists, capable of completing their life cycle in lakes or reservoirs (Travnichek et al. 1995). We hypothesized that lentic tolerants would increase and fluvial specialists would decrease with increased urbanization. Sensitive species were those species found in the Etowah River basin that were expected to be sensitive to disturbance (due to specific life history or habitat requirements), and exhibited a negative response to increased urban land cover based on fishes sampled at other sites in earlier studies (Seth J. Wenger, University of Georgia, *unpublished data*).

Landscape variables

Subsequent to site selection, impervious cover for the Etowah River basin was calculated using a more accurate algorithm by the Natural Resources Spatial Analysis Laboratory (Institute of Ecology, University of Georgia, Athens, GA, USA). A classification and regression tree (CART) model was created using Cubist® software. A training set was developed using three 1999 color infrared digital ortho quarter quads (~30,000 hectares each; DOQQs) for the region encompassing the Etowah River basin and by classifying 1 m pixels as impervious or non-impervious. These data were then associated with *Landsat* ETM+ satellite imagery for September 2000, March 2001, and December 2001 to build regressions based on the training set. The regressions were extrapolated for regions outside the training set in order to determine %

impervious for the entire region. Accuracy of the CART model was assessed based on comparison with a sub-set of data withheld from the training set (SE = 7.5%, $r^2 = 0.89$). The CART model should produce more accurate impervious cover estimates than the methods used for site selection because it calculates regressions for each land cover category (vs. only categories of low- and high-density urban cover and transportation in original classification).

The number and area of impoundments in each sub-catchment were determined to understand the importance of these hydrologic features in mediating hydrologic alteration. Percent open water was calculated for each sub-catchment using 2001 *Landsat* TM satellite imagery that was classified using a 17-class system. In addition, impoundments were mapped from 1999 digital aerial ortho-photos and used to calculate number of impoundments per basin area in each sub-catchment.

Data analyses

Throughout the study period, many of the AquaRods® (24 of the 30) experienced periods of failure or improper function. To deal with this incomplete data set, analyses were divided by seasons based on AquaRod® downloading dates. Data for each season included different sets of study sites depending on which had complete data records (Table 1). Seasonal periods of record (PORs) included early and late spring (16 January-14 April; n = 14, and 15 April-14 May, n = 12), summer (15 May-7 August, n = 17), autumn (15 August-4 November, n = 22), and winter (11 November-28 January, n = 20). The full data set was used for analyses that did not include hydrologic variables.

All variables were tested for normality using Shapiro-Wilk goodness of fit test and transformed when necessary. All fish abundance metrics were transformed using log (x+1) and percentage variables were transformed using arcsin(sqrt(%/100). Correlation analysis (Pearson's *r*) was used to relate % impervious cover to hydrologic variables and fish assemblage measures.

Principal components analysis (PCA) was used to reduce baseflow and stormflow variables into a few metrics and determine the stage variables most important in explaining the variation in hydrology across sites. We used multiple linear regression analysis (stepwise regression, forward selection, $p < 0.05$) to predict fish assemblages using mean % fines in riffles and PCA axis scores for summer and autumn baseflow and stormflow variables. Correlation analysis was also used to relate hydrologic variables to mean % fines, stream slope, % water in basin, and number of impoundments per basin area. All analyses were preformed using JMP® Version 4.0 statistical software (SAS Institute Inc., Cary, NC, USA).

Results

Hydrologic alteration due to increased impervious cover

Impervious cover was slightly lower using the CART model with 2001 land cover data (mean = 12.0% impervious, range = 1.7%-31.0%) compared to the methods used for site selection with 1998 land cover data (mean = 15.7 % impervious, range = 0.7-39.1). In general, sites in the low impervious class (< 10% impervious) for site selection had similar impervious cover, whereas 18 of the 20 sites in the medium and high impervious classes had lower impervious cover with the CART model, with a maximum decrease of 13.8% impervious cover and a maximum increase of 4.3% impervious cover. These changes obscured the impervious classes used during site selection, so sites were analyzed exclusively as a gradient.

Average annual daily discharge in the Etowah River was below average (< 70% of the 50-year average discharge) for the 4 years preceding the study (1999-2002). In 2003, mean daily discharge was 20% higher than the 50-year average and nearly double the discharge of the previous 4 years (Fig.4.3). The sites experienced an average of 6.9 storms that exceeded the 0.5-yr RI flood mean stage over the period of record (~ 1 year; Table 4.3). Storms primarily occurred during late spring (Apr.-May) and summer (May-Aug.), exceeding the 0.5-yr RI flood mean stage at rates equivalent to 26 and 24 storms per year, respectively. Storms rarely occurred during autumn (Aug.-Nov.), when sites experienced ≤ 1 storm exceeding the 0.5-yr RI flood mean stage (Table 4.3).

Baseflow stage variables generally showed no relationships with impervious cover (Table 4.4). However, in autumn (Aug.-Nov.) there was a significant relationship between the duration of low flow events < 25th percentile flow and impervious cover ($r = 0.69$, $p < 0.001$), although duration of even lower flows was not related to imperviousness. This relationship occurred at the lowest flow time of the year, when we would expect measurable reductions in baseflows associated with urbanization.

Stormflow variables were related to impervious cover primarily during summer, autumn, and winter (Table 4.5). The frequency and duration of excursions above 50%, 75% and 100% mean stage of the 0.5-yr RI flood and the rate of the rising and falling limbs of the storm hydrograph consistently increased with increasing impervious cover (Table 4.5). The magnitude of high flow events was significantly correlated to impervious cover only in autumn (Aug.-Nov.). In spring, the number of times the stage rose > 5 cm per hour increased with increasing imperviousness in Jan.-Apr.; however, there were no significant relationships with stormflow variables in Apr.--May (Table 4.5). Fig. 4 shows example storm hydrographs for sites with high (site 57) and low (site 27) impervious cover. The high impervious cover site experienced greater frequency and magnitude of storm events relative to the 0.5-yr RI flood mean stage, although stage height during the six largest storms was similar (Fig. 4.4).

Variation in fish assemblages

Sites ranged in richness from 7 to 20 species collected in the three consecutive 50 m reaches, with estimated richness ranging from 8 to 25 species per stream (Table 4.6). We collected an average of 153 individuals per 50 m stream reach (range 37-435 individuals), and estimated overall abundance to be 37 to 512 individuals in one 50 m stream reach based on the 3-pass removal. All richness and abundance values reported henceforth are estimated values based on calculated capture efficiencies.

There were no significant relationships between percent impervious cover and total fish species richness or abundance; however, richness and abundance of fluvial specialist and lentic tolerants responded as hypothesized: richness and abundance of fluvial specialists was reduced in sites with high impervious cover, and lentic tolerants increased with increasing impervious cover (Table 4.6). Endemic species abundance and ratio of endemics to cosmopolitan abundance also both decreased with increasing impervious cover. We did not find significant relationships between impervious cover and abundance of fishes that were expected to be sensitive to urbanization; however, as expected, all species groups except cosmopolitan and lentic tolerant species varied inversely with percent impervious cover (Table 4.6).

Linking fish assemblages and hydrology

Principal components analysis (PCA) was used to reduce the set of baseflow and stormflow variables into a smaller, uncorrelated subset of variables for summer and autumn. These two seasons were chosen because they reflected high (summer) and low (autumn) flow periods that are most likely to impact fish assemblages and had the most complete hydrologic data. For the 9 baseflow variables, 2 principal component axes accounted for a majority of variation in the data (≥ 80% cumulative variance explained for each season; Table 7). For both seasons, the first PCA axis was related to baseflow magnitude variables and the second PCA axis was related to duration of the low flow events. PCA reduced the set of 18 stormflow variables to 3-4 principal components (≥ 89% cumulative variance explained for each season). The first PCA axis included equal weightings of the 18 variables in both seasons. In summer, the second PCA was highly weighted with stormflow magnitude variables, while the third and fourth PCA axes were related to stormflow volume. In autumn, stormflow PCAs 2 and 3 had variable loadings including magnitude, volume, and measures of storm flashiness (Table 4.7).

Autumn baseflow and summer and autumn stormflow hydrology (as summarized using PCA axes) explained some variation in fish richness and abundance assemblage measures (Table 4.8). Endemic richness was predicted by decreased stormflow alteration (frequency, magnitude, duration, volume, and "flashiness") in summer months. Increased cosmopolitan richness was predicted by longer duration of autumn baseflows and lower stormflow volume. Baseflow duration in autumn explained 63% of the variation in lentic tolerant richness across sites, with longer baseflow durations resulting in more lentic species. The abundance of lentic tolerants was predicted by decreased magnitude of summer stormflows and autumn baseflows. The richness and abundance of sensitive species were best explained by stormflow alteration, % fines in riffles, and baseflow duration (abundance only). Although hydrologic alteration was related to fish assemblages, stormflow and baseflow PCAs typically explained < 50% of the variation in fish assemblages (Table 4.8).

Other correlates of hydrologic alteration and fish assemblages

Mean % fines in riffles was negatively related to the abundance of endemic species and the ratio of endemics to cosmopolitans (Table 4.6). Percent fines also explained 33% and 46% of the variation in sensitive species richness and abundance, respectively. Summer and autumn hydrologic PCAs were not correlated with % fines; however, stream slope was negatively related to % fines ($r^2 = 0.39$, $p < 0.001$) and the magnitude of summer storms (Table 4.9).

In addition to impervious cover, the area and number of reservoirs in the catchment were important predictors of hydrologic alteration. Percent open water land cover in the sub-catchment was positively related to summer baseflow duration and negatively related to summer stormflow magnitude (Table 4.9). The number of impoundments was also negatively related to summer stormflow magnitude, but positively related to autumn increased stormflows. The number of impoundments per basin area was correlated with summer baseflow magnitude and autumn baseflow duration (Table 4.9). Impervious cover was not correlated with percent of open water ($r = 0.21$, $p > 0.05$) or the number of impoundments per area ($r = 0.26$, $p > 0.05$).

Discussion

Impervious impacts in small streams

Although previous studies have indicated predictable alterations to stormflow and baseflow hydrology with increasing urbanization, many of these results are based on data from large streams and rivers that have long-term gauges (Ferguson and Suckling 1990, Rose and Peters 2001) or data extrapolated from streams with gauges (Booth and Jackson 1997, but see Utah sites in McMahon et al. 2003). For the small streams in our study (8-20 km^2), we demonstrated an increase in the frequency, magnitude, and duration of high storm flows associated with higher % impervious cover in the catchment. There was also increased "flashiness" in our urbanized streams, as demonstrated by increases in the rates of the rising and falling limbs of the storm hydrograph. These results support theories of urban effects on stream storm hydrology (Hollis 1974, Booth 1991, Shaw 1994).

Interestingly, we did not find strong evidence that increased urbanization resulted in reduced baseflows at our sites. Previous studies have suggested a decrease in the magnitude of low flows associated with urbanization due to reduced groundwater recharge (Ferguson and Suckling 1990). One possible explanation for our lack of response is that the groundwater table was not lowered during 2003, a year with higher than average precipitation. Further, the high density of septic systems in the Etowah River basin (74-94% of households have on-site treatment systems;

Evans et al. 1999) may provide sufficient groundwater recharge in urban areas. However, we did find evidence in autumn (during the lowest flow months) that the duration of low flows was significantly correlated with increasing imperviousness. The abnormally high flows in 2003 may have masked the response of baseflows to urbanization during other seasons. It is also possible that reduced baseflows may not be an important aspect of altered hydrology associated with urbanization, especially in regions of the world with relatively high precipitation. Although McMahon et al. (2003) hypothesized that streams occurring in highly urban areas would have reduced duration of low flows, they were not able to confirm their prediction with data from Alabama, Massachusetts, or Utah. Low flows are not only influenced by groundwater, but also depend on topography, evapotranspiration, and in-stream hyporheic processes (Nilsson et al. 2003). Further, groundwater recharge from leaky pipes (Yang et al. 1999) and increased irrigation in wealthy areas (Al-Rashed and Sherif 2001) may offset any reductions in infiltration in urban areas. Cumulatively, these factors suggest that reduced baseflows in urban settings may not be a typical symptom of the "urban stream syndrome" characterized by Meyer et al. (*in review*).

Stormflow hydrology was largely unrelated to impervious cover in the spring months. During high flow times of the year (e.g. spring) when the ground is saturated and evapotranspiration is lowest, precipitation events in relatively forested catchments may behave similarly to catchments that have high impervious cover (Hollis 1974). In contrast, precipitation that falls during drier months is often able to infiltrate in forested catchments, thus minimizing surface runoff. The relatively frequent occurrence of large storms during 2003 probably also limited relations between impervious cover and storm magnitude and volume to autumn months.

We recorded very few storms that exceeded the mean depth of the 1-year RI flood (2.4 over ~ 1 yr) and higher stage thresholds, thus forcing us to calculate stormflow variables using lower thresholds (i.e. mean depth of the 0.5-yr RI flood and percentages thereof). Our data show that these more frequent storm events are important predictors of fish assemblages. However, larger storms may also be important, especially since they are likely to affect stream geomorphology (e.g. 1- to 2.3-yr RI flood determines bankfull conditions; Williams 1978). A longer dataset is necessary to determine the relations between % impervious cover and larger storm events, and also the importance of inter-annual variation in storm hydrology, both of which may be important influences on fish assemblages.

Natural and anthropogenic drivers of fish assemblages

Increasing urbanization across our study sites was associated with a shift toward fish assemblages that favor habitat generalist species (tolerant of lentic conditions) and a loss of stream-dependent species. The shift toward lentic tolerant species, in turn, was associated with lower and more prolonged low-flow conditions during autumn. Conversely, we observed lower abundances of lentic tolerant species with increased summer stormflow magnitude. Elevated stormflows have the potential to depress juvenile survival of many stream fishes, but may particularly affect species that reside and/or spawn in habitats more susceptible to scour (such as pools, as do most of our lentic tolerant species).

We also found evidence that the abundance of endemic species was reduced in highly urban sites, supporting geographic patterns described by Burkhead et al. (1997) and results of Walters et al. (2003a). Richness of endemic species and richness and abundance of sensitive species were predicted by altered stormflow; however, these variables were also predicted by % fines. Although changes in streambed coarseness could be a consequence of altered stormflow hydrology, we found no significant correlations between % fines and stormflow variables. Depending on time since urbanization and the stability of the channel, stormflows could have different effects on sedimentation (Henshaw and Booth 2000), and, subsequently, fish assemblages. For example, in unstable, urbanizing streams, high flow events might increase bank erosion and sediment inputs to streams, or mobilize historic bed sediment within the stream (Trimble 1997). In contrast, a stream that has been urban for several decades may experience streambed coarsening and scouring during high flow events (Finkenbine et al. 2000). Both scenarios are likely occurring within the 30 study streams, potentially offsetting the ability to predict fish assemblages based on stormflow variables.

Stream slope has also been shown to be an important factor driving streambed coarseness, and, subsequently, fish assemblages in the Etowah River basin (Walters et al. 2003a, b). Specifically, Walters et al. (2003a) found that lower slope streams had finer bed sediments and fishes dominated by cosmopolitan species. Although slope was negatively related to % fines in riffles ($r^2 = 0.39$), this relationship was not as strong as that found in Walters et al. ($r^2 = 0.62$), and there were few significant relationships between slope and fish assemblage variables.

There were two critical times of year when we expected altered hydrology to have the biggest influence on fish assemblages: (1) in late spring and summer during spawning; and (2) in autumn during low flow periods with limited habitat availability. During spring spawning events, high stormflows may increase mortality of eggs and young of year (YOY) fishes (Power et al. 1996, Poff et al. 1997, Freeman et al. 2001, Bunn and Arthington 2002). Even though we did not have complete enough spring data to test our hypotheses, we saw reduced richness of three species groups associated with summer stormflows. The extent and magnitude of the effects of storms during 2003 may be best reflected by the relative abundance of fishes in future years. We also saw evidence that longer and lower baseflows in autumn were related to increased richness of cosmopolitan and lentic tolerant species. The lack of relations with other fishes, however, suggests that reduced baseflows were not a dominant mechanism of fish assemblage alteration. The few storms that occurred in autumn (and consequently raised the water level in the urban streams) may have been sufficient to offset impervious effects of reduced baseflows, as well as alleviate stress on fishes.

Studies have suggested that total impervious area is not a useful measure of urban impacts on streams because it does not account for connections between impervious surfaces and the stream network (Brabec et al. 2002, Walsh et al. *this issue*). For example, precipitation that falls on roads in forested landscapes may infiltrate into the ground next to the road and not negatively impact stream ecosystems. By measuring in-stream hydrology, we were essentially measuring the effectiveness of impervious cover (or drainage connection) at linking surface runoff from the landscape to streams. Effective imperviousness may be a good surrogate for altered hydrology in basins that are not affected by small impoundments and other water detention ponds. However, in-stream hydrologic measurements will provide the best picture of direct physical impacts to stream ecosystems.

Management applications

This research is part of a project designed to understand the impacts of development on imperiled fish assemblages in the Etowah River basin. An interdisciplinary team of scientists, lawyers, educators and policy analysts have been working with local governments in the creation of a comprehensive Habitat Conservation Plan (HCP) for the imperiled aquatic species of the Etowah River Basin under the U.S. Endangered Species Act. The findings of this study will be used to develop ordinances and management guidelines to minimize the impacts of future urban development on fish assemblages. For example, counties and city jurisdictions that fall within the basin might adopt stricter stormwater regulations, erosion and sedimentation control ordinances, or impervious surface limits as part of the regional HCP. To this end, we have characterized stream hydrology using variables that are both biologically relevant and potentially manageable. The challenge for scientists; however, is not only to identify important stressors to the system, but also to ensure that regulations are sufficient to protect imperiled species. This process will necessarily include continued assessment of the effectiveness of regulations and adaptive management to adjust for new findings (Wilhere 2002).

Historically, "stormwater management" has involved moving the water off streets and paved parking lots as efficiently as possible to ensure maximum public safety and comfort (Arnold and Gibbons 1996), but has failed to account for maintenance of ecological function in aquatic systems (Postel et al. 1996). Stormwater management for in-stream needs requires a revolution in the way policy-makers and public safety officers think about precipitation. In order to minimize fish assemblage alteration in urban streams, ordinances must be adopted to avoid exacerbating the frequency and magnitude of high storm events, and reduce the flashiness of streams during storm events. This is especially important during medium and low precipitation times of the year. Since impervious cover may also be reducing groundwater recharge in the Etowah basin, stormwater detention/retention basins might not be the best tool for managing stormwater for in-stream needs (Booth et al. 2002, Heitz et al. 2000). Alternative management tools include increasing the perviousness in urban areas through porous road and parking lot materials, raingardens, and/or drainage swales (Konrad and Burges 2001, Booth et al. 2002). Increasing the amount of infiltration within the catchment and decreasing the connectedness of stormwater pipes (Walsh et al. *this issue*) in urban areas should be best at mimicking natural landscapes and minimizing alteration to stream ecosystems.

Conclusions

This study has provided direct evidence linking hydrologic alteration by urbanization to altered fish assemblages. Although we demonstrated that both altered stormflows and reduced baseflows impact fish assemblages, future work is necessary to tease apart the mechanisms of hydrologic impact on fishes (as outlined in Fig. 1). These relations are complicated by the positive and negative pathways between stressors and fishes (e.g. between increased stormflows and % fines), and other variables (e.g. slope and impoundments) which may either mask or exacerbate hydrologic effects on fish assemblages. Although hydrologic alteration is a major pathway by which urban development may affect fishes, other urban stressors must also be addressed to evaluate the relative effectiveness of hydrologic management at protecting fish integrity.

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Table 4.1. Study sites listed in order of increasing % impervious cover based on 2001 land cover. Seasons where hydrologic data were complete are reported (Sp1 = spring, 16 Jan--14 Apr.; Sp2 = spring, 15 Apr.--14 May; Su = summer, 15 May--7 Aug.; Au = autumn, 15 Aug.--4 Nov.; Wi = winter, 11 Nov.--28 Jan.). Number of sites (n) and summary data (minimum, maximum, and mean) are reported.

Site no.	Stream name	County	UTM Easting (X)	UTM Northing (Y)	Complete hydrologic data for season					Basin area (km^2)	% Impervious cover (2001)	% Water (2001)	# Impoundments/ basin area	Stream slope	Mean % fines in riffle*
					Sp1	Sp2	Su	Au	Wi						
31	Gorman Branch	Cherokee	721372	3794025	X	X	X		X	12.1	1.7	0.0	0.2	0.007	6.9
33	Boston Creek	Bartow	715143	3789238	X	X	X	X	X	13.5	3.3	0.3	0.3	0.001	54.5
76	Upper Little Pumpkinvine	Paulding	695481	3749043			X	X	X	11.4	3.5	0.1	0.4	0.001	62.5
75	Ward Creek	Bartow	701808	3776978						11.2	4.9	0.9	0.8	0.003	12.2
27	Hickory Log Creek	Cherokee	733291	3797543	X	X	X	X	X	10.7	5.0	0.6	1.1	0.003	1.5
26	Murphy Creek	Cherokee	736664	3800322			X	X	X	9.5	5.0	0.5	1.2	0.004	7.0
44	Board Tree Creek	Cherokee	750901	3797621				X	X	17.5	6.5	1.2	1.5	0.003	28.2
11	West Fork Pumpkinvine	Paulding	701004	3767553				X		10.5	6.7	0.5	1.1	0.002	24.5
72	Lane Creek	Paulding	696519	3753818	X	X	X	X	X	16.4	7.1	1.7	1.6	0.005	37.9
80	Buzzard Flapper	Cherokee	747492	3796053	X				X	10.3	7.2	0.9	0.9	0.005	16.3
38	Black Mill Creek	Dawson	766692	3805091	X	X	X	X	X	16.4	7.2	0.2	0.6	0.003	18.6
45	Upper Smithwick Creek	Cherokee	745884	3793849	X	X	X	X		15.8	8.0	0.9	1.7	0.003	28.5
67	Westbrook Creek	Paulding	707590	3770001	X	X		X	X	19.0	8.5	1.1	0.7	0.003	59.5
46	Scott Mill Creek	Cherokee	732028	3789619	X	X	X	X	X	12.0	9.4	0.6	2.3	0.008	2.4
66	Possum Creek	Paulding	704003	3763567				X	X	14.6	9.7	1.4	1.2	0.003	29.6
71	Lawrence Creek	Paulding	701124	3758149		X	X	X	X	12.5	9.8	1.1	0.6	0.002	49.3
24	Polecat Branch	Pickens	738871	3810736	X	X	X	X		12.1	10.1	1.3	2.0	0.007	5.9
63	Upper Allatoona Creek	Cobb	713981	3762082						14.5	12.7	0.7	1.4	0.004	4.0

Table 4.1. Continued.

18	Town Creek	Pickens	734417	3813276		X	X	X	X	15.4	13.0	0.3	0.9	0.009	3.1
65	Picketts Mill	Paulding	708236	3761223				X	X	19.9	13.4	0.4	1.5	0.004	43.5
54	Trib. Sweat Mountain	Cherokee	736119	3775362			X	X		8.5	13.7	1.5	2.7	0.005	13.9
52	Copper Sandy Creek	Fulton	746302	3777695	X		X	X	X	15.6	13.9	2.0	1.2	0.006	30.3
48	Toonigh Creek	Cherokee	728571	3781614						16.2	15.8	0.4	0.7	0.003	12.7
60	Clark Creek	Cherokee	717590	3773997						17.0	16.7	0.6	1.3	0.005	26.0
55	East Fork Rubes Creek	Cobb	730400	3772584	X		X	X	X	11.2	20.5	0.7	2.0	0.002	21.7
56	West Fork Rubes Creek	Cobb	730386	3772578						11.2	21.8	1.1	0.9	0.002	43.9
62	Butler Creek	Cobb	715391	3766443	X			X	X	9.8	21.9	0.4	1.2	0.006	3.3
57	Lower Noonday Creek	Cobb	728024	3770707	X	X	X	X	X	17.0	23.7	0.9	0.6	0.001	16.3
61	Proctor Creek	Cobb	715642	3770400						19.8	29.8	0.5	0.6	0.003	89.0
58	Upper Noonday Creek	Cobb	726765	3767250			X	X	X	9.8	31.0	0.5	1.8	0.003	6.9
n					14	12	17	22	20	30	30	30	30	30	27
Minimum										8.5	1.7	0.0	0.2	0.001	1.5
Maximum										19.9	31.0	2.0	2.7	0.009	89.0
Mean										13.7	12.0	0.8	1.2	0.004	25.3

Table 4.2. Hydrologic variables calculated for each season January 2003-January 2004 for each site. Percentiles are based on median stage over period of record (POR). Mean stage of 0.5-year recurrence interval (RI) floods and 75% and 50% of the 0.5-year RI floods were calculated based on basin area, stream slope, cross-sectional area, and Manning's n using HEC-RAS®. Mean stage of these RI floods were adjusted by a correction factor, the minimum daily stage, to account for site differences in AquaRod® locations.

Baseflow variables		
Category	Code	Description
Magnitude	MinDaily	Minimum daily stage
Magnitude	Min7dayMean	Minimum 7-day mean stage/ mean daily stage
Magnitude	Min7dayMax	Minimum 7-day maximum stage/ mean daily stage
Magnitude	MeanLow<25	Mean low stage over POR (hr); low stage <25th percentile/ mean daily stage
Magnitude	MeanLow<10	Mean low stage over POR (hr); low stage <10th percentile/ mean daily stage
Magnitude	MeanLow<5	Mean low stage over POR (hr); low stage <5th percentile/ mean daily stage
Duration	DurLow<25	Maximum duration of low stage over POR (hr); low stage <25th percentile
Duration	DurLow<10	Maximum duration of low stage over POR (hr); low stage <10th percentile
Duration	DurLow<5	Maximum duration of low stage over POR (hr); low stage <5th percentile

Table 4.2. Continued.

Stormflow variables

Category	Code	Description
Frequency	Freq>Q0.5	Number of excursions > mean stage of 0.5-year RI flood
Frequency	Freq>75%Q0.5	Number of excursions > 75% mean stage of 0.5-year RI flood
Frequency	Freq>50%Q0.5	Number of excursions > 50% mean stage of 0.5-year RI flood
Magnitude	MeanHigh>Q0.5	Mean stage height of excursions > mean stage of 0.5-year RI flood/ mean daily stage
Magnitude	MeanHigh>75%Q0.5	Mean stage height of excursions > 75% mean stage of 0.5-year RI flood/ mean daily stage
Magnitude	MeanHigh>50%Q0.5	Mean stage height of excursions > 50% mean stage of 0.5-year RI flood/ mean daily stage
Duration	Dur>Q0.5	Number of hours of high stage over POR; high stage > mean stage of 0.5-year RI flood
Duration	Dur>75%Q0.5	Number of hours of high stage over POR; high stage > 75% mean stage of 0.5-year RI flood
Duration	Dur>50%Q0.5	Number of hours of high stage over POR; high stage > 50% mean stage of 0.5-year RI flood
Volume	Vol>Q0.5	Area (number of hours * stage height) of stage > mean stage of 0.5-year RI flood
Volume	Vol>75%Q0.5	Area (number of hours * stage height) of stage > 75% mean stage of 0.5-year RI flood
Volume	Vol>50%Q0.5	Area (number of hours * stage height) of stage > 50% mean stage of 0.5-year RI flood
Rate of Change	RateRise5	Number of time periods (hrs) when stage rises by at least 5 cm
Rate of Change	RateRise10	Number of time periods (hrs) when stage rises by at least 10 cm
Rate of Change	RateRise20	Number of time periods (hrs) when stage rises by at least 20 cm
Rate of Change	RateFall5	Number of time periods (hrs) when stage falls by at least 5 cm
Rate of Change	RateFall10	Number of time periods (hrs) when stage falls by at least 10 cm
Rate of Change	RateFall20	Number of time periods (hrs) when stage falls by at least 20 cm

Table 4.3. Range of values for frequency of storm events above 100%, 75%, and 50% mean stage of the 0.5-year recurrence interval (RI) flow for each season. Seasonal rates (mean per day) were extrapolated to the number of storms that would occur in the year for comparison.

	Mean	Mean/ day	Mean/ year (at that rate)	Range	Standard deviation
Spring (16 Jan.--14 Apr.)					
0.5-yr RI flood (100%)	1.59	0.018	6.5	(0-5)	1.18
0.5-yr RI flood (75%)	2.18	0.024	8.9	(0-6)	1.38
0.5-yr RI flood (50%)	3.00	0.034	12.3	(1-7)	1.87
Spring (15 Apr.--14 May)					
0.5-yr RI flood (100%)	1.08	0.036	13.2	(0-3)	0.67
0.5-yr RI flood (75%)	1.50	0.050	18.3	(1-3)	0.80
0.5-yr RI flood (50%)	2.17	0.072	26.4	(1-4)	1.19
Summer (15 May--7 Aug.)					
0.5-yr RI flood (100%)	3.00	0.035	12.9	(0-10)	2.72
0.5-yr RI flood (75%)	4.06	0.048	17.4	(0-11)	3.17
0.5-yr RI flood (50%)	5.67	0.067	24.3	(0-13)	3.51
Autumn (15 Aug.--4 Nov.)					
0.5-yr RI flood (100%)	0.17	0.002	0.8	(0-1)	0.39
0.5-yr RI flood (75%)	0.57	0.007	2.5	(0-2)	0.79
0.5-yr RI flood (50%)	1.00	0.012	4.5	(0-3)	1.09
Winter (11 Nov.--28 Jan.)					
0.5-yr RI flood (100%)	1.05	0.013	4.8	(0-4)	1.21
0.5-yr RI flood (75%)	1.59	0.020	7.4	(0-5)	1.44
0.5-yr RI flood (50%)	2.45	0.031	11.3	(0-7)	1.77

Table 4.4. Pairwise correlations (Pearson's *r*) between % impervious cover (2001) and baseflow stage variables for each season from January 2003-2004 (n = number of sites). *** $p < 0.001$.

	Spring (Jan - Apr) (n = 14)	Spring (Apr - May) (n = 12)	Summer (May - Aug) (n = 17)	Autumn (Aug - Nov) (n = 22)	Winter (Nov - Jan) (n = 20)
MinDaily	-0.18	-0.21	0.17	0.14	0.03
Min7dayMean	-0.21	-0.37	-0.03	-0.24	-0.01
Min7dayMax	-0.19	-0.11	-0.02	-0.25	0.01
MeanLow<25	0.26	-0.14	-0.03	-0.18	-0.01
MeanLow<10	0.21	-0.28	-0.01	-0.17	0.01
MeanLow<5	0.20	-0.37	0.0	-0.16	0.01
DurLow<25	-0.36	-0.31	-0.26	0.69 ***	-0.06
DurLow<10	-0.41	-0.33	0.17	0.28	-0.13
DurLow<5	0.29	-0.33	0.06	0.18	0.07

Table 4.5. Pairwise correlations (Pearson's *r*) between % impervious cover (2001) and stormflow stage variables for each season from January 2003-2004 (n = number of sites). * $p < 0.05$, ** *p* ,< 0.01, *** $p < 0.001$.

	Spring (Jan - Apr) (n = 14)	Spring (Apr - May) (n = 12)	Summer (May - Aug) (n = 17)	Autumn (Aug - Nov) (n = 22)	Winter (Nov - Jan) (n = 20)
Freq>Q0.5	0.03	0.52	0.79 ***	0.71 ***	0.53 *
Freq>75%Q0.5	0.05	0.53	0.79 ***	0.54 **	0.69 ***
Freq>50%Q0.5	0.3	0.14	0.78 ***	0.57 **	0.80 ***
MeanHigh>Q0.5	0.28	-0.4	-0.38	0.70 ***	0.18
MeanHigh>75%Q0.5	-0.05	-0.39	-0.41	0.62 **	0.15
MeanHigh>50%Q0.5	-0.08	-0.33	-0.43	0.53 *	0.10
Dur>Q0.5	0.23	0.47	0.52 *	0.68 ***	0.52 *
Dur>75%Q0.5	0.21	0.39	0.54 *	0.58 **	0.47 *
Dur>50%Q0.5	0.16	0.2	0.25	0.53 *	0.09
Area>Q0.5	-0.09	0.18	0.27	0.45 *	0.40
Area>75%Q0.5	0.01	0.26	0.37	0.60 **	0.45 *
Area>50%Q0.5	0.07	0.28	0.38	0.56 **	0.19
RateRise5	0.57 *	0.15	0.69 **	0.57 **	0.65 **
RateRise10	0.26	0.37	0.73 **	0.65 ***	0.60 **
RateRise20	-0.24	0.52	0.66 **	0.47 **	0.59 **
RateFall5	0.34	0.49	0.71 **	0.61 ***	0.66 **
RateFall10	0.14	0.49	0.64 **	0.57 **	0.61 **
RateFall20	0.11	0.31	0.37	0.37 *	0.58 **

Table 4.6. Mean, range, and standard deviation for fish assemblage metrics across the 30 sites. Pairwise correlations (Pearson's *r*) between fish variables and % impervious cover (2001) and mean % fines in riffles are reported. * $p < 0.05$, ** $p < 0.01$.

	Mean	Range	Standard deviation	% Impervious cover[a]	Mean % fines[a]
Fish Richness (no. species)					
Total	17.4	(8 - 25)	4.1	-0.27	-0.14
Endemics	2.4	(0 - 6)	1.6	-0.29	-0.28
Cosmopolitans	10.1	(5 - 17)	3.0	-0.11	-0.11
Endemics/Cosmopolitans[b]	0.26	(0 - 0.75)	0.20	-0.23	-0.13
Fluvial Specialists	12.3	(5 - 21)	4.2	-0.49 **	-0.15
Lentic Tolerants	5.4	(3 - 11)	2.0	0.44 *	0.06
Fluvial/Lentic	2.59	(0.45 - 5.67)	1.2	-0.47 **	-0.01
Sensitive	2.7	(0 - 7)	1.9	-0.48 **	-0.33
Fish Abundance (no. individuals)					
Total[b]	184.9	(37 - 512)	123.8	-0.07	-0.15
Endemics[b]	46.6	(0 - 335)	70.4	-0.38 *	-0.44 *
Cosmopolitans[b]	88.5	(12 - 431)	80.1	0.28	0.19
Endemics/Cosmopolitans[b]	1.11	(0 - 13.08)	2.52	-0.43 *	-0.42 *
Fluvial Specialists[b]	128.5	(10 - 469)	108.7	-0.42 *	-0.34
Lentic Tolerants[b]	56.9	(2 - 412)	76.4	0.45 *	0.08
Fluvial/Lentic[b]	7.03	(0.53 - 87.49)	15.7	-0.47 **	-0.23
Sensitive[b]	16.7	(0 - 103)	21.6	-0.39 *	-0.46 *

[a] Transformed for analysis using arcsin(sqrt(x/100))

[b] Transformed for analysis using log(x+1)

Table 4.7. Principal Components Analysis (PCA) axes for baseflow and stormflow stage variables for summer and autumn months. Axes all have an eigenvalue greater than 1. Cumulative % variance explained by the selected axes is in parentheses. Variables loading on each axis are listed.

	% Variance explained	Variables loadings
Summer (15 May--7 Aug.)		
Baseflow (80.0%)		
PCA 1[a]	58.7	All except DurLow<25, DurLow<10, DurLow<5
PCA 2	80.0	DurLow<25, DurLow<10, DurLow<5
Stormflow (94.9%)		
PCA 1	54.8	All variables
PCA 2	21.0	MeanHighQ0.5, MeanHigh75%Q0.5, MeanHigh50%Q0.5
PCA 3[a]	12.0	Vol>50%Q0.5
PCA 4[a]	7.1	Vol>Q0.5, Vol>75%Q0.5
Autumn (15 Aug.--4 Nov.)		
Baseflow (85.6%)		
PCA 1	71.2	All except DurLow<25, DurLow<10, DurLow<5
PCA 2	14.3	DurLow<25, DurLow<10, DurLow<5
Stormflow (89.7%)		
PCA 1	70.1	All variables
PCA 2	13.7	MeanHigh50%Q0.5, Vol>Q0.5, RateRise20, RateFall20
PCA 3	5.9	Freq>Q0.5, MeanHigh75%Q0.5, Vol>Q0.5, RateFall20

[a] Not normally distributed

Table 4.8. Multiple linear regression models for fish assemblage metrics using stepwise regression (n = 16 sites, forward selection, $p < 0.05$). Predictor variables included in model for selection were mean % fines in riffles and baseflow and stormflow PCA axes for summer (Su) and autumn (Au). Baseflow PCA 1 for summer was excluded in models due to autocorrelation with baseflow PCA 1 for autumn ($r = 0.85$).

	Adjusted R^2	Partial R^2	*p*	Predictor variables
Fish Richness				
Endemic	0.25	0.30	0.028	(-) stormflow PCA 1 (Su)
Cosmopolitan	0.55	0.35	0.016	(+) baseflow PCA 2 (Au)
		0.26	0.011	(-) stormflow PCA 4 (Su)
Lentic Tolerant	0.63	0.65	<0.001	(+) baseflow PCA 2 (Au)
Fluvial/Lentic	0.20	0.25	0.046	(-) baseflow PCA 2 (Au)
Sensitive	0.60	0.34	0.017	(-) stormflow PCA 1 (Su)
		0.20	0.033	(-) mean % fines in riffles
		0.14	0.044	(+) stormflow PCA 3 (Su)
Fish Abundance				
Lentic Tolerant	0.66	0.40	0.008	(-) stormflow PCA 2 (Su)
		0.30	0.003	(-) baseflow PCA 1 (Au)
Fluvial/Lentic	0.42	0.27	0.040	(+) stormflow PCA 2 (Su)
		0.23	0.031	(+) baseflow PCA 1 (Au)
Sensitive	0.72	0.46	0.004	(-) mean % fines in riffles
		0.20	0.016	(+) stormflow PCA 3 (Au)
		0.12	0.026	(-) baseflow PCA 2 (Au)

Table 9. Pairwise correlations (Pearson's *r*) between summer and autumn hydrologic alteration variables (principal components axes) and environmental variables. * $p < 0.05$, ** $p < 0.01$.

	Mean % Fines	Stream Slope	% Water in Basin	# Impoundments/ Basin Area
Summer Hydrology				
Baseflow PCA 1	-0.39	0.22	0.14	0.57 *
Baseflow PCA 2	0.00	0.16	0.51 *	-0.08
Stormflow PCA 1	0.08	-0.56 *	0.05	-0.03
Stormflow PCA 2	0.33	-0.56 *	-0.55 *	-0.52 *
Stormflow PCA 3	0.43	-0.12	0.17	-0.47
Stormflow PCA 4	0.12	-0.04	0.3	0.17
Autumn Hydrology				
Baseflow PCA 1	-0.36	0.29	0.05	0.30
Baseflow PCA 2	-0.36	0.20	0.33	0.54 *
Stormflow PCA 1	-0.11	-0.34	0.15	0.29
Stormflow PCA 2	-0.03	0.25	0.04	0.50 *
Stormflow PCA 3	-0.02	0.00	-0.08	0.49

Figure Legends

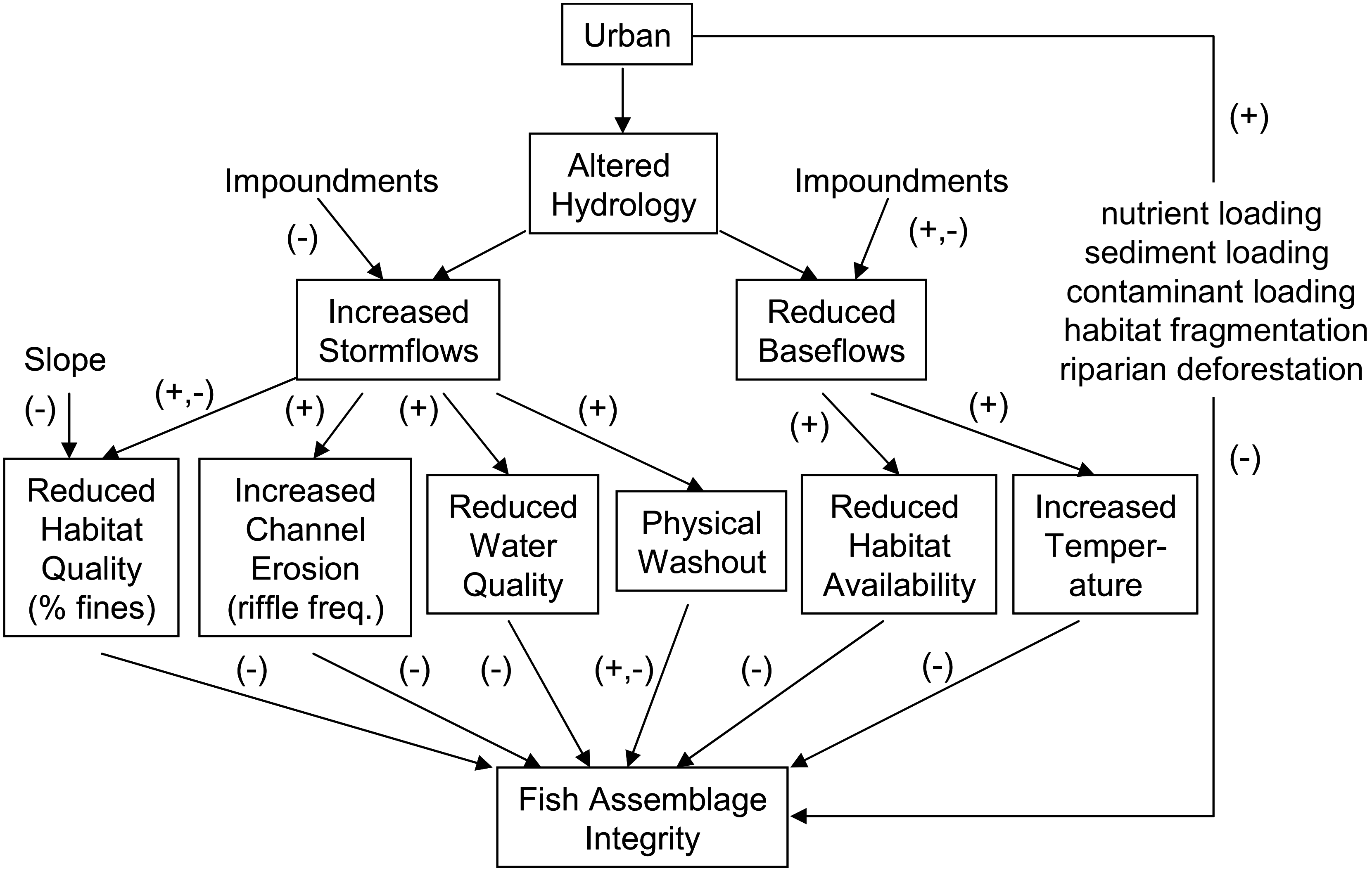
Fig. 1. A conceptual model of relations between urbanization and fish assemblage integrity. Potential mechanisms of impacts of altered hydrology (increased stormflows and reduced baseflows) on fishes are included. The independent contribution of impoundments in the landscape may reduce the effect of urbanization on increased stormflows, and have positive or negative effects on reduced baseflows. Stream slope may mediate the effects of increased stormflows on increased sediment inputs to streams. Finally, physical washout due to increased storms is expected to negatively affect all fishes, but may have disproportionate effects on tolerant species that spawn in depositional areas, resulting in positive and negative influences on fish assemblage integrity.

Fig. 2. Map of the 30 sub-catchments within the Etowah River basin (Georgia: USA) used for this study. Sub-catchments are divided into impervious cover categories (<10%, 10-20%, >20%) based on 1998 land cover data used for site selection. Abbreviations on Georgia map correspond to physiographic provinces: RV = ridge and valley, BR = blue ridge, P = piedmont.

Fig. 3. U.S. Geological Survey mean daily stream discharge data for 1994-2004 (tick marks indicate January of each year) for the Etowah River at Canton, Georgia, (USGS Station No. 02392000; upstream of Allatoona Reservoir; http://ga.waterdata.usgs.gov/ga/nwis/).

Fig. 4. Mean hourly stage data for a site with high impervious cover (Site 57, Lower Noonday Creek, 23.7% impervious) and a site with low % impervious cover (Site 27, Hickory Log Creek, 5.0% impervious). The mean stage of the 0.5-year recurrence interval flood (as calculated using HEC-RAS®) for each site is included.

Fig. 4.1.


Urban
(+)
Altered Hydrology
Impoundments
(-)
Impoundments
(+,-)
nutrient loading
sediment loading
contaminant loading
habitat fragmentation
riparian deforestation
Increased Stormflows
Reduced Baseflows
Slope
(-)
(+,-)
(+)
(+)
(+)
(+)
(+)
(-)
Reduced Habitat Quality (% fines)
Increased Channel Erosion (riffle freq.)
Reduced Water Quality
Physical Washout
Reduced Habitat Availability
Increased Temper-ature
(-)
(-)
(-)
(+,-)
(-)
(-)
Fish Assemblage Integrity

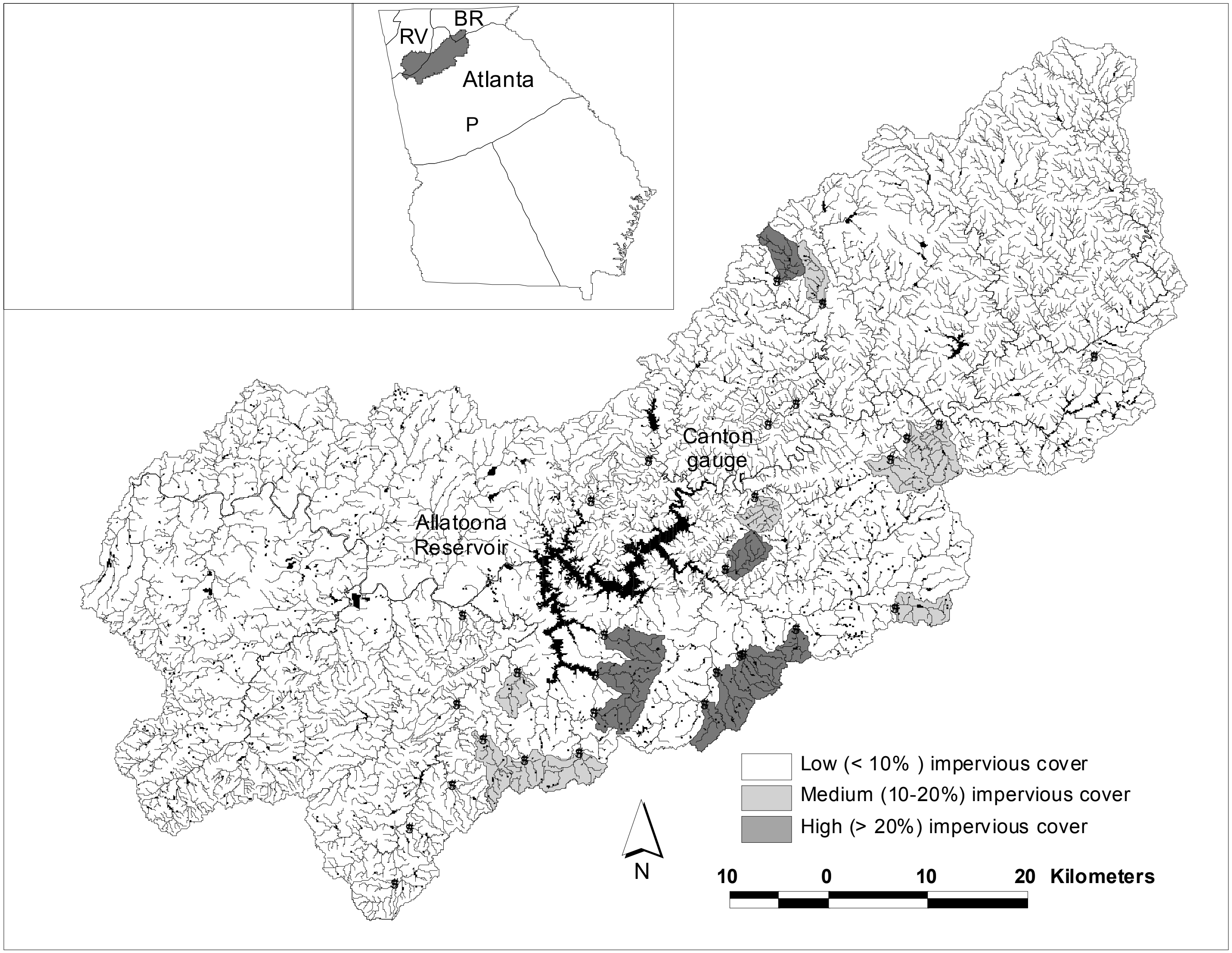

BR
RV
Atlanta
P
Canton gauge
Allatoona Reservoir
Low (< 10%) impervious cover
Medium (10-20%) impervious cover
High (> 20%) impervious cover
N
10
0
10
20
Kilometers

Figure 4.3.

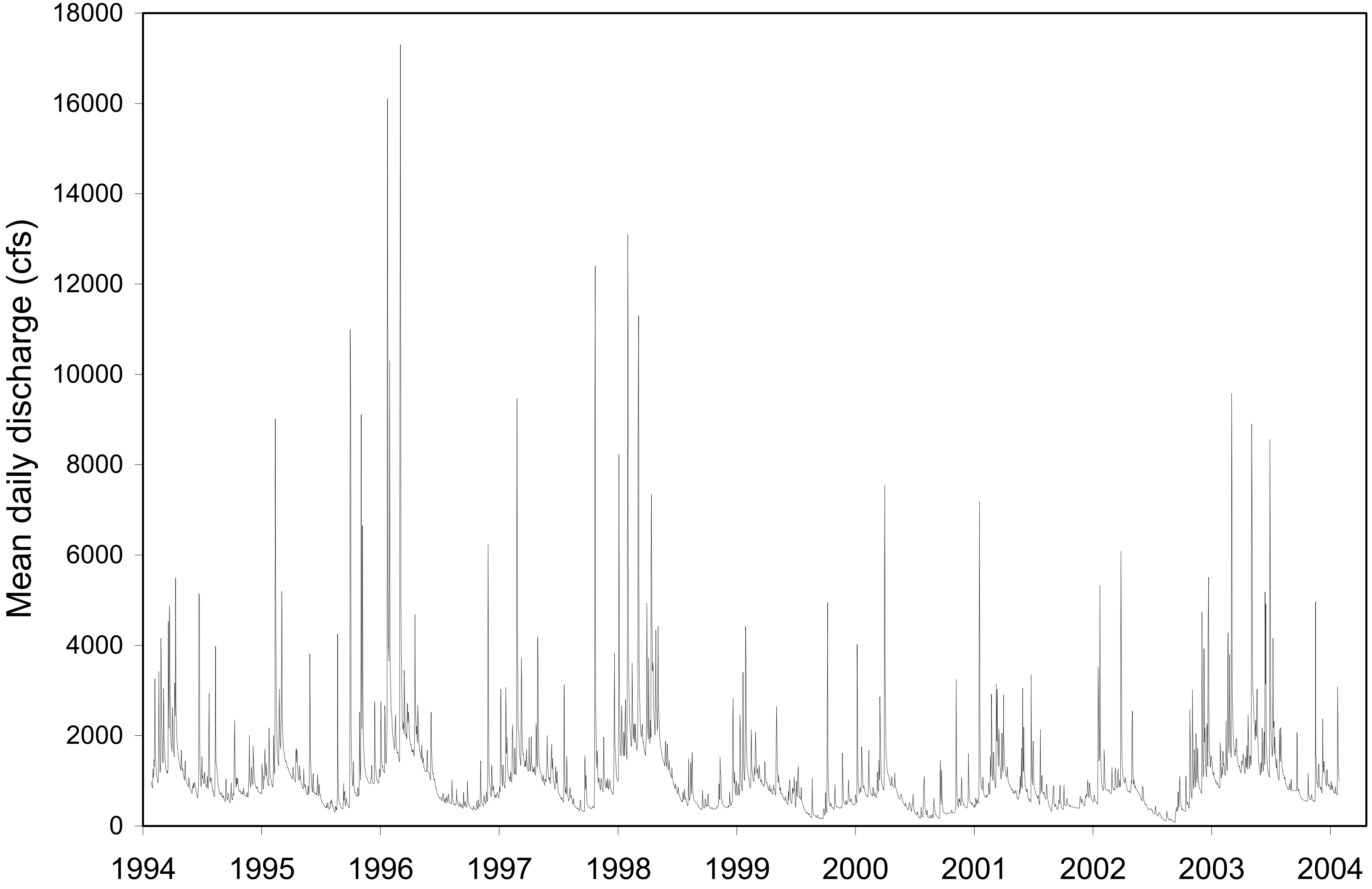
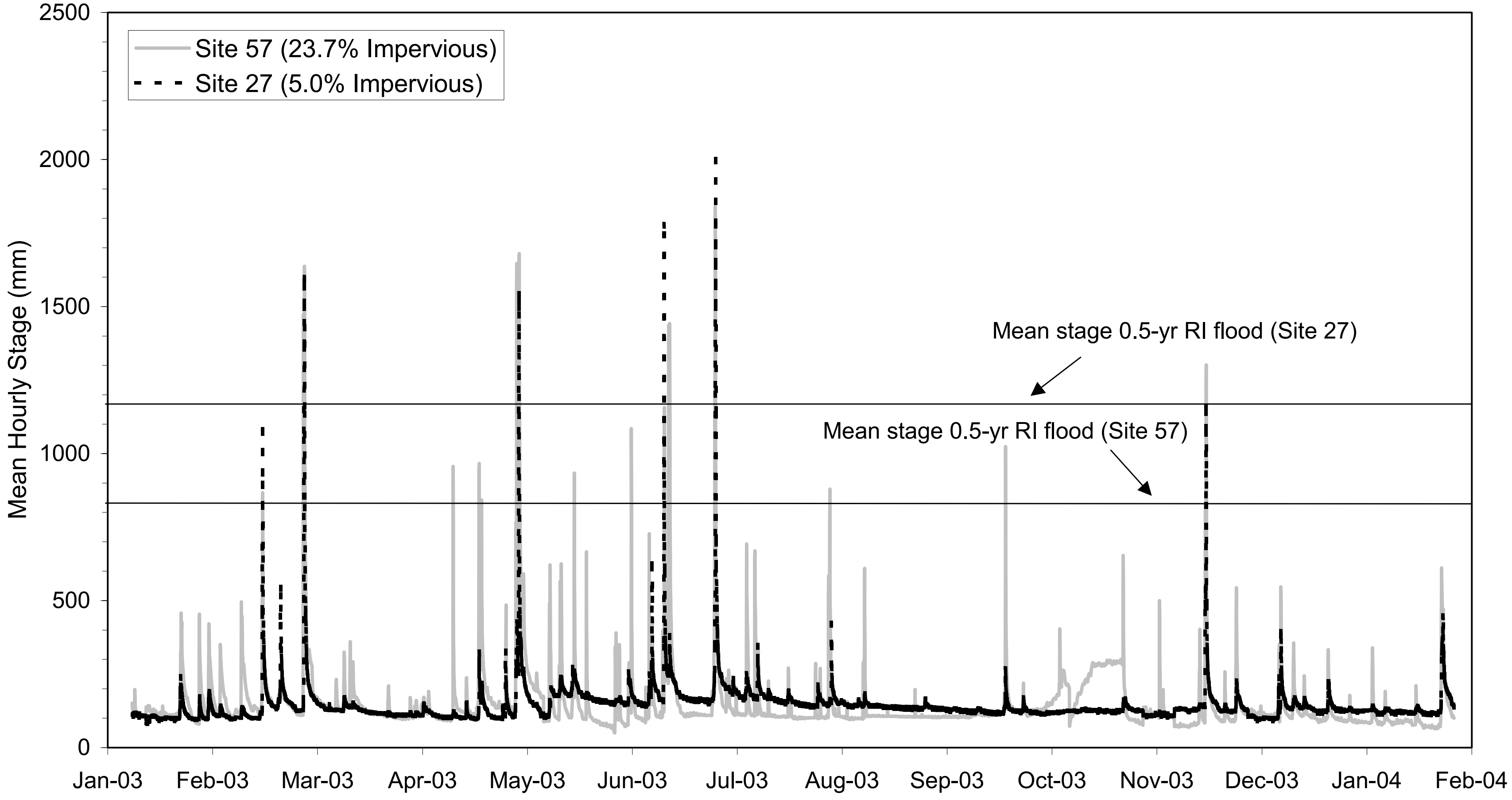

Mean daily discharge (cfs)
18000
16000
14000
12000
10000
8000
6000
4000
2000
0
1994
1995
1996
1997
1998
1999
2000
2001
2002
2003
2004

Figure 4.4.


Site 57 (23.7% Impervious)
Site 27 (5.0% Impervious)
Mean Hourly Stage (mm)
0
500
1000
1500
2000
2500
Mean stage 0.5-yr RI flood (Site 27)
Mean stage 0.5-yr RI flood (Site 57)
Jan-03
Feb-03
Mar-03
Apr-03
May-03
Jun-03
Jul-03
Aug-03
Sep-03
Oct-03
Nov-03
Dec-03
Jan-04
Feb-04

Appendix 4.1. Fishes collected and frequency of occurrence within the 30 study streams. Habitat preference is either fluvial specialist (FLU) or lentic tolerant (LEN). Highland endemic species (END) and cosmopolitan, widespread species (COS) are indicated (after Walters et al. 2003). Determination of sensitive fishes (SEN) was based on life history traits (i.e. expected sensitivity) and negative relationships with % urban land cover from previous fish collection data (Seth J. Wenger, University of Georgia, *unpublished data*).

Family Name	**Common Name**	**Composition**	**Freq. of**	**Family Name**	**Common Name**	**Composition**	**Freq. of**
Scientific Name		**Categories**	**Occurence**	***Scientific Name***		**Categories**	**Occurence**
Petromyzontidae				**Salmonidae**			
Ichthyomyzon sp. cf. gagei	southern brook lamprey	FLU	12	*Onchorhynchus mykiss*	rainbow trout	n/a	1
Cyprinidae				**Fundulidae**			
Campostoma oligolepis	largescale stoneroller	FLU, COS	29	*Fundulus stellifer*	southern studfish	FLU	24
Cyprinella callistia	Alabama shiner	FLU, SEN	12	**Poecilidae**			
Cyprinella trichoristia	tricolor shiner	FLU, END, SEN	3	*Gambusia affinis*	eastern mosquitofish	LEN, COS	6
Cyprinella venusta	blacktail shiner	FLU	5	*Gambusia holbrooki*	western mosquitofish	LEN, COS	2
Hybopsis lineapunctata	lined chub	FLU, END	1	*Gambusia holbrooki x affinis*	hydrid mosquitofish	LEN, COS	4
Hybopsis sp. cf. winchelli	clear chub	FLU, END	2	**Cottidae**			
Luxilus zonistius	banfin shiner	FLU	2	*Cottus carolinae zopherus*	Coosa banded sculpin	FLU, END	23
Nocomis leptocephalus	bluehead chub	FLU	5	**Centrarchidae**			
Notemigonus crysoleucas	golden shiner	LEN, COS	1	*Ambloplites ariommus*	shadow bass	FLU, COS	1
Notropis chrosomus	rainbow shiner	FLU, END, SEN	2	*Lepomis auritus*	redbreast sunfish	LEN, COS	30
Notropis longirostris	longnose shiner	FLU, COS	3	*Lepomis cyanellus*	green sunfish	LEN, COS	22
Notropis lutipinnis	yellowfin shiner	FLU	3	*Lepomis gulosus*	warmouth	LEN, COS	6
Notropis stilbius	silverstripe shiner	FLU, SEN	4	*Lepomis macrochirus*	bluegill sunfish	LEN, COS	29
Notropis xaenocephalus	Coosa shiner	FLU, END	15	*Lepomis microlophus*	redear sunfish	LEN, COS	5
Phenacobius catostomus	riffle minnow	FLU, END, SEN	1	*Lepomis punctatus*	spotted sunfish	LEN, COS	9
Semotilus atromaculatus	creek chub	FLU, COS	23	*Lepomis macrochirus x auritus*	hybrid sunfish	LEN, COS	1

Appendix 4.1. Continued.

Catastomidae			
Hypentelium etowanum	Alabama hog sucker	FLU	30
Minytrema melanops	spotted sucker	FLU, COS	3
Moxostoma duquesnei	black redhorse	FLU, COS, SEN	10
Moxostoma erythrurum	golden redhorse	FLU, COS	8
Moxostoma poecilurum	blacktail redhorse	FLU, COS	3
Ictaluridae			
Ameiurus brunneus	snail bullhead	FLU, COS, SEN	7
Ameiurus natalis	yellow bullhead	LEN, COS	5
Ameiurus nebulosus	brown bullhead	LEN, COS	2
Ictalurus punctatus	channel catfish	LEN, COS	3
Noturus leptacanthus	speckled madtom	FLU, COS	5
Micropterus coosae	Coosa bass	FLU	23
Micropterus punctulatus	spotted bass	FLU, COS	4
Micropterus salmoides	largemouth bass	LEN, COS	22
Pomoxis annularis	white crappie	LEN, COS	1
Pomoxis nigromaculatus	black crappie	LEN, COS	2
Percidae			
Etheostoma scotti	Cherokee darter	FLU, END	18
Etheostoma stigmaeum	speckled darter	FLU, COS, SEN	10
Perca flavescens	yellow perch	LEN, COS	2
Percina kathae	Mobile logperch	FLU	14
Percina nigrofasciata	blackbanded darter	FLU, COS	27
Percina palmaris	bronze darter	FLU, END, SEN	1

CHAPTER 5

IMPORTANCE OF STREAMSIDE RIPARIAN FORESTS IN URBAN AREAS CONTINGENT ON SEDIMENT AND HYDROLOGIC REGIME[1]

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[1] Roy, A.H., M.C. Freeman, B.J. Freeman, W.E. Ensign, S.J. Wenger, and J.L. Meyer. To be submitted to *Environmental Management*.

Abstract

Forested riparian areas have been used extensively in the last decade to minimize impacts of landscape disturbance on stream ecosystems, yet little is known about the effectiveness of streamside forests at mitigating disturbance in urbanizing watersheds. In 30 small streams (8-20 km^2) along a gradient of urban disturbance (1-65% urban land cover), we tested the hypothesis that response of fish assemblage integrity to riparian forests depends on the relative degree of sediment or hydrologic alteration. Species expected to be sensitive to disturbance (i.e. fluvial specialists and "sensitive" species that respond negatively to urbanization) were best predicted by models including % forest cover in riparian area and a principal components axis describing sediment disturbance. Only sites with coarse bed sediment and low bed mobility (vs. sites with high levels of sediment disturbance) had increased richness and abundances of sensitive species with higher % riparian forests, supporting our hypothesis that response to riparian forests is contingent on the level of sediment disturbance. Abundances of *Etheostoma scotti*, the federally threatened Cherokee darter, were best predicted by models with single variables representing stormflow disturbance ($r^2 = 0.34$) and sediment disturbance ($r^2 = 0.23$). Lentic tolerant species responded only to a variable representing prolonged duration of low flow conditions, explaining 65 % of the variance in richness and 29 % of the variance in abundance. For these species, hydrologic alteration overwhelmed any influence of riparian forests on stream biota. These results suggest that, at a minimum, watershed management strategies must simultaneously address hydrologic, sediment, and riparian disturbance in order to protect all aspects of fish assemblage integrity. Because most of these disturbances were related to catchment % forest and % urban land cover, we recommend addressing management at the scale of the catchment where most disturbance is initiated. Minimizing impervious surface cover and disconnecting

impervious surface from hydrological pathways to allow for natural processes to occur should be effective prescriptions for allowing development while maintaining healthy fish assemblages.

Introduction

Urban development and associated increases in impervious surface cover can drastically alter geomorphology, hydrology, water quality, and function of stream ecosystems, resulting in subsequent biotic impairment (see reviews Schueler 1994, Paul and Meyer 2001). Comparisons among streams along gradients of urban land cover suggest that urbanization can alter richness, diversity, density and biotic integrity of fish assemblages (Steedman 1988, May et al. 1997, Wang et al. 1997 & 2001, Snyder et al. 2003). In the southeastern United States, homogenization of fish assemblages, as indicated by changes in the ratio of highland endemic to widespread cosmopolitan species, has been documented with increased urbanization (Scott and Helfman 2001, Walters et al. 2003a). Landscape development has also been associated with diet shifts of generalist fishes with foraging flexibility (Weaver and Garman 1994; M.J. Paul, Howard University, *pers. comm.*). These community- and species-level changes in fish assemblages appear to occur at relatively low levels of urbanization (e.g. 10-15% impervious surface cover; Schueler 1994, Wang et al. 2001, Miltner et al. 2004).

This study took place in the Etowah River basin, an area that harbors rich biological diversity and is threatened by rapid sub-urbanization from metropolitan Atlanta (Burkhead et al. 1997). Of the 76 native fish species in the basin, 8 are protected by the federal Endangered Species Act or listed as impaired under the Georgia Wildlife Protection Act. Concern over protection of these species has led to development of an Etowah Regional Aquatic Habitat Conservation Plan (HCP; http://www.etowahhcp.org/). In order to guide management within the basin, researchers are working to identify critical stressors to the fish assemblages. Previous research by Walters et al. (2003b) found that decreases in bed texture in low-slope streams led to increased fish impairment in small and medium-sized streams in the Etowah (11-126 km^2). In addition to altered sediment regimes, Roy et al. (*in review*) found that increases in magnitude and frequency of storm events and prolonged duration of low flow conditions resulted in reduced numbers of sensitive fishes and increased numbers of tolerant fishes.

Managers have an extensive toolbox for minimizing the effects of development on stream ecosystems; however, many of these techniques have been employed inconsistently and with variable effectiveness within watersheds. In the United States, sedimentation remains the second most cited cause of impairment to streams and rivers, despite erosion and sedimentation ordinances which aim to control sediment from construction sites (US EPA 2000). Best management practices to protect streams from stormflow impairment act to minimize peak flows in large storms, but many do not adequately address other aspects of hydrologic alteration, such as storm volume and infiltration (Andoh and Declerck 1997). Further, the inconsistent application of these management tools (e.g. on a project-to-project basis) within watersheds does not allow for complete watershed protection.

Forested riparian areas have also been used extensively throughout the United States to protect aquatic resources (Lowrance 1998, Pusey and Arthington 2003); however, much of our knowledge of the function of riparian forests stems from research in areas dominated by agricultural and silvicultural activities. The translation of this management tool to urban areas where it has unknown benefits can lead to ineffective protection (at best) or false assumptions that riparian forests can mitigate the negative effects of non-point source stressors (at worst). Researchers have predicted that effectiveness of streamside forests will depend on landscape context and upland stressors (Roth et al. 1996, Naiman and Decamps 1997, Lowrance 1998), yet there have been few empirical tests of this idea, particularly in urban settings (but see Hession et al. 2003). Since sediment and hydrologic alteration have been identified as critical stressors for fishes in the Etowah River basin, we asked whether the capacity of riparian forests to buffer upland disturbance was contingent on the level of 1) sediment or 2) hydrologic alteration in the stream. Specifically, we predicted that fish assemblages in streams that have been impaired by high amounts of fine sediment would not be influenced by the extent of riparian forests, whereas streams with coarse bed texture would have a range in fish assemblage integrity based on the extent of riparian forest cover. Similarly, we expected riparian forests to provide minimal benefits for fish assemblages in streams that are highly impaired by hydrologic alteration. This information will be used to guide policymakers interested in determining what components of watershed management are necessary for maintaining healthy fish assemblages.

Methods

Study Sites

Thirty small streams (8-20 km^2) within the Piedmont physiographic region of the Etowah River basin in north-central Georgia, USA were selected for this study. Streams were selected to encompass an expected range of hydrologic alteration. We stratified sites by 1) % impervious surface cover (<10%, 10-20%, >20%), and 2) baseflow yield determined in the field at one sample date. For sites with 10-20% and > 20% impervious surface cover, we randomly selected 5 streams above the median baseflow yield and 5 streams below the median baseflow yield within each impervious surface category. For sites with <10% impervious surface cover, there was geographic pattern of higher baseflow yields in the northeast portion and lower yields at sites in the southwest portion of the Etowah basin, potentially reflecting natural differences in soils or geology. Thus, in the <10% impervious surface class, we randomly selected 5 sites located in the northeast (most of which were above the median baseflow yield) and 5 sites located in the southwest (mostly below the median baseflow yield) portions of the basin. Sites with known point sources of impairment or impoundments across all upstream tributaries were excluded from selection.

Fish sampling

Fishes were sampled in August-October 2003 using a backpack electroshocker (Model 12-B; Smith-Root® Inc), 8-ft. seine, and dip nets. Block nets were set every 50 m for a 150 m reach in each stream. The 3 adjacent 50 m reaches were sampled in a single pass, and 1 randomly selected reach of the 3 was sampled with 3 consecutive passes. Fishes were identified, measured, and released in the field or euthanized with buffered MS-222 and preserved in ~8% formalin for identification in the laboratory.

The program CAPTURE® (White et al. 1978) was used to calculate richness estimates using species detectability based on species caught in single-pass samples in 3 consecutive 50 m reaches. We used model M(h), which assumes heterogeneity of capture probabilities among species, to estimate species richness (Williams et al. 2002). The removal function in CAPTURE® was used to calculate capture probabilities for each species that declined in abundance among the three passes conducted in one 50 m reach. These capture probabilities were used to estimate fish abundance for each species at each site. For species that did not have depletion among passes we were unable to estimate abundance, so we used the total number of individuals captured. We note that our abundance estimates potentially are biased to an unknown extent because, even for taxa exhibiting depletion, we had to assume that individual capture probabilities remained constant across passes. This was an untested assumption.

Fish assemblage structure was evaluated based on richness and abundance of fish assemblage subsets: sensitive species, fluvial specialist species, and lentic tolerant species (Appendix 5.1). Sensitive species were those species found in the Etowah River basin that were expected to be sensitive to disturbance (due to specific life history or habitat requirements), and exhibited a negative response to increased urban land cover based on fishes sampled at other sites in earlier studies (S. J. Wenger, University of Georgia, *unpublished data*). Fluvial specialists are species that require lotic environments for at least part of their life cycle; we classified these species using Etnier and Starnes (1993) and Mettee et al. (1993). Lentic tolerants were fishes that are habitat generalists, capable of completing their life cycle in lakes or reservoirs. We hypothesized that lentic tolerants would increase and fluvial specialists would decrease with increased disturbance. We also assessed the abundance of Cherokee darters, *Etheostoma scotti*, the only federally threatened fish species that is commonly found in small streams where our sampling took place.

Landscape

Land use/cover (hereafter referred to as land cover) was quantified using ArcView© 4.0 Geographic Information Systems (GIS). *Landsat* Thematic Mapper (TM) satellite imagery was used to calculate percentages of land cover categories within sub-catchments for 2001 and land cover change for 1992-2001. Land cover was categorized as urban (high density and low density urban), agriculture (cultivated/exposed land and cropland/grassland), forest (evergreen, deciduous and mixed forests, and forested wetlands), and open water. We also determined percent impervious cover for each sub-catchment from a classified dataset created by the

Georgia Land Use Trends Project (GLUT, Natural Resources Spatial Analysis Laboratory, Institute of Ecology, University of Georgia, Athens, GA, USA). See Roy et al. (*in review*) for a detailed explanation of impervious cover classification.

A drainage network created from Digital Elevation Models (DEMs), which was similar to a 1:24,000 scale stream network, was used to create 30 m buffers (60 m corridor) around the entire drainage network upstream of each site. Riparian forest cover was calculated as the percent forest within this 30 m buffer.

We also calculated the number of road crossings and impoundments per km stream length in the network. Road crossings were mapped by overlaying 1999 Georgia Department of Transportation road coverage with the drainage network. Impoundments were mapped from 1999 digital aerial ortho-photos.

Geomorphology

Ten variables intended to describe the habitat conditions for fishes were measured in each stream reach (see Table 5.1). Velocity, depth, and modal bed sediment size (phi, for 0.5-m^2 area) were measured at 70 equidistant points along the thalweg. Rugosity, a measure of complexity of the channel bottom (e.g. highest rugosity = highest complexity), was determined by dividing the length of a heavy chain laid along the bottom of the stream divided by the wetted width of the stream, and averaged for 10 random transects along the stream reach (Garcia-Charton and Perez-Ruzafa 2001). Length of stream designated as riffle habitat was measured to determine the percent riffle habitat in each stream reach. Percent fines in riffle habitats was determined by collecting 3 liters of bed sediment from each of 3 riffles. Samples were dried, sieved, and weighed in the lab to determine mean % fines (< 2 mm) for each stream.

Steady-flow analysis in HEC-RAS® (Version 2.2, Hydrologic Engineering Center, U.S. Army Corps of Engineers) was used to determine the mean hydraulic depth, mean velocity, and Froude number (a descriptor of main channel flow) for the 0.5-yr recurrence interval (RI) flood at each site. Discharges for the 0.5-yr RI flood were calculated based on basin area at each site using flood-frequency formulas derived for rural streams in the Georgia Piedmont (Stamey and Hess 1993). Manning's n, stream slope, and cross-sectional area at an AquaRod® water level sensor were determined from field measurements and incorporated into the HEC-RAS® model for each stream. A Topcon® AT-F6 level and stadia rod were used to obtain elevations for a channel cross-section at the location of the AquaRod® and for calculating the energy grade line slope between riffle-tops for a 150-m reach. Bed mobility was calculated as a ratio of the mean velocity for the 0.5-yr RI flood divided by critical velocity to move the mean thalweg bed sediment size (Gordon et al. 1992).

Hydrology

Streams were gauged at the base of each watershed during 2003 using 2-m AquaRod® water level sensors (Advanced Measurements & Controls, Inc., Woodinville, WA, USA), which use electrical capacitance to measure stage height. Hydrologic variables were calculated for summer (15 May-7 August, a high flow period) and autumn (15 August-4 November, a low flow period) at 16 sites where we had complete continuous data. We calculated 9 baseflow and 18 stormflow variables that we expected to respond to impervious cover and that could affect fish assemblages. Baseflow variables included minimum daily stage, minimum 7-day mean stage, minimum 7-day maximum stage, and magnitude and duration of low stage events below 25%, 10% and 5% of the median stage. Baseflow magnitude variables were divided by the mean daily stage to adjust for differences in stream size and gauge location. Stormflow variables included frequency (i.e.

number of flow excursions above a certain stage), magnitude, duration, and volume (stage height * hour) during events above a certain stage, and rate of change associated with the ascending and descending limbs of storms. Proportions (100%, 75%, and 50%) of the mean stage of the 0.5-yr RI flood divided by the mean daily stage (to adjust for differences in gauge location across sites) were used to calculate stormflow variables.

For each season, we ran principal components analyses (PCA) separately on baseflow and stormflow variables to reduce the variables to a useful metric describing seasonal hydrologic alteration. We represented stormflow alteration using summer stormflow PCA 1, which included a combination of all variables and explained 54.8% of the variation across sites (Roy et al. *in review*). Summer (May-August) was selected because it was the time of year with highest abundances of spawning and young of year fishes when stormflows may have the strongest effects on fish assemblages. Autumn baseflow alteration was represented at the low flow time of year when we expected the strongest influence of baseflow conditions on fish assemblages. Baseflow PCA 2 reflected increased duration of low flow conditions, and was previously shown to explain variation in some fish assemblage metrics (Roy et al. *in review*).

Data analyses

All variables were tested for normality using Shapiro-Wilk goodness of fit test and transformed when necessary. All fish abundance metrics were transformed using log(x+1) and percentage variables were transformed using arcsin(sqrt(%/100)).

PCA was used to reduce geomorphic variables into a single metric describing sediment alteration. We used multiple linear regression analysis to predict fish assemblages using PCA axis scores for sediment, summer stormflow, and autumn baseflow. We compared 5 least-squared models including single variables of % riparian forest cover and sediment PCA, addition of the two variables, and % riparian forests or sediment PCA plus an interaction term between % riparian forests and sediment PCA to predict various fish assemblage metrics based on 30 sites. For hydrologic analyses, we used a subset of 16 sites (where we had complete hydrologic data), and similarly compared models of single variables and various combinations of % riparian forest cover, stormflow PCA 1, baseflow PCA 2, riparian by stormflow interaction, and riparian by baseflow interaction (total of 9 models). Although non-normal variables were transformed, thus increasing the linearity of models, this approach necessarily assumes that variables are linearly related.

We used an information-theoretic approach to determine the best-supported model from the group (sediment or hydrologic model sets) for predicting each fish assemblage metric. In contrast to hypothesis testing, this approach avoids overfitting models by identifying scientifically appropriate models *a priori*, and evaluates the relative support for each model within a set of plausible models based on model likelihood and parameters measured (Burnham and Anderson 2002). Akaike's Information Criterion, adjusted for small sample size (AIC_c) was calculated to assess fit of candidate models, with lowest AIC_c indicating the best-supported model (Burnham and Anderson 2002). AIC_c seeks the simplest, best-supporting model by incorporating a penalty for inclusion of additional variables into the model. Akaike weights (w_i) were computed as $w_i = \exp(-½\Delta_i)/\sum\exp(-½\Delta_i)$, where Δ_i equals the difference in AIC_c for each model compared to the best-supported model (i.e.$\Delta_i = 0$ for best-supported model) and the denominator is a sum of $\exp(-½\Delta_i)$ for all models in the set. We used Akaike weights (which vary from 0 to 1 with the best-fitting model having the highest weight) to measure the weight of evidence for each model given the data (Burnham and Anderson 2002). We summed weights across models that included each variable to estimate the relative variable importance. Although adjusted R^2 values provide useful information about the variance explained in a model, we feel that AIC_c is the best approach to compare candidate models and determine the best-supporting model relative to the model set. Because AIC_c assumes that a favorable model exists within the model set, we did not compare candidate models within a group if all the models exhibited poor fit (i.e. low adjusted R^2).

Correlation analysis (Pearson's r) was used to relate riparian, sediment, and hydrologic variables to stream slope and catchment land cover and land cover change. Sites were also divided into categories based on sediment regime. Sites that had highest % fines in riffles and bed mobility (within top 40 % of sites) and had the lowest sediment PCA values were considered to have high sediment alteration (n=9 sites). Sites that had lowest % fines in riffles and bed mobility (within bottom 40 % of sites) and had the highest sediment PCA values were considered to have low sediment alteration (n=9 sites). If the best-supported linear regression models included an interaction between riparian forest cover and sediment, then regressions between fishes and riparian variables were analyzed separately for the sediment alteration categories. All analyses were performed using JMP® Version 4.0 statistical software (SAS Institute Inc., Cary, NC, USA).

Results

Sediment alteration and riparian forests

Land cover in the catchments was dominated by % urban and % forest land uses, and sites exhibited the highest variation in these land cover types (Table 5.2). Percent riparian forest cover reflected similar differences in land cover, ranging from 39 to 100% across sites. Sites also exhibited a range in fish assemblages, with an average estimated richness of 17.4 species and abundance of 184.9 individuals per 150-m stream reach (Table 5.2).

Sites exhibited a range in % fines (1.5-89.0%) and variability in % fines (0.9-26.1%) in riffles. Bed mobility ranged from 0.7 to 28.0, with 26 of the 30 sites able to move the mean bed sediment size with the 0.5-yr RI flood (i.e. bed mobility > 1, Table 5.1). When combining the 10 geomorphic variables using principal components analysis, the first principal component explained 32.2% of the variance in geomorphic variables across sites, and the eigenvector was heavily weighted on variables reflecting the bed sediment size and variability, amount of riffle habitat, and bed mobility (Table 5.1). This first principal component, hereafter referred to as "sediment PCA," increased with increased sediment disturbance, as reflected by increased % fines, increased bed mobility, and decreased variability in bed sediment size.

Richness and abundance of sensitive fish species was best explained by a model including % riparian forest plus an interaction between % riparian forest and sediment PCA (Table 5.3). The model of % riparian forest plus sediment PCA (i.e. linear addition rather than an interaction term) was also a strong model. Akaike weights suggest that models with sediment PCA alone or sediment PCA plus an interaction term with % riparian forest were not as well supported. Although an interaction between % riparian forest and sediment PCA explained the highest amount of variation in the richness and abundance of fluvial specialist species (i.e. highest adjusted R^2), a model with just % riparian forest was slightly better supported. The best supported model for abundance of *E. scotti* was with sediment PCA alone, while the model with % riparian forest alone was least supported (Table 5.3). Richness and abundance of lentic tolerant species were not predicted well by any of the models with % riparian forest and sediment PCA (i.e. low adjusted R^2), so we did not compare fit among models.

We added the weights of all models with % riparian forest (4) and all models with sediment PCA (4) included to compare the relative importance of these variables in explaining fish assemblage metrics. The total weights were similar (ratios of the highest to lowest ranging from 1.1 to 1.5), suggesting that % riparian forest and sediment PCA variables are equally important in predicting richness and abundance of sensitive species and fluvial specialist species, and abundance of *E. scotti* (Table 5.4).

Hydrologic alteration and riparian forests

For a subset of 16 sites for which we had complete hydrologic data, we analyzed the relative importance of stormflow and baseflow alteration separately and in combination with % riparian forest cover to test whether relationships between % riparian forest and fishes were contingent on levels of hydrologic alteration. Richness of lentic tolerant species was best predicted by autumn baseflow PCA 2, and these models were weighted at least 3 times stronger than any other models. This provides strong evidence that baseflow alteration is the best-supported model among the candidate models for explaining variation in lentic tolerant species richness within this dataset (Table 5.5). When comparing the summed weights of 4 models with stormflow vs. 4 models with baseflow, there was substantial evidence that baseflow was more important than stormflow in predicting richness of these fish assemblages (i.e. combined weights of all baseflow PCA 2 models were 7.8 times higher than weights of stormflow PCA 1 models, Table 5.4). Abundance of lentic tolerant species was equally well-predicted by % riparian forest and baseflow PCA; however, there was little support for including an interaction term in the models (Table 5.5).

Richness and abundance of sensitive species were best explained by the model with stormflow PCA 1 (Table 5.5). Combined weights for stormflow and baseflow models revealed that stormflow models were 3.3 and 2.0 times better supported than baseflow models, respectively (Table 5.4). Abundance of *E. scotti* was also best explained by stormflow PCA 1 relative to baseflow PCA 2, % riparian forest, or models with combinations of these variables (Table 5.5). Richness and abundance of fluvial specialist species were not well predicted by any of the models including stormflow, baseflow, and/or riparian alteration with this data set (i.e. low adjusted R^2), so we did not compare models using $\mathrm{AIC_c}$.

Importance of sediment, hydrology, and riparian forests for fish assemblages

Due to differences in sample size between sediment and hydrologic model sets, we were unable to directly compare models including sediment PCA, stormflow PCA 1, and baseflow PCA 2 using $\mathrm{AIC_c}$. Thus, we compared adjusted R^2 values for the best-supported models determined by $\mathrm{AIC_c}$ (while accounting for differences in sample size between sediment and hydrologic model sets) to assess whether riparian forest cover, sediment, stormflow hydrology, or baseflow hydrology explained the most variance in fish assemblages. Richness and abundance of sensitive species were best explained by a combination of riparian forest and sediment variables, accounting for 40 and 46 % of the variation in fishes, respectively (Table 5.3, Fig. 5.1). When sites were divided into classes of low and high sediment alteration, regressions indicated a significant positive relationship between % riparian forest and richness and abundance of sensitive fishes at sites with low sediment alteration, but no relationship at sites with high levels of sediment alteration (Fig. 5.1).

Fluvial specialist species were best predicted exclusively by % riparian forests; however, this variable explained a mere 17 % of the variation in richness and abundance in the 30 sites (Table 5.3, Fig. 5.2A-B). Richness and abundance of lentic tolerant species were best predicted by increases in the duration of low flow conditions (baseflow PCA 2, Table 4, Fig. 5.2C-D).

Abundance of *E. scotti* was related to reduced stormflow and sediment alteration (Tables 5.3 & 5.4, Fig 5.2E-F). The relationship with summer stormflow revealed a threshold-type response, where no individuals were found at sites with high levels of stormflow disturbance (Fig. 5.2E). The relationship with sediment revealed a wedge-shaped pattern, indicating that increases in sediment alteration result in an upper limit in the abundance of this species; however, there are other variables explaining low abundances in sites with coarse bed texture (Fig. 5.2F). If sites without *E. scotti* are excluded, sediment explains a higher amount of the variation in abundances of *E. scotti* across sites ($r^2 = 0.35$).

We also asked whether stream slope (a natural factor) or landscape disturbance (anthropogenic factors) were related to riparian forest, sediment, and hydrologic conditions at these sites. Altered sediment condition was related to decreased slope and greater reductions in % forest cover from 1992-2001, but no other land cover variables (Table 5.6). Percent riparian forest was strongly correlated with catchment land cover (% urban, % forest) and changes in land cover (% urban, % agriculture) in the last decade. Stormflow and baseflow alterations were also negatively related to % forest cover and positively related to % urban cover (Table 5.6).

Discussion

Predicting fish assemblage response

Previous studies have shown negative relations between fish assemblage integrity and loss of riparian forest cover in urbanizing areas (Steedman 1988, May et al. 1997). In our study, local scale sediment disturbance (increased % fines, increased bed mobility, decreased variability in bed sediment size) affected the ability of riparian forest cover to predict sensitive fish and fluvial specialist species. Interestingly, these fishes responded primarily to % riparian forest (vs.

sediment regime), although inclusion of an interaction between these variables was necessary to best explain richness and abundance of sensitive species. Increased fines (and other sediment disturbance measures) were found at streams with lowest slopes and greatest loss of catchment forest cover in the last decade. This suggests that both natural factors (slope) and anthropogenic factors (change in forest cover) create reaches impaired by excess fine sediment, subsequently influencing sensitive and fluvial specialist species. Under these conditions, riparian forests will not be sufficient to protect these fishes. Conversely, streams that have coarse bed texture may not harbor high abundances of sensitive species where streams have low (< *ca.* 70 %) riparian forest cover along the upstream network. Thus, maintaining streams with both coarse bed texture and adequate riparian forest cover may be an effective strategy for protecting sensitive fish species.

The impacts of urbanization on hydrologic alteration overwhelmed any response of lentic tolerant species to loss of forest cover. Whereas bed texture is a constraint to fishes at the microhabitat or channel unit scale, and riparian land cover is a constraint at the reach scale, hydrologic effects due to urbanization act at the scale of the basin (Poff 1997). Within this concept of landscape filters, it is not surprising that prolonged low flow conditions (represented by baseflow PCA 2) create a dominant control on fish assemblages, increasing richness and abundance of lentic tolerant species. In small streams, prolonged baseflows may create conditions of depleted oxygen, thus favoring species that are tolerant to oxygen depletion typical of lentic environments (Mulholland et al. 1997). Thus, maintenance of adequate low flows throughout the year will be important to prevent shifting of assemblages to dominance by tolerant species.

This study was designed, in part, to advise local governments toward planning and management in north Georgia that will ensure protection of imperiled fish species as part of the Etowah Regional Aquatic Habitat Conservation Plan (HCP). Within this regional HCP, local governments will adopt policies and amend zoning codes and development regulations so that any new activities will have minimal impact on fish assemblages. In turn, these governments will have the authority to issue incidental take permits to developers in concordance with the U.S. Endangered Species Act. Since the focus of the HCP is on imperiled fish species, stakeholders are primarily interested in how to protect the Cherokee darter, *Etheostoma scotti*, from extirpation within small streams of the Etowah River basin. Our results demonstrate that streams with highest stormflow alteration (the top 5) did not have *E. scotti*. In streams that harbored *E. scotti*, abundances of these fish were reduced with increased sediment disturbance. Although we did not test an interaction between stormflow and sediment disturbance (because of differences in sample sizes for each variable), these results confirm that management must account for both types of disturbance in order to protect populations of *E. scotti*.

Despite the significant relations observed between fish assemblage measures and riparian, hydrologic, and sediment stressors, these relations explain relatively low amounts of variation in the fish data. For example, the best-supported model for richness and abundance of fluvial specialist species was with % forested riparian cover, explaining 20 and 17 % of the variation in fish metrics, respectively. Even when including an interaction with sediment, three quarters of the variation was still unexplained. These weak relations are typical of studies relating in-stream biological or habitat parameters to large-scale disturbances (e.g. Roth et al. 1996, Wang et al. 2001, Walters et al. 2003a, Miltner et al. 2004). Empirical modeling using data with known limitations (e.g. inability to accurately assess richness and abundance) has been criticized for oversimplifying complex ecological systems. However, this technique is one of the best available approaches to forecasting biological responses to environmental threats (Nilsson et al. 2003). These weak relations suggest that natural and/or anthropogenic stressors other than riparian deforestation and altered hydrologic and sediment regimes (e.g. man-made impoundments) are influencing these fish assemblages.

Role of riparian forests

Many studies relating biotic integrity to landscape disturbance cite riparian buffers as critical components of stream mitigation. The importance of riparian buffers for stream mitigation has been successfully translated into legislation; many state and local regulations require riparian buffers for gaining permits to change land use adjacent to streams (Lee et al. 2004). However, Naiman and Decamps (1997) suggest that the role and extent of influence of riparian areas on aquatic systems is dependent on landscape context and associated upland stressors. In other words, riparian forests may only be important to maintain high-quality habitat if upstream processes do not have an overwhelming impact on biotic assemblages (Roth et al. 1996, Hession et al. 2003). Our study provided empirical data to support this idea, suggesting that stream protection using riparian buffers exclusively will not be sufficient to maintain healthy fish assemblages.

Although sediment, hydrology, and riparian forests were considered separately in these analyses, they are necessarily dependent on each other. Altered hydrology can increase bank erosion, acting as a source of sediment disturbance, while large, scouring flows associated with hydrologic alteration can also transport fine sediments, acting to increase streambed coarseness (Booth and Jackson 1997, Finkenbine et al. 2000). Further, riparian forests have been used to trap sediment and slow down overland flows (Lowrance et al. 2000). Riparian forests can offset sediment and hydrological disturbances only if stormwater conveyances do not short-circuit hydrologic pathways, effectively bypassing riparian buffers.

The function of riparian forests in urban areas depends on the amount of contact with overland and subsurface flow, and the effectiveness of riparian vegetation at "treating" this flow and providing quality organic matter for aquatic systems. When transport through pipes and gaps in riparian forests dominate (e.g. at road crossings and gullies), riparian forests provide minimal benefits to streams in terms of minimizing sediment and hydrologic disturbance. In addition to limited hydrological effectiveness, riparian areas in urban landscapes also differ in the quality of vegetation (e.g. higher amounts of invasive plants), potentially altering the capacity to provide important functions for stream ecosystems (Naiman and Decamps 1997). For example, studies have shown that invasive plants that differ in quality from native plants will have different leaf breakdown times, subsequently affecting the aquatic food web (Albarino and Balseiro 2002). Channel incision and widening associated with urbanization also limits interaction between stream channels and bank habitats, minimizing potential benefits of riparian areas due to their proximity to stream water (Wissmar et al. 2003). Hession et al. (2003) noted that the influence of riparian forests and catchment urbanization on stream ecosystems are complex and variable, and will depend on landscape characteristics (e.g. relief, upstream stressors) and the attributes of stream ecosystems measured. Even when riparian forests seem to function properly, it is difficult to tease apart the mechanism of benefit due to the complexity of multiple factors affecting streams within the urban landscape (Pusey and Arthington 2003).

Because watershed urbanization is the dominant landscape filter affecting these stream fishes, maintenance of riparian forests will necessarily be beneficial since this area is part of the catchment land cover. However, the tight correlation between catchment and riparian forest cover ($r = 0.95$) does not allow us to distinguish between the importance of maintaining forests in upland catchments vs. riparian areas adjacent to streams. In urbanizing landscapes that have a mosaic of land cover types within each basin, the location of forests within a catchment may not be critical for stream protection, as long as adequate forest cover is maintained throughout the catchment. In efforts to protect streams, we must focus on sources of impairment at the scale of impairment (i.e. catchment), rather than assuming that riparian forests will provide adequate stream protection regardless of upstream uses.

Holistic approach for watershed management

Because multiple stressors influence fishes, maintenance of various aspects of fish assemblage integrity will require a holistic approach to watershed management. At a minimum, this must include:

1. Managing stormwater to reduce the frequency, magnitude, volume, and duration of peak flows and reduce flow “flashiness”;
2. Maintaining adequate low flows in streams throughout the year by infiltrating precipitation and minimizing water diversions;
3. Reducing fine and unstable bed sediments in streams by enforcing erosion and sediment control on construction sites and minimizing hydrologic connection to streams; and
4. Minimizing impervious cover and maintaining forest cover in catchment and riparian areas.

Although structural solutions are available to address many of these management issues (e.g. detention basins), minimization of source impacts will be the most effective and potentially least costly solution (Andoh and Declerck 1997). For example, clustering development to minimize impervious surface cover (Arnold and Gibbons 1996, Booth et al. 2002) and disconnecting drainage systems (Walsh 2004) should allow development to occur with less loss of assemblage integrity than conventional development.

It is important to acknowledge that even small transformations of the natural landscape may result in aquatic impacts, and maintaining healthy biotic assemblages will necessarily require limits to impervious surface cover (Booth et al. 2002). Stakeholder appreciation of essential services provided natural stream ecosystems is necessary to connect these scientific findings to management solutions (Palmer et al. 2004). For the Etowah Regional Aquatic HCP, the ultimate goal is to maintain viable populations of the threatened or endangered species (and any species likely to become threatened or endangered). Since many of these species reside in the main stem of the Etowah River, protection of the entire watershed, especially the headwater regions, will be critical. The HCP may also include prioritizing watersheds for protection: maximizing protection in some sub-catchments while allowing for development to occur in areas that do not harbor sensitive fish species. For protection of small stream fishes such as *E. scotti*, incorporation of stormwater management, sediment controls, and riparian forests, among other management tools, must be considered.

At a time when development on the urban fringe is inevitable, planners must look to design urban land in ways that will minimize impact on ecological systems by mimicking natural processes or allowing natural processes to occur (Lloyd et al. 2002). Many have suggested a degradation threshold of ~10 % impervious surface cover, but poorly designed urban land use can have impacts on aquatic systems at levels much less than 10 % impervious cover (Walsh 2004). Conversely, greater development may be possible if we can disconnect stormwater pathways (Walsh 2004) and maintain functional riparian buffers (May et al. 1997). This offers opportunities to creatively design stormwater management on the fringe of urban development to treat stormwater as a resource and encourage open space greenways (Tourbier 1994, Lloyd et al. 2002).

Our empirical results suggest that exclusive use of riparian forests to mitigate impacts of urban disturbance will have minimal benefits, if not coupled with other management tools. These results will be used to inform ordinances and other policies in concordance with the Etowah Regional Aquatic HCP. The effectiveness of these policies at maintaining healthy fish assemblages will continue to be tested via adaptive management (Wilhere 2002). For this basin and likely other watersheds threatened by urbanization, this research has highlighted the importance of combining multiple management tools to protect fish assemblage integrity.

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Table 5.1. Description of habitat variables used in principal components analysis (PCA). The first principal component, hereafter referred to as "sediment PCA," explained 32.2% of the variation in habitat variables. Numbers in bold indicate large weightings on the first eigenvector.

Habitat variable	Eigenvector #1	Description
Velocity variability ($m\ s^{-1}$)	-0.070	standard deviation of velocity ($m\ s^{-1}$) at 70 points along thalweg for reach
Bed sediment variability (phi)	**-0.343**	standard deviation of modal bed sediment size (phi)[a] measured at 70 0.5-m^2 plots along thalweg for reac
Depth variability (m)	0.206	standard deviation of depth (m) at 70 points along thalweg for reach
Rugosity	0.209	length of wetted surface divided by water width, average of 10 transects
% Riffle habitat	**-0.388**	% of total reach length designated as riffle habitat
Mean % fines in riffles	**0.493**	mean % dry sieved riffle material < 2 mm, average of three 3-L replicates
Variability % fines in riffles	**0.357**	standard deviation % dry sieved riffle material < 2 mm, average of three 3-L replicates
Froude number	-0.140	channel froude number for 0.5-yr recurrance interval flood[b]
Bed mobility	**-0.436**	mean velocity for 0.5-yr recurrance interval flood[b] divided by critical velocity to move mean thalweg phi
Turbidity (NTU)	0.248	mean turbidity (NTU) at baseflow conditions, average of 3-4 dates

[a] Phi > 0 (i.e. particle size < 1 mm) converted to 0 for consistency among observers

[b] Calculated using HEC-RAS®

Table 5.2. Land cover, geomorphology, and fish assemblage summary data for 30 study sites. Land cover was calculated from *Landsat* TM imagery for each sub-catchment. Geomorphic and fish assemblage variables were measured at a 150 m reach at the base of each catchment. Reported richness and abundance data are estimates using CAPTURE® to determine species detectibility from 3 consecutive 50-m reaches and to calculate capture probabilities from 3-pass removal in one 50-m reach, respectively.

Land cover and Geomorphology			Fish assemblage		
	Mean	Range		Mean	Range
Basin area (km^2)	13.6	(8.5 to 19.9)	Total		
Road crossings (no. km^{-1})	0.4	(0 to 1.0)	Richness	17.4	(8 to 25)
Impoundments (no. km^{-1})	0.5	(0.1 to 1.1)	Abundance	184.9	(37 to 512)
% Riparian forest cover[a]	76.3	(39.1 to 100)	Sensitive		
Catchment land cover (2001)			Richness	2.7	(0 to 7)
% Impervious[a]	12.0	(1.7 to 31.0)	Abundance[d]	16.7	(0 to 103)
% Urban[a]	25.2	(1.4 to 65.1)	Fluvial specialist		
% Forest[a]	56.6	(29.0 to 96.4)	Richness	12.3	(5 to 21)
% Agriculture[a]	9.2	(0.1 to 27.3)	Abundance[d]	128.5	(10 to 469)
% Open water[a]	0.8	(0 to 2.0)	Lentic tolerant		
Geomorphology			Richness	5.4	(3 to 11)
Slope (%)	0.39	(0.10 to 0.88)	Abundance[d]	56.9	(2 to 412)
% Riffles[a]	22.8	(0 to 62.3)	*Etheostoma scotti*		
Bed sediment variability (phi)	2.3	(0.6 to 4.6)	Abundance[d]	6.2	(0 to 40)
Mean % fines in riffles[a]	25.3	(1.5 to 89.0)			
Variability % fines in riffles[b]	8.2	(0.9 to 26.1)			
Bed mobility[c]	4.9	(0.7 to 28.0)			

[a] Transformed using arcsin(sqrt(%/100)) for analysis

[b] Transformed using log(arcsin(sqrt(%/100))+1) for analysis

[c] Transformed using y^{-1} for analysis

[d] Transformed using log(x+1) for analysis

Table 5.3. Multiple linear regression models predicting fish assemblages with % forest in 30 m riparian buffer and sediment alteration listed in decreasing order of model support based on Akaike weights (n=30 sites). Richness and abundance of lentic tolerant species were excluded because models indicated poor fit. Adjusted R^2, differences in Akaike's Information Criterion from minimum (Δi), and Akaike weights (w_i) of each model are reported. Bold type indicates best-supported models.

	Adj. R^2	Δ_i	w_i
Sensitive Richness			
Riparian + Riparian*SedimentPCA	**0.402**	**0.00**	**0.41**
Riparian + SedimentPCA	**0.367**	**0.72**	**0.28**
Riparian	**0.221**	**1.24**	**0.22**
SedimentPCA	0.053	3.78	0.06
SedimentPCA + Riparian*SedimentPCA	0.109	5.19	0.03
Sensitive Abundance			
Riparian + Riparian*SedimentPCA	**0.460**	**0.00**	**0.49**
Riparian + SedimentPCA	**0.406**	**1.25**	**0.26**
SedimentPCA	0.201	2.92	0.11
SedimentPCA + Riparian*SedimentPCA	0.295	3.47	0.09
Riparian	0.101	4.44	0.05
Fluvial Specialist Richness			
Riparian	**0.174**	**0.00**	**0.41**
Riparian + Riparian*SedimentPCA	**0.238**	**1.15**	**0.23**
Riparian + SedimentPCA	**0.225**	**1.38**	**0.21**
SedimentPCA	0.002	2.46	0.12
SedimentPCA + Riparian*SedimentPCA	-0.013	4.86	0.04
Fluvial Specialist Abundance			
Riparian	**0.171**	**0.00**	**0.31**
Riparian + Riparian*SedimentPCA	**0.259**	**0.28**	**0.27**
Riparian + SedimentPCA	**0.246**	**0.51**	**0.24**
SedimentPCA	0.032	1.57	0.14
SedimentPCA + Riparian*SedimentPCA	0.006	4.12	0.04
Etheostoma scotti Abundance			
SedimentPCA	**0.201**	**0.00**	**0.34**
Riparian + Riparian*SedimentPCA	**0.292**	**0.63**	**0.25**
Riparian + SedimentPCA	**0.281**	**0.83**	**0.23**
SedimentPCA + Riparian*SedimentPCA	0.194	2.33	0.11
Riparian	-0.001	2.94	0.08

Table 5.4. Total combined Akaike weights (w_i) of all four models with riparian or sediment variables included (n=30) and all models with stormflow or baseflow alteration variables included (n=16). Ratio of highest to lowest total combined weights for variables reported. Hydrology models for fluvial specialist fish species and sediment models for lentic tolerant fish species indicated poor fit (i.e. low adjusted R^2), and thus were not compared using Akaike weights.

Sediment (n=30)			Hydrology (n=16)		
	Total w_i	Ratio w_i		Total w_i	Ratio w_i
Sensitive					
Richness					
Riparian	0.94	1.2	Stormflow	0.66	3.3
Sediment	0.78		Baseflow	0.20	
Abundance					
Sediment	0.95	1.1	Stormflow	0.54	2.0
Riparian	0.89		Baseflow	0.27	
Fluvial specialist					
Richness					
Riparian	0.89	1.5			
Sediment	0.60				
Abundance					
Riparian	0.86	1.2			
Sediment	0.69				
Lentic tolerant					
Richness					
			Baseflow	0.78	7.8
			Stormflow	0.10	
Abundance					
			Baseflow	0.48	1.9
			Stormflow	0.25	
Etheostoma scotti					
Abundance					
Sediment	0.93	1.4	Stormflow	0.70	4.1
Riparian	0.67		Baseflow	0.17	

Table 5.5. Multiple linear regression models predicting fish assemblages with % forest in 30 m riparian buffer, summer stormflow alteration (stormPCA1), and autumn baseflow alteration (basePCA2) listed in decreasing order of model support based on Akaike weights (n=16 sites). Richness and abundance of fluvial specialist species were excluded because models indicated poor fit. Adjusted R^2, differences in Akaike's Information Criterion from minimum (Δi), and Akaike weights (w_i) of each model are reported. Bold type indicates best-supported model(s).

	Adj. R^2	Δ_i	w_i
Lentic Tolerant Richness			
BasePCA2	**0.625**	**0.00**	**0.51**
Riparian	0.438	2.82	0.12
Riparian + BasePCA2	0.621	3.20	0.10
BasePCA2 + Riparian*BasePCA2	0.613	3.35	0.10
Riparian + Riparian*BasePCA2	0.571	4.06	0.07
StormPCA1	0.248	4.83	0.05
Riparian + StormPCA1	0.425	6.09	0.02
Riparian + Riparian*StormPCA1	0.426	6.09	0.02
StormPCA1 + Riparian*StormPCA1	0.203	8.36	0.01
Lentic Tolerant Abundance			
Riparian	**0.236**	**0.00**	**0.28**
BasePCA2	**0.235**	**0.01**	**0.28**
StormPCA1	0.043	1.57	0.13
BasePCA2 + Riparian*BasePCA2	0.245	2.76	0.07
Riparian + Riparian*BasePCA2	0.258	2.91	0.07
Riparian + BasePCA2	0.228	3.19	0.06
Riparian + Riparian*StormPCA1	0.179	3.63	0.05
Riparian + StormPCA1	0.177	3.64	0.05
StormPCA1 + Riparian*StormPCA1	-0.028	5.19	0.02
Sensitive Richness			
StormPCA1	**0.295**	**0.00**	**0.45**
Riparian	0.020	2.29	0.14
BasePCA2	-0.069	2.89	0.10
StormPCA1 + Riparian*StormPCA1	0.248	3.57	0.07
Riparian + StormPCA1	0.247	3.58	0.07
Riparian + Riparian*StormPCA1	0.219	3.83	0.07
Riparian + Riparian*BasePCA2	0.079	4.98	0.04
Riparian + BasePCA2	0.010	5.48	0.03
BasePCA2 + Riparian*BasePCA2	-0.007	5.60	0.03

Table 5.5. Continued.

	Adj. R^2	Δ_i	w_i
Sensitive Abundance			
StormPCA1	**0.158**	**0.00**	**0.36**
Riparian	-0.041	1.47	0.17
BasePCA2	-0.060	1.60	0.16
Riparian + StormPCA1	0.115	3.46	0.06
StormPCA1 + Riparian*StormPCA1	0.098	3.59	0.06
Riparian + Riparian*StormPCA1	0.090	3.66	0.06
BasePCA2 + Riparian*BasePCA2	0.074	3.78	0.05
Riparian + Riparian*BasePCA2	-0.097	4.96	0.03
Riparian + BasePCA2	-0.121	5.11	0.03
Etheostoma scotti Abundance			
StormPCA1	**0.295**	**0.00**	**0.45**
Riparian	-0.017	2.54	0.12
BasePCA2	-0.049	2.76	0.11
Riparian + Riparian*StormPCA1	0.287	3.20	0.09
StormPCA1 + Riparian*StormPCA1	0.272	3.35	0.08
Riparian + StormPCA1	0.270	3.37	0.08
Riparian + Riparian*BasePCA2	-0.088	6.13	0.02
Riparian + BasePCA2	-0.094	6.17	0.02
BasePCA2 + Riparian*BasePCA2	-0.100	6.22	0.02

Table 5.6. Pearson's correlations (*r*) between land cover characteristics of the sub-catchments and sediment PCA (higher with increased alteration), % riparian forest along stream network, and hydrologic variables (higher PCA scores indicate increased alteration). Numbers in parentheses indicate number of sites in analysis. Bold numbers indicate $r \geq 0.40$.

	Sediment PCA (30)	% Riparian Forest (30)	% Riparian Forest (16)	Stormflow PCA 1 (16)	Baseflow PCA 2 (16)
Slope	**-0.54**	0.03	-0.12	**-0.56**	0.20
Road crossings (no. km^{-1})	-0.26	**-0.63**	**-0.65**	**0.48**	**0.59**
Impoundments (no. km^{-1})	-0.30	**-0.44**	-0.37	-0.10	0.16
Catchment land cover (2001)					
% Impervious	0.00	**-0.88**	**-0.93**	**0.69**	**0.71**
% Urban	-0.17	**-0.90**	**-0.97**	**0.69**	**0.66**
% Forest	0.12	**0.95**	**0.93**	**-0.68**	**-0.69**
% Agriculture	-0.07	0.28	**0.50**	**-0.63**	-0.24
% Open water	0.06	**-0.40**	-0.39	0.05	0.33
Land cover change (1992-2001)					
% Urban	0.15	**-0.72**	**-0.77**	**0.47**	**0.63**
% Forest	**-0.48**	0.22	-0.14	-0.30	-0.14
% Agriculture	0.31	**0.56**	**0.55**	-0.23	-0.39
% Open water	0.23	0.10	**0.48**	**-0.45**	-0.16

Figure legends.

Fig. 5.1. Relations between sensitive species richness (A) and abundance (B) and % forest cover in 30 m riparian buffer for upstream network. Lines represent best fit linear regressions for low (●) and high (○) sediment alteration (n=9 sites for each category). Sites were designated as "high sediment alteration" if they were among the top 40% of sites with high % fines in riffles and high bed mobility, and were designated as "low sediment alteration" if they were among the top 40% of sites with low % fines in riffles and low bed mobility.

Fig. 5.2. Linear regressions models (r^2) for fish assemblage measures and independent variables for the best supported model (based on Akaike weight comparisons). Relations with % riparian forest and sediment PCA include 30 sites, and relations with hydrologic variables include 16 sites.

Fig. 5.1

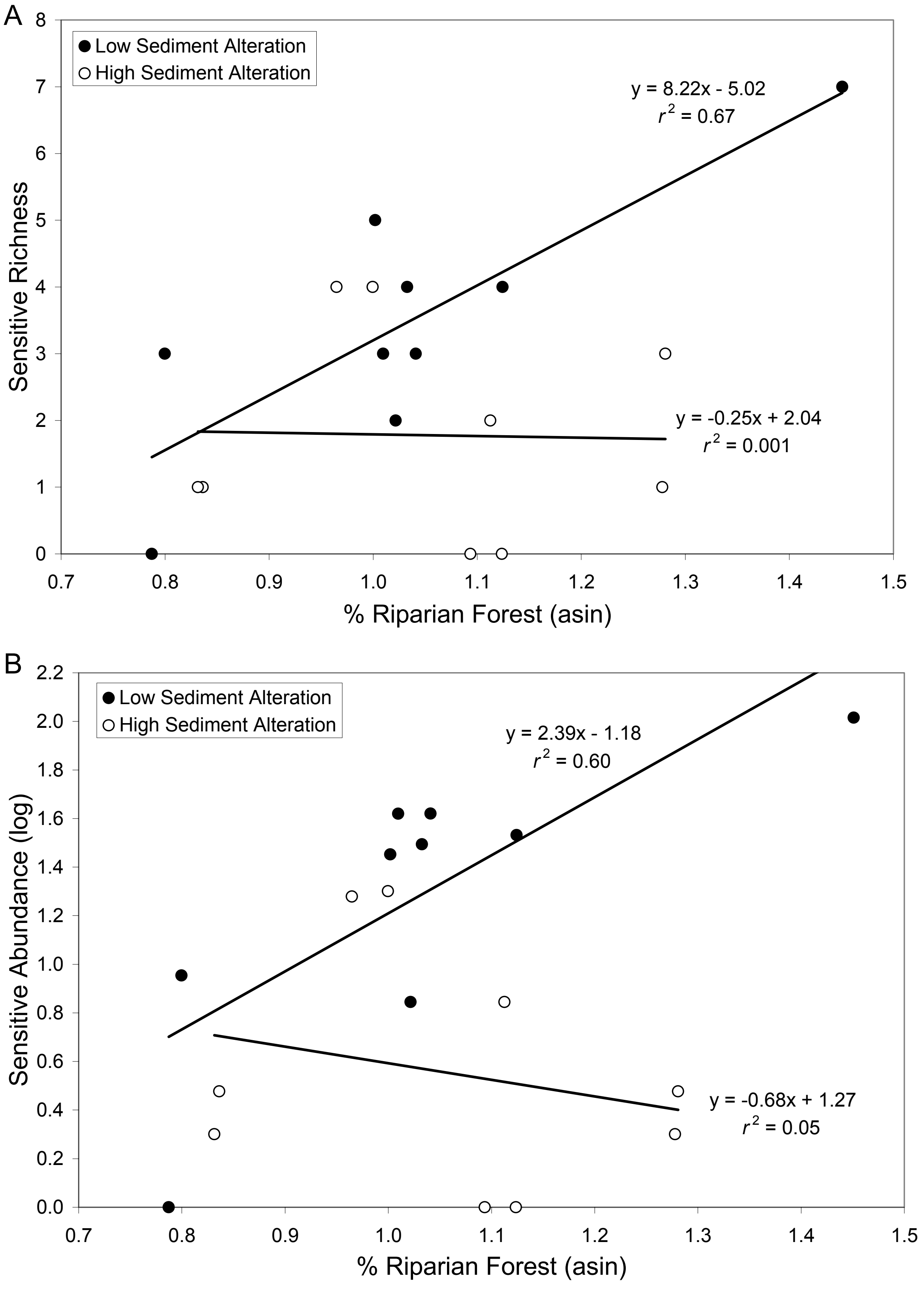
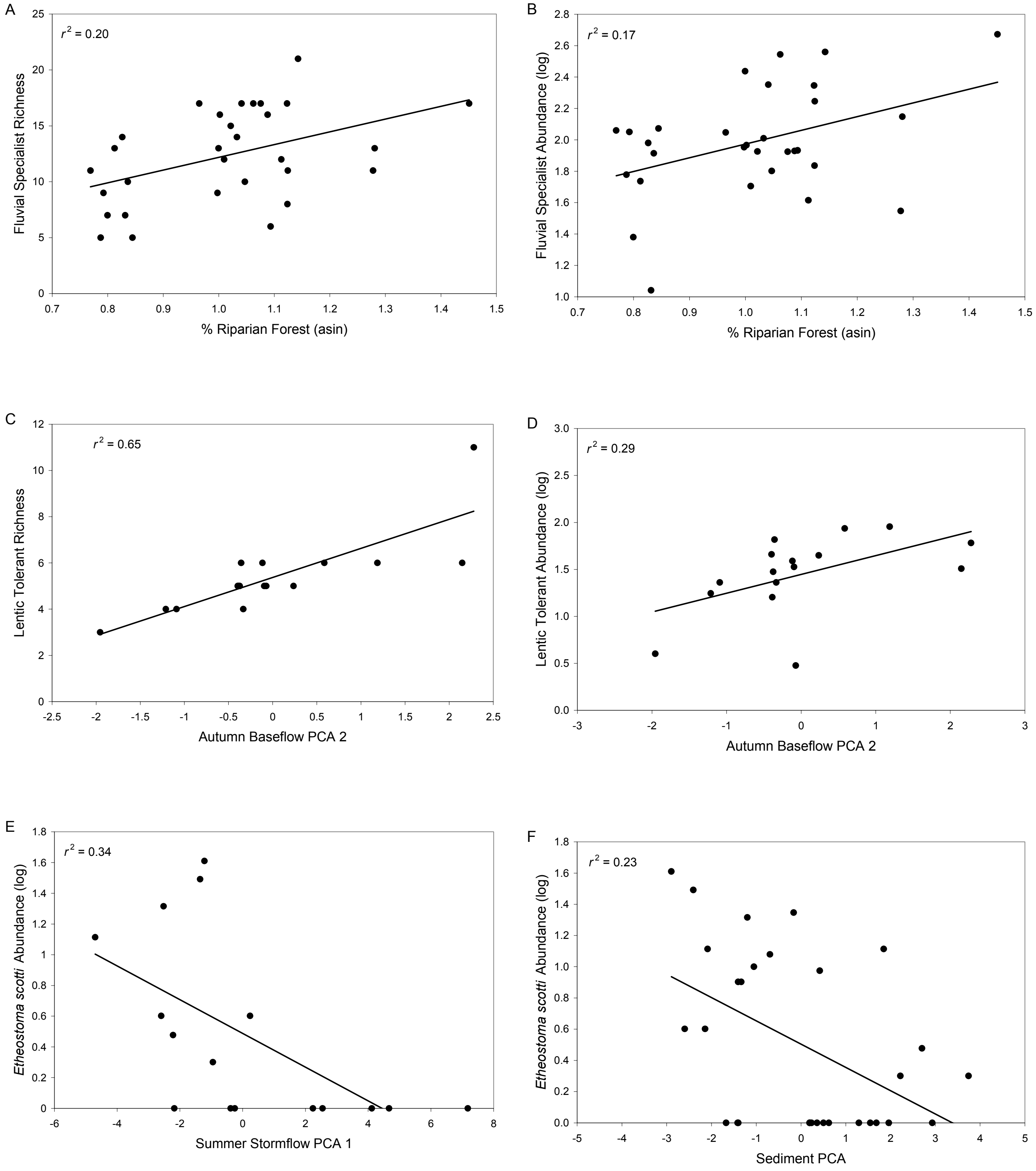

A
● Low Sediment Alteration
○ High Sediment Alteration
y = 8.22x - 5.02
r^2 = 0.67
y = -0.25x + 2.04
r^2 = 0.001
Sensitive Richness
0
1
2
3
4
5
6
7
8
0.7
0.8
0.9
1.0
1.1
1.2
1.3
1.4
1.5
% Riparian Forest (asin)
B
● Low Sediment Alteration
○ High Sediment Alteration
y = 2.39x - 1.18
r^2 = 0.60
y = -0.68x + 1.27
r^2 = 0.05
Sensitive Abundance (log)
0.0
0.2
0.4
0.6
0.8
1.0
1.2
1.4
1.6
1.8
2.0
2.2
0.7
0.8
0.9
1.0
1.1
1.2
1.3
1.4
1.5
% Riparian Forest (asin)

Fig. 5.2


A
r^2 = 0.20
Fluvial Specialist Richness
0
5
10
15
20
25
0.7
0.8
0.9
1.0
1.1
1.2
1.3
1.4
1.5
% Riparian Forest (asin)
B
r^2 = 0.17
Fluvial Specialist Abundance (log)
1.0
1.2
1.4
1.6
1.8
2.0
2.2
2.4
2.6
2.8
0.7
0.8
0.9
1.0
1.1
1.2
1.3
1.4
1.5
% Riparian Forest (asin)
C
r^2 = 0.65
Lentic Tolerant Richness
0
2
4
6
8
10
12
-2.5
-2
-1.5
-1
-0.5
0
0.5
1
1.5
2
2.5
Autumn Baseflow PCA 2
D
r^2 = 0.29
Lentic Tolerant Abundance (log)
0.0
0.5
1.0
1.5
2.0
2.5
3.0
-3
-2
-1
0
1
2
3
Autumn Baseflow PCA 2
E
r^2 = 0.34
Etheostoma scotti Abundance (log)
0
0.2
0.4
0.6
0.8
1
1.2
1.4
1.6
1.8
-6
-4
-2
0
2
4
6
8
Summer Stormflow PCA 1
F
r^2 = 0.23
Etheostoma scotti Abundance (log)
0.0
0.2
0.4
0.6
0.8
1.0
1.2
1.4
1.6
1.8
-5
-4
-3
-2
-1
0
1
2
3
4
5
Sediment PCA

Appendix 5.1. Fishes collected and frequency of occurrence within the 30 study streams. All species were classified as either fluvial specialist (FLU) or lentic tolerant (LEN) based on Etnier and Starnes (1993) and Metee et al. (1993). Sensitive fishes (SEN) were classified based on negative relationships with % urban land cover from previous fish collection data (Seth J. Wenger, University of Georgia, *unpublished data*).

Family Name	**Common Name**	**Composition**	**Freq. of**
Scientific Name		**Categories**	**Occurence**
Petromyzontidae			
Ichthyomyzon sp. cf. gagei	southern brook lamprey	FLU	12
Cyprinidae			
Campostoma oligolepis	largescale stoneroller	FLU	29
Cyprinella callistia	Alabama shiner	FLU, SEN	12
Cyprinella trichoristia	tricolor shiner	FLU, SEN	3
Cyprinella venusta	blacktail shiner	FLU	5
Hybopsis lineapunctata	lined chub	FLU	1
Hybopsis sp. cf. winchelli	clear chub	FLU	2
Luxilus zonistius	banfin shiner	FLU	2
Nocomis leptocephalus	bluehead chub	FLU	5
Notemigonus crysoleucas	golden shiner	LEN	1
Notropis chrosomus	rainbow shiner	FLU, SEN	2
Notropis longirostris	longnose shiner	FLU	3
Notropis lutipinnis	yellowfin shiner	FLU	3
Notropis stilbius	silverstripe shiner	FLU	4
Notropis xaenocephalus	Coosa shiner	FLU	15
Phenacobius catostomus	riffle minnow	FLU, SEN	1
Semotilus atromaculatus	creek chub	FLU	23
Salmonidae			
Onchorhynchus mykiss	rainbow trout	n/a	1
Fundulidae			
Fundulus stellifer	southern studfish	FLU	24
Poecilidae			
Gambusia affinis	eastern mosquitofish	LEN	6
Gambusia holbrooki	western mosquitofish	LEN	2
Gambusia holbrooki x affinis	hydrid mosquitofish	LEN	4
Cottidae			
Cottus carolinae zopherus	Coosa banded sculpin	FLU	23
Centrarchidae			
Ambloplites ariommus	shadow bass	FLU	1
Lepomis auritus	redbreast sunfish	LEN	30
Lepomis cyanellus	green sunfish	LEN	22
Lepomis gulosus	warmouth	LEN	6
Lepomis macrochirus	bluegill sunfish	LEN	29
Lepomis microlophus	redear sunfish	LEN	5
Lepomis punctatus	spotted sunfish	LEN	9
Lepomis macrochirus x auritus	hybrid sunfish	LEN	1

Appendix 5.1. Continued.

Catastomidae			
Hypentelium etowanum	Alabama hog sucker	FLU	30
Minytrema melanops	spotted sucker	FLU	3
Moxostoma duquesnei	black redhorse	FLU, SEN[a]	10
Moxostoma erythrurum	golden redhorse	FLU, SEN[a]	8
Moxostoma poecilurum	blacktail redhorse	FLU	3
Ictaluridae			
Ameiurus brunneus	snail bullhead	FLU, SEN	7
Ameiurus natalis	yellow bullhead	LEN	5
Ameiurus nebulosus	brown bullhead	LEN	2
Ictalurus punctatus	channel catfish	LEN	3
Noturus leptacanthus	speckled madtom	FLU, SEN	5
Micropterus coosae	Coosa bass	FLU	23
Micropterus punctulatus	spotted bass	FLU	4
Micropterus salmoides	largemouth bass	LEN	22
Pomoxis annularis	white crappie	LEN	1
Pomoxis nigromaculatus	black crappie	LEN	2
Percidae			
Etheostoma scotti	Cherokee darter	FLU, SEN	18
Etheostoma stigmaeum	speckled darter	FLU, SEN	10
Perca flavescens	yellow perch	LEN	2
Percina kathae	Mobile logperch	FLU	14
Percina nigrofasciata	blackbanded darter	FLU	27
Percina palmaris	bronze darter	FLU, SEN	1

[a] *Moxostoma duquesnei* and *M. erythrurum* were combined and considered one sensitive species

CHAPTER 6

CONCLUSIONS

Riparian buffers have been used for managing non-point source disturbances in the United States since late 1960s (Calhoun 1988), with the primary goal of isolating upland disturbances from aquatic resources (Lee et al. 2004). In Georgia, the Erosion and Sedimentation Control Act of 1975 (O.C.G.A. 12-7) requires a 25-foot minimum undisturbed riparian buffer on all streams. However, prior to this study little was known about the effectiveness of forested riparian areas for maintaining stream quality and function in urbanizing areas. Although riparian forests have been used as tools to mitigate the negative impacts of upland disturbance, there is potential for these areas to be less effective in urban and suburban settings. Thus, this study was designed to test the importance of riparian areas at a (a) local, reach scale; (b) catchment scale, and (c) given varying levels of hydrologic and sediment disturbance in streams.

At the local scale (Chapter 2), forested canopies provided minimal benefits to streams in terms of habitat quality or biotic integrity (as measured by aquatic macroinvertebrates, salamanders, and fishes) compared to adjacent reaches with open canopies. I observed increases in algae and abundances of herbivorous fishes (*Campostoma oligolepis*) in open reaches, suggesting that openings in riparian canopies may provide local patches of increased primary productivity and shifts in the trophic basis of production. Overall, these streams had relatively poor biotic integrity, suggesting that the multiple impacts of urbanization within these catchments may overwhelm the benefits of local riparian forests for protection of stream ecosystems. Our results were comparable to a similarly designed study by Hession et al. (2002), who reported increases in algae and higher-level trophic responses with riparian deforestation, but minimal effects of local riparian cover on habitat quality and biota integrity. The combined results of this study and the Hession et al. (2002) study provide strong evidence that although riparian forests may be important regulators of stream width, temperature, and the food/energy base at the reach-scale, they do not dictate stream habitat quality or biotic integrity.

Since local patches of riparian forests did not influence fish assemblage integrity in urbanized catchments, I asked whether the % forest in the riparian area along the entire upstream network was related to fishes (Chapter 3). Catchment and riparian land cover for the entire network upstream of the sampling location were highly correlated, so I was unable to distinguish between those variables. Nonetheless, I found that endemic and sensitive fish species were best predicted by increased % forest and decreased % urban land cover within the catchment, while cosmopolitan and tolerant fish species were positively related to % agriculture and negatively related to % forest at the local (200 m and 1 km reach length) riparian scale. Catchment-wide development seemed to provide an initial filter of fish species, with losses of sensitive species occurring at levels of riparian deforestation exceeding *ca*. 30%. Local scale % riparian forest cover secondarily affected tolerant species, which had consistently high abundances in streams with < *ca.* 60 % local riparian forest cover. Although riparian regulations imply that stream conditions in the catchment can be mitigated by protecting land adjacent to streams (Allan et al. 1997, Harding et al. 1998), results from this study support other studies suggesting that catchment land cover is an important driver of biotic assemblages (Roth et al. 1996, Allan and Johnson 1997, Wang et al. 2001).

Researchers have predicted that effectiveness of streamside forests will depend on landscape context and upland stressors (Roth et al. 1996, Naiman and Decamps 1997, Lowrance 1998), yet there have been few empirical tests of this idea, particularly in urban settings. In the Etowah River basin, sediment and hydrologic disturbance were identified as two important stressors to fish assemblages (Freeman et al. 2002). Previous research by Walters et al. (2003) found that decreases in bed texture in low-slope streams led to increased fish impairment in small and medium-sized streams in the Etowah. Hydrologic alteration was also expected to be an important stressor of fish assemblages, so I assessed effects of increased stormflows and decreased baseflows on fishes along a gradient of urban disturbance (Chapter 4). Percent impervious surface cover was associated with increased frequency, magnitude, and duration of storm events and prolonged duration of low flow conditions, and these hydrologic variables explained 20-66 % of the variation in fish assemblage richness and abundance. This was some of the first empirical evidence showing that altered hydrology due to increased amounts of impervious surface cover can affect small stream fish assemblages.

Since sediment and hydrologic alteration were identified as critical stressors for fishes in the Etowah River basin, I asked whether the capacity of riparian forests to maintain healthy fish assemblages was dependent on the level of hydrologic and/or sediment alteration in the stream (Chapter 5). I assessed the relative predictive ability of models including % riparian forest cover and principal component axes describing stormflow, baseflow, and sediment regime. Prolonged duration of low flow conditions explained 65 % and 29 % of the variance in richness and abundance of lentic tolerant species, respectively, and this level of hydrologic alteration overwhelmed any influence of riparian forests on these fishes. Fluvial specialist and sensitive fish species were best predicted by models with both % forest cover in riparian area and sediment disturbance. Only sites with low levels of sediment disturbance (i.e. coarse bed sediment and low bed mobility) exhibited increased richness and abundances of sensitive species in relation to higher % riparian forest cover. Together, these results suggest that, at a minimum, watershed management strategies must simultaneously address hydrologic, sediment, and riparian disturbance in order to protect fish assemblage integrity.

Despite riparian buffer regulations in Georgia, riparian areas exhibited an average 10.3% decrease in forest cover and 8.5% increase in urban land cover between 1973 and 1997 (Roy et al. 2003), suggesting that these regulations and/or current enforcement of these regulations does not seem effective at protecting stream ecosystems from continued loss of forest cover and subsequent declines in fish assemblage integrity. In the last decade, many counties and jurisdictions within the Etowah River watershed have adopted riparian buffer ordinances that are wider and more stringent than the state ordinance of 25 ft (England and Roy 2001). Riparian areas clearly provide essential social, economic, and ecological benefits through increasing greenspace, terrestrial habitat, and essential functions for aquatic environments. However, scientists may have oversold the importance of local governments adopting of good riparian buffer ordinances relative to adoption and enforcement of other important management tools for protecting stream ecosystems (Seth J. Wenger, University of Georgia, *pers. comm.*). This study highlights the importance of simultaneous management of multiple stressors of landscape alteration in order to maintain healthy fish assemblages.

The results from this study will be used to inform county ordinances and policies in concordance with the Etowah Regional Aquatic Habitat Conservation Plan (HCP). I recommend minimizing impervious surface cover in new developments and maintaining essential amounts of forest cover within catchments and riparian areas of streams with sensitive fish species. Although I identified hydrologic alteration as an important stressor to fishes in the Etowah River basin, further research is necessary to better understand specific components of the hydrologic regime that must be maintained to allow for protection of fish assemblages. Ecological design to mimic vital ecosystem services in urban areas should be an essential component of conservation and restoration (Palmer et al. 2004). For example, if developments are designed to minimize runoff and emphasize infiltration, mimicking natural processes of stormwater retention, then it may be possible to minimize the effects of development on stream ecosystems (Lloyd et al. 2002, Walsh et al. 2004). Results of this study also demonstrate that sediment alteration is also a primary stressor to fish assemblages, and effective erosion and sedimentation ordinances are necessary components to effectively maintain healthy fish assemblages. Because the goal of the HCP is to ensure protection of threatened and endangered species, we also need to monitor population sizes and use adaptive management to ensure long-term protection of critical species (Wilhere 2002). Ultimately, a holistic approach to watershed management, including management of multiple stressors, will be necessary to allow for development to occur while maintaining the services provided by stream ecosystems.

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APPENDIX A

PATTERNS OF LAND USE CHANGE IN UPLAND AND RIPARIAN AREAS IN THE ETOWAH RIVER BASIN[1]

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[1] Roy, A. H., M. C. Freeman, J. L. Meyer, and D. S. Leigh. 2003a. Patterns of land use change in upland and riparian areas in the Etowah River basin. Pages 331-334 *in* K. J. Hatcher (editor). Proceedings of the 2003 Georgia Water Resources Conference, Institute of Ecology, University of Georgia. Reprinted here with permission of publisher.

PATTERNS OF LAND USE CHANGE IN UPLAND AND RIPARIAN AREAS IN THE ETOWAH RIVER BASIN

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REFERENCE: *Proceedings of the 2003 Georgia Water Resources Conference*, held April 23-24, 2003, at the University of Georgia, Kathryn J. Hatcher, Editor, Institute of Ecology, The University of Georgia, Athens, Georgia.

Abstract. Streams are influenced by the upstream landscape, but may be differentially affected by conversion of forests in the entire catchment vs riparian areas adjacent to streams. We used geographic information system (GIS) analyses of the stream network and land cover in the Piedmont of the Etowah River basin to assess development patterns in upland catchment and riparian areas of streams. Landsat images (1973, 1987, 1997) were used to determine land cover and land cover change in a 100 m buffer on each side of the stream and the catchment as a whole. Agricultural and urban uses covered a larger percentage of the catchment area compared to the riparian area. Streams exhibited an average 13% decrease in forest cover and 11% increase in urban land cover in the catchments over the 24 year period, with riparian areas changing at a slower rate. Small (~15 km^2) and large (~100 km^2) catchments had similar proportions of buffer vs catchment forest land cover. Although rates of development were less in riparian areas, the continued trends of increased urban and decreased forest cover suggest that current policies may not be adequate at protecting stream ecosystems.

INTRODUCTION

Forested land is being converted to agricultural and urban land uses nationwide (USDA 2000). These land conversions may occur discriminately, based on elevation, geology, or location relative to landscape resources. For example, the function of water as a resource for irrigation, livestock, mining, transportation and land design aesthetics may encourage development adjacent to streams, while increased land losses due to erosion and other negative effects of riparian deforestation may deter development adjacent to streams (e.g., through buffer ordinances or best management practices).

Land cover changes in the catchment can impair water quality and biotic assemblages (Allan and Johnson 1997). Similarly, decreased forest cover in the riparian area adjacent to the stream increases nutrients, temperature, and primary productivity and decreases bed texture and allochthonous inputs (Sweeney 1992). These changes, in turn, can impact biotic assemblages (Jones et al. 1999). Although stream integrity is a function of the entire upstream catchment, the critical location of riparian areas within the landscape may constitute a disproportional influence on aquatic ecosystems (Weller et al. 1998).

We examined patterns of landscape development in the Piedmont physiographic region of the Etowah River basin by directly comparing land cover changes between 1973 and 1997 in riparian areas to those in corresponding catchments. We hypothesized that 1) proportional forested land cover is higher and agricultural and urban land cover are lower in riparian vs catchment areas, 2) trends in land cover change demonstrate less deforestation of riparian vs catchment areas through time, and 3) large streams have higher % forested land cover in riparian areas vs catchment relative to small streams.

METHODS

Sites used in this study were located within the Piedmont physiographic region of the Etowah River basin. For the land cover and land cover change analyses, we conducted a census of all non-nested small streams (10-20 km^2) within this region (n = 83 streams). For the catchment size comparison, 10 streams from 15, 50, and 100 km^2 ± 25% catchments were randomly selected (Leigh et al. 2002).

1973 *Landsat* MSS images (60 m pixels) and 1987 and 1997 *Landsat* TM images (30 m pixels) were used to obtain land coverages (Lo and Yang 2000). Classifications were grouped according to the six class system used in 1973, which included high density urban, low density urban, cultivated/exposed land, cropland/grassland & golf courses, forest land, and open water.

We created a drainage network from Digital Elevation Models (DEMs) which was similar to a 1:24,000 scale stream network. This drainage network was used to create 100 m buffers for the entire extent of the drainage network. For the 30 streams used in size analyses, we also calculated a 100 m buffer for the 1 km reach at the downstream-most portion of the drainage to test whether patterns of riparian land use were locally patchy. Arcview 3.2© was used to tabulate catchment and 100 m buffer areas based on land cover (Environmental Systems Research Institute, Inc., Redlands, CA).

We divided the proportion land cover in the riparian area (100 m buffer) by the proportion land cover in the entire catchment to analyze relative changes in buffer vs catchment. Land cover change variables were calculated for 1973-1987 and 1987-1997. Paired t-tests were used to compare mean differences in change in catchment vs riparian land cover for the 83 streams. A one-way ANOVA was used to compare mean land cover variables among the three stream size classes.

RESULTS

Catchments exhibited a range in 1997 % forest (25-96%), urban (1-67%), and agriculture (3-42%) land cover. Mean forested land cover was higher in the riparian areas than in the catchment, with only six of the 83 sites having lower relative forest cover in the riparian area. Percent urban and agricultural land cover were both lower in the riparian area relative to the catchment. Across all sites, the proportion of open water in the riparian area was double the amount in the catchment (Table 1).

Changes in land cover demonstrated an average 13% decrease in forest (-37% to +4%) and 11% increase in urban land cover (-9% to +47%) in the catchments over 24 years. Agricultural land cover in the catchments decreased between 1973 and 1987 and increased between 1987 and 1997. Similar trends existed for land cover in the riparian area; however, the changes in forest and urban were smaller in magnitude. Open water increased more in the riparian area (0.8%) than in the catchment (0.4%) through time (Figure 1; Table 2).

Table 1. Ratio of 1997 riparian to catchment percent land cover in 83 small catchments. Numbers >1 indicate higher percent land cover in riparian vs catchment.

	Mean	St. Error
Total Forest Cover	1.11	0.011
Total Urban Cover	0.73	0.019
High Density Urban	0.70	0.039
Low Density Urban	0.74	0.019
Total Agriculture Cover	0.85	0.018
Cultivated/Exposed Land	0.61	0.041
Crop/Grassland & Golf Courses	0.87	0.018
Open Water	1.99	0.053

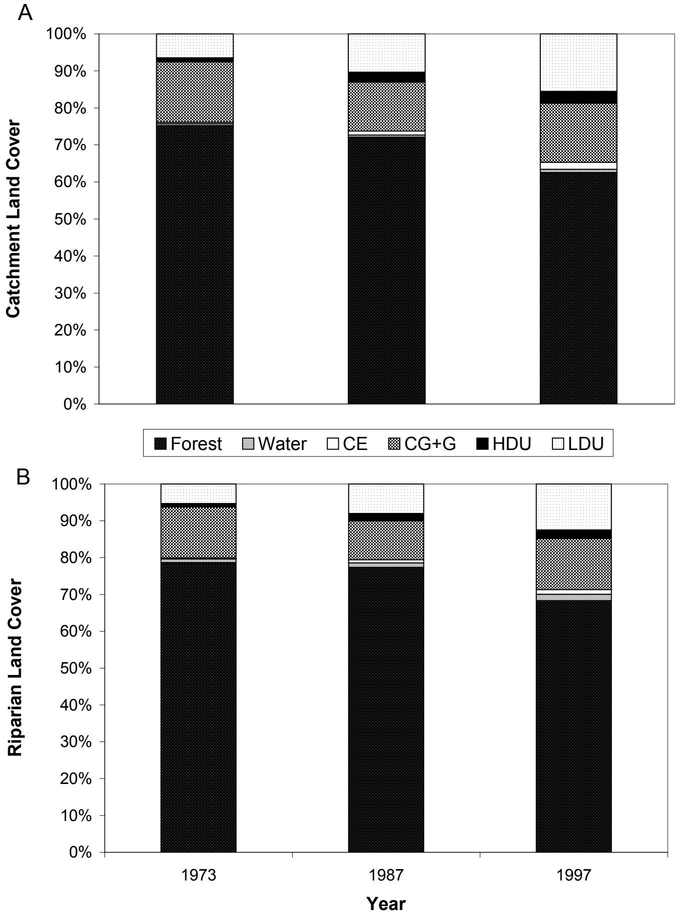

Figure 1. Percent catchment (A) and riparian (B) land cover in 1973, 1987 and 1997. LDU = low density urban, HDU = high density urban, CG+G = crop/grassland + golf courses, CE = cultivated/exposed land.

There were no differences in mean % urban, forest, or agricultural land in the riparian vs catchment across stream size classes (Table 3). The 1 km reach had higher variability in riparian land cover relative to the catchment compared to the riparian area calculated for the entire upstream network. For example, percent forest was higher in the riparian area vs catchment in 27 of the 30 sites based on buffering the entire stream, but half of the sites had lower percent forest in the riparian area vs catchment when considering only the 1 km reach (Figure 2).

Table 2. Mean (SE) of riparian, catchment, and riparian/catchment ratio (R/C) for change in land cover from 1973-87 and 1987-97 for the 83 streams. Sign (+/-) indicates direction of change. Significant differences between catchment and riparian land cover are indicated (paired *t*-test); *** = $p<0.001$, ** = $p<0.01$.

	1973 to 1987 % Land Cover Change				1987 to 1997 % Land Cover Change			
	Forest***	Urban***	Agriculture	Water***	Forest**	Urban***	Agriculture	Water***
Catchment	-3.10 (1.00)	+5.42 (0.96)	-2.35 (1.02)	+0.03 (0.03)	-9.57 (0.83)	+5.71 (0.70)	+3.52 (0.71)	+0.34 (0.05)
Riparian	-1.31 (1.12)	+3.71 (0.84)	-2.63 (0.95)	+0.24 (0.06)	-9.02 (0.81)	+4.79 (0.65)	+3.70 (0.61)	+0.53 (0.06)
R/C	+0.03 (0.00)	-0.06 (0.04)	-0.05 (0.04)	+0.50 (0.09)	+0.03 (0.00)	+0.01 (0.03)	+0.06 (0.02)	-0.07 (0.06)

DISCUSSION

In the Piedmont portion of the Etowah basin, loss of forest cover is occurring faster in the catchment than the riparian area. This may be a result of statewide protection of riparian buffers, although only 25 ft (~8 m) is protected under Georgia's Erosion and Sedimentation Control Act (OCGA 12-7). Perceived problems associated with developing in riparian areas along with increased knowledge of the benefits of having a forested riparian area may also be contributing to higher percent forest cover in riparian areas.

Although the proportion of development is less in riparian areas, there has been an increase of 8.5% urban and a decrease of 10.3% forest in riparian areas in the last 24 years. If these trends continue, 94% of small streams will have >10% urban in their riparian areas (vs 43% in 1997) and 87% of small streams will have >15% urban in upland catchments (vs 39% in 1997) by 2021. These high levels of urbanization typically correspond to impaired stream ecosystems (Paul and Meyer 2001). Thus, current policy may not be adequate at protecting streams from land cover change.

This pattern of higher forest in riparian areas relative to catchment is not consistent for all areas of the US. A study in the Blue Ridge physiographic region of Georgia indicated a trend of *higher* deforestation in the area relative to the catchment (riparian/catchment ratio 0.91 vs 1.11 in Piedmont). Of the 30 sites sampled in that study, 24 sites had lower forest land cover in riparian vs catchment, presumably due to the higher ease of developing in valleys adjacent to streams (Kundell et al. 2002). These contradictory patterns in location of land development offer an excellent opportunity to understand relations between development patterns and stream ecosystem quality.

Table 3. Ratio of 1997 percent riparian land cover (within 100 m buffer on 1 km reach and entire extent stream network) to percent catchment land cover in small (15 km^2), medium (50 km^2) and large (100 km^2) catchments for 30 sites. F and *p* values are from a one-way ANOVA based on log(x+1) transformed data.

	15 km^2	50 km^2	100 km^2	F	p
Urban					
1 km	0.74 (0.19)	0.83 (0.24)	0.51 (0.17)	0.91	0.41
entire	0.73 (0.05)	0.77 (0.02)	0.73 (0.03)	0.90	0.42
Forest					
1 km	0.81 (0.12)	1.02 (0.11)	1.00 (0.09)	1.31	0.29
entire	1.14 (0.02)	1.11 (0.02)	1.11 (0.02)	0.68	0.52
Agriculture					
1 km	1.44 (0.26)	1.72 (0.61)	1.36 (0.40)	0.04	0.96
entire	0.76 (0.05)	0.85 (0.04)	0.89 (0.04)	1.84	0.18

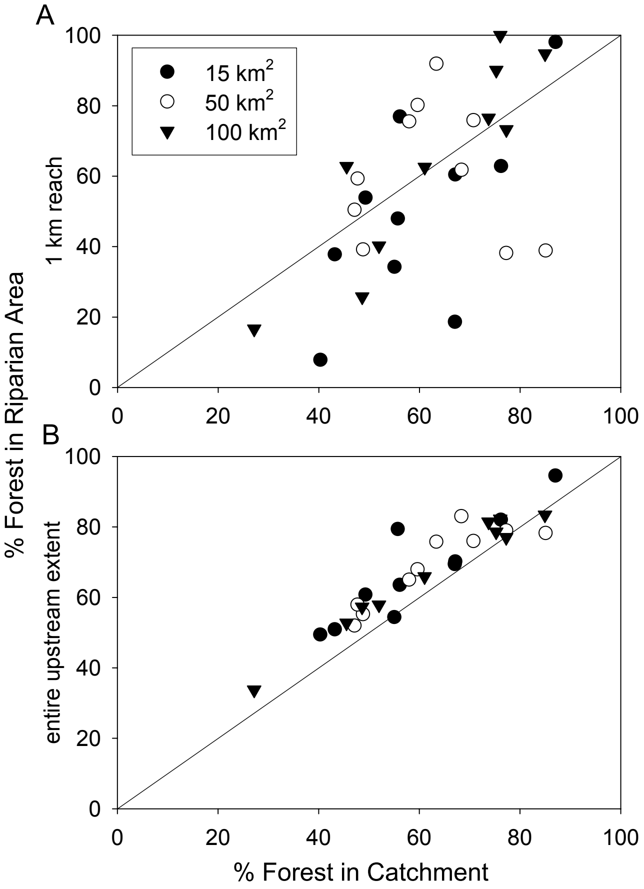

Figure 2. 1997 Percent forested riparian land cover (100 m buffer) for 1 km reach (A) and entire upstream extent (B) vs percent forest in catchment for ten small (15 km^2), medium (50 km^2) and large (100 km^2) catchments. Line indicates 1:1.

We hypothesized that larger streams would have a more intact riparian area relative to the catchment because of higher protection afforded to larger streams and that larger streams are more of a public resource (i.e., small streams often transect properties while large streams border property lines). However, we found no evidence of differences across stream catchment size. Such a pattern may exist with even smaller streams (e.g., <10 km^2) that have fewer landowners. Further, the scale of the stream network (1:24,000) used in this study may be too large to detect land uses occurring on very small streams (Meyer and Wallace 2001).

All size streams had highly variable land cover within the 1 km buffer, indicating that that forested riparian land cover is extremely spatially patchy. Studies suggest that the degree of patchiness may be related to stream quality. For example, Jones et al. (1999) showed a significant relationship between the length of deforested riparian patches and fish assemblage changes. Because gaps in riparian areas may dictate water conduits in the landscape, the number and extent of deforested reaches, rather the proportion of deforestation within riparian areas may be more related to stream quality (Weller et al. 1998).

Many regulations give higher protection to larger streams relative to smaller streams. For example, the Metropolitan River Protection Act (OCGA 12-5-440 to 12-5-457) mandates wider buffer protection along the main stem of the Chattahoochee River than on smaller tributaries. This discrimination based on catchment size seems unwarranted, as a larger percentage of the catchment is in closer contact with small streams. Since stream quality is a function of all upstream uses, equal protection of large and small streams is recommended (Meyer and Wallace 2001).

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APPENDIX B

SUPPLEMENTAL MATERIAL TO CHAPTER 2

Appendix B.1. Mean canopy cover and tree density for open and forest reaches of 5 study streams.

	Clark		Copper Sandy		Lawrence		Picketts Mill		Westbrook		OPEN		FOREST			
	open	forest	open	forest	open	forest	open	forest	open	forest	mean	st. dev.	mean	st. dev.	t	p
Cover																
% Canopy cover (July 2002)	10.63	91.25	44.17	82.11	62.81	85.64	22.97	83.75	20.83	58.85	32.28	20.98	77.59	12.49	4.73	0.005
% Canopy cover (Sept. 2003)	20.05	94.27	63.39	95.73	60.05	90.47	30.31	90.83	25.52	80.89	39.86	20.31	90.44	5.79	6.00	0.002
Trees																
# Trees/ m^2	0.01	0.17	0.01	0.22	0.05	0.11	0.09	0.22	n/a	n/a	0.04	0.04	0.18	0.05	4.53	0.010
Tree Diameter (cm)	39.34	12.47	21.43	7.91	17.54	14.01	10.40	12.11	n/a	n/a	22.18	12.32	11.62	2.61	1.68	0.096
# Trees/ m^2 >10m	0.01	0.18	0.01	0.20	0.05	0.13	0.10	0.24	n/a	n/a	0.04	0.04	0.19	0.04	6.15	0.004
Tree Diameter (cm)>10m	44.07	11.97	27.22	9.18	19.84	13.61	10.05	12.03	n/a	n/a	25.30	14.36	11.70	1.84	1.84	0.082
# Trees/ m^2 >10m	0.00	0.13	0.02	0.26	0.04	0.06	0.07	0.19	n/a	n/a	0.03	0.03	0.16	0.09	2.68	0.038
Tree Diameter (cm) <10m	39.79	13.75	22.99	6.58	15.89	18.09	10.34	11.23	n/a	n/a	22.25	12.79	12.41	4.81	1.43	0.124
Basal area (ft^2) per acre (total)	199	2257	138	1684	723	1146	818	3300	n/a	n/a	469	350	2097	922	3.66	0.018
Basal area (ft^2) per acre (right bank)	168	2293	264	1206	58	854	1635	2547	n/a	n/a	531	741	1725	822	3.83	0.016
Basal area (ft^2) per acre (left bank)	229	2222	13	2161	1388	1438	0	4053	n/a	n/a	408	662	2468	1115	2.52	0.043

Appendix B.2. Mean annual baseflow water quality (average of 5 sample dates throughout year) for open and forest reaches of 5 study streams.

	Clark		Copper Sandy		Lawrence		Picketts Mill		Westbrook		OPEN		FOREST			
	open	forest	open	forest	open	forest	open	forest	open	forest	mean	st. dev.	mean	st. dev.	t	p
Discharge (m^3 s^{-1})	0.032	0.040	0.036	0.027	0.028	0.029	0.049	0.043	0.038	0.035	0.037	0.008	0.035	0.007	0.69	0.264
Dissolved oxygen (mg L^{-1})	10.2	9.3	7.8	7.3	9.9	8.9	8.9	8.7	10.0	10.1	9.36	1.00	8.88	1.00	2.51	0.033
Specific conductance (µg cm^{-1})	94.8	97.6	67.4	51.6	115.2	84.6	110.4	110.6	108.2	105.8	99.20	19.32	90.04	23.64	1.47	0.108
pH	8.7	8.4	7.8	8.1	8.0	7.2	8.3	8.2	8.0	8.0	8.15	0.35	7.98	0.46	0.87	0.216
Turbidity (NTU)	3.8	4.6	11.1	12.0	15.4	13.7	6.6	7.4	27.2	16.1	12.82	9.17	10.77	4.68	0.82	0.229
SRP (µg L^{-1})	0.000	0.002	0.016	0.003	0.000	0.006	0.002	0.002	0.005	0.002	0.005	0.007	0.003	0.002	0.52	0.315
NH$_4$ (µg L^{-1})	0.000	0.000	0.018	0.008	0.005	0.008	0.005	0.001	0.000	0.000	0.005	0.007	0.003	0.004	0.95	0.197
NO$_2$/NO$_3$ (µg L^{-1})	0.102	0.112	0.151	0.166	0.084	0.093	0.150	0.166	0.065	0.064	0.110	0.039	0.120	0.045	3.24	0.016
DIN:SRP	729.6	51.2	10.6	52.4	--	18.0	71.0	71.7	12.6	34.2	205.9	350.2	45.5	20.3	0.88	0.223
Total Suspended Solids (mg L^{-1})	13.60	19.52	23.20	19.52	30.24	23.52	24.00	24.00	30.40	26.72	24.29	6.86	22.66	3.11	0.75	0.247

Appendix B.3. Mean temperature variables for open and forest reaches of 5 study streams.

	Clark		Copper Sandy		Lawrence		Picketts Mill		Westbrook		OPEN		FOREST			
	open	forest	open	forest	open	forest	open	forest	open	forest	mean	st. dev.	mean	st. dev.	t	p
Annual minimum (°C)[a]	-3.85	-0.61	1.60	-3.85	1.60	-1.06	0.29	-0.16	-0.16	-0.16	-0.10	2.24	-1.17	1.55	0.74	0.251
Annual maximim (°C)[a]	37.88	27.52	29.50	30.71	47.96	37.00	32.34	31.93	27.52	27.52	35.04	8.21	30.94	3.91	1.53	0.101
Annual mean (°C)[a]	16.91	15.76	17.23	17.03	16.00	15.79	16.08	15.84	11.77	11.77	15.60	2.20	15.24	2.01	1.78	0.075
Annual standard deviation (°C)[a]	7.88	6.58	6.97	7.29	6.64	6.66	6.91	6.46	5.96	5.99	6.87	0.69	6.60	0.47	0.97	0.194
Annual daily diel (°C)[a]	5.6186	2.822	2.873	3.211	4.6222	4.639	4.255	3.267	4.312	3.81	4.3	1.0	3.5	0.7	1.43	0.113
Annual degrees days[a, b]	6190	5769	6214	6225	5780	5857	5796	5885	n/a	n/a	5995	239	5934	200	0.51	0.324
January minimum (°C)	-0.61	-0.61	1.60	1.17	1.60	-1.06	0.29	-0.16	-0.16	-0.16	0.54	1.02	-0.16	0.83	1.42	0.114
January maximim (°C)	15.23	15.23	13.70	13.70	21.71	14.47	17.14	17.90	14.09	14.47	16.37	3.27	15.15	1.63	0.81	0.232
January mean (°C)	6.44	6.24	6.40	6.36	6.73	6.42	6.69	6.72	5.86	5.77	6.42	0.35	6.30	0.35	1.98	0.060
January standard deviation (°C)	3.61	3.58	2.76	2.80	3.24	3.55	3.48	3.55	3.35	3.41	3.29	0.33	3.38	0.33	1.59	0.094
Winter daily minimum (°C)	5.2346	5.784	7.069	5.401	6.3901	6.149	5.95	5.925	5.476	5.414	6.0	0.7	5.7	0.3	0.78	0.239
Winter daily maximum (°C)	12.25	9.292	9.558	9.886	9.7175	9.365	9.696	9.639	9.031	8.666	10.1	1.3	9.4	0.5	1.17	0.154
Winter daily diel (°C)	7.0154	3.509	2.489	4.485	3.3274	3.216	3.746	3.714	3.555	3.252	4.0	1.7	3.6	0.5	0.44	0.340
July minimum (°C)[b]	22.09	20.95	22.09	22.86	19.42	19.42	20.57	20.57	n/a	n/a	21.04	1.30	20.95	1.43	0.24	0.415
July maximim (°C)[b]	34.01	26.34	29.50	30.71	27.12	32.76	29.90	28.31	n/a	n/a	30.13	2.86	29.53	2.80	0.22	0.421
July mean (°C)[b]	27.67	23.83	26.00	26.08	23.81	23.71	24.48	23.98	n/a	n/a	25.49	1.72	24.40	1.12	1.18	0.162
July standard deviation (°C)[b]	2.86	1.11	1.54	1.38	1.47	1.52	2.11	1.44	n/a	n/a	2.00	0.64	1.36	0.18	1.58	0.107
Summer daily minimum (°C)[b]	22.04	20.27	21.44	20.05	19.99	19.94	20.76	20.27	n/a	n/a	21.06	0.88	20.13	0.17	2.34	0.051
Summer daily maximim (°C)[b]	29.40	24.91	27.56	27.81	24.99	25.01	26.57	25.72	n/a	n/a	27.13	1.85	25.86	1.35	1.15	0.166
Summer daily mean (°C)[b]	26.76	22.90	24.70	24.56	22.71	22.69	24.13	22.90	n/a	n/a	24.58	1.68	23.26	0.87	1.47	0.119
Summer daily st. deviation (°C)[b]	1.86	1.21	1.56	1.84	1.23	1.22	1.38	1.25	n/a	n/a	1.51	0.27	1.38	0.31	0.66	0.278
Summer daily diel (°C)[b]	6.754	1.921	3.947	3.43	2.9229	3.675	5.421	2.319	n/a	n/a	4.8	1.7	2.8	0.8	1.53	0.112
Summer 7-day maximim (°C)[b]	28.67	24.44	26.65	26.93	24.53	24.45	26.06	24.82	n/a	n/a	26.48	1.71	25.16	1.19	1.29	0.144

[a] Clark open 4/1 to 6/26 missing, replaced with forest data; Westbrook data ends 6/11

[b] No data for Westbrook Creek

Appendix B.4. Habitat and geomorphic variables measured at each 200 m stream reach. Habitat diversity measures were from particle size (phi), depth, and velocity measurements taken along five longitudinal transects (10, 30, 50, 70, 90 percentile) for a total 204 random locations. Shannon Index (H') reflects overall habitat diversity, whereas J' is a measure of evenness.

	Clark		Copper Sandy		Lawrence		Picketts Mill		Westbrook		OPEN		FOREST			
	open	forest	open	forest	open	forest	open	forest	open	forest	mean	st. dev.	mean	st. dev.	t	p
Habitat area																
Riffle Area	89.85	522.80	111.83	127.77	39.80	83.78	199.50	176.40	64.55	158.25	101.11	61.27	213.80	176.26	1.37	0.121
% Riffle Area	17.97	58.76	27.36	19.78	6.74	13.24	26.76	27.04	6.96	16.45	17.16	10.12	27.05	18.45	1.12	0.148
Pool/Run Area	410.25	366.95	296.94	518.06	550.58	548.95	546.13	476.05	863.10	803.88	533.40	212.27	542.78	161.39	0.17	0.436
% Pool Area	82.03	41.24	72.64	80.22	93.26	86.76	73.24	72.96	93.04	83.55	82.84	10.12	72.95	18.45	1.12	0.148
Width mean (m)	4.12	6.36	2.84	4.48	3.18	3.60	4.26	4.56	5.91	5.90	4.06	1.20	4.98	1.13	2.12	0.051
Depth maximum (m)	1.16	0.58	0.73	0.55	0.46	1.01	1.37	0.70	0.30	1.13	0.80	0.45	0.79	0.26	0.04	0.485
Depth mean (m)	0.25	0.15	0.17	0.26	0.13	0.20	0.35	0.21	0.21	0.46	0.22	0.08	0.25	0.12	0.46	0.334
Depth st. deviation (m)	0.21	0.10	0.11	0.15	0.07	0.19	0.23	0.14	0.12	0.32	0.15	0.07	0.18	0.09	0.51	0.319
Bed texture																
Bed sediment mean (phi)	-4.86	-3.78	-2.88	-0.93	-1.14	-1.84	-2.17	-2.88	-1.28	-2.30	-2.47	1.51	-2.34	1.07	0.21	0.423
Bed sediment st. dev. (phi)	1.44	1.85	1.94	0.90	1.25	1.74	2.13	2.37	1.70	2.88	1.69	0.36	1.95	0.74	0.71	0.259
% Embeddedness	31.43	30.00	37.78	53.33	64.00	40.00	48.00	56.67	45.00	40.00	45.24	12.31	44.00	10.90	0.18	0.432
St. dev. % embeddedness	10.69	10.95	18.56	17.32	8.94	17.89	22.80	19.66	19.15	24.49	16.03	5.93	18.06	4.87	0.91	0.206
% Fines cover	15.00	35.63	50.91	60.00	89.09	70.42	60.00	56.88	80.00	60.00	59.00	28.93	56.58	12.79	0.31	0.387
St. dev. % fines	18.03	32.12	38.33	34.31	18.00	38.76	36.97	44.32	36.51	33.29	29.57	10.57	36.56	5.01	1.45	0.110
% Fines in riffle	0.40	0.69	n/a	n/a	39.55	14.65	8.24	3.59	n/a	n/a	16.07	20.72	6.31	7.37	1.27	0.167
Habitat Diversity																
H' phi	0.54	0.74	0.62	0.33	0.37	0.51	0.56	0.67	0.50	0.58	0.52	0.09	0.57	0.16	0.59	0.292
H' depth	0.60	0.54	0.53	0.57	0.49	0.59	0.53	0.59	0.57	0.49	0.54	0.04	0.55	0.04	0.35	0.373
H' velocity	0.46	0.48	0.60	0.51	0.51	0.52	0.36	0.47	0.39	0.33	0.47	0.09	0.46	0.08	0.04	0.485
H' phi-depth	1.10	1.23	1.16	0.87	0.83	1.05	1.05	1.22	1.05	0.98	1.04	0.12	1.07	0.16	0.33	0.381
H' phi-vel	0.97	1.12	1.13	0.74	0.83	0.92	0.86	1.08	0.82	0.84	0.92	0.13	0.94	0.16	0.15	0.443
H' phi-depth-vel	1.45	1.57	1.55	1.24	1.22	1.38	1.30	1.57	1.30	1.17	1.36	0.13	1.39	0.18	0.20	0.426
J' phi	0.57	0.77	0.64	0.35	0.39	0.54	0.58	0.70	0.53	0.58	0.54	0.10	0.59	0.16	0.50	0.311
J' depth	0.99	0.90	0.88	0.94	0.81	0.97	0.88	0.98	0.95	0.81	0.90	0.07	0.92	0.07	0.34	0.377
J' velocity	0.76	0.80	0.99	0.85	0.85	0.87	0.60	0.79	0.65	0.55	0.77	0.16	0.77	0.13	0.04	0.485
J' phi-depth	0.70	0.79	0.75	0.56	0.54	0.67	0.68	0.78	0.67	0.61	0.67	0.08	0.68	0.10	0.26	0.403
J' phi-vel	0.62	0.72	0.73	0.47	0.53	0.56	0.56	0.70	0.53	0.52	0.59	0.08	0.59	0.11	0.01	0.495
J' phi-depth-vel	0.67	0.73	0.72	0.57	0.56	0.64	0.60	0.73	0.60	0.53	0.63	0.06	0.64	0.09	0.15	0.443

Appendix B.5. Large woody debris in the 200 m stream reach, mean benthic organic matter (BOM) from 5 replicate riffle samples, and algae and biofilm ash free dry mass (AFDM) on dominant substrate averaged for 10 transects within reaches.

	Clark		Copper Sandy		Lawrence		Picketts Mill		Westbrook		OPEN		FOREST			
	open	forest	open	forest	open	forest	open	forest	open	forest	mean	st. dev.	mean	st. dev.	t	p
Large Woody Debris																
Total # pieces	4.0	39.0	17.0	39.0	13.0	51.0	7.0	25.0	9.0	5.0	10.0	5.1	31.8	17.6	2.92	0.022
Total pieces >20 cm dia.	1.0	16.0	1.0	1.0	1.0	1.0	2.0	6.0	0.0	1.0	1.0	0.7	5.0	6.5	1.41	0.116
% pieces >20 cm dia.	25.0	41.0	5.9	2.6	7.7	2.0	28.6	24.0	0.0	20.0	13.4	12.6	17.9	16.3	0.80	0.233
% Pool-forming	25.0	7.7	11.8	10.3	15.4	13.7	42.9	12.0	11.1	0.0	21.2	13.3	8.7	5.4	2.28	0.042
% Attached	25.0	35.9	70.6	71.8	7.7	51.0	42.9	64.0	44.4	60.0	38.1	23.5	56.5	13.8	2.62	0.029
Mean length (cm)	170.0	140.3	57.1	47.8	165.4	109.9	240.0	190.0	203.3	174.0	167.2	68.5	132.4	56.5	4.20	0.007
Mean diameter (cm)	17.5	19.4	12.1	11.6	20.5	13.4	17.4	17.7	13.3	15.6	16.2	3.4	15.5	3.1	0.37	0.365
Mean volume (m^3)	0.08	0.07	0.01	0.01	0.08	0.02	0.06	0.11	0.03	0.03	0.05	0.03	0.05	0.04	0.38	0.361
Total volume (m^3)	0.31	2.72	0.11	0.21	1.04	0.89	0.44	2.68	0.30	0.14	0.44	0.35	1.33	1.29	1.51	0.103
Benthic Organic Matter																
FBOM AFDM (g m^{-2})	1.01	1.08	1.64	1.43	1.24	2.50	0.64	3.14	0.67	0.57	1.04	0.42	1.74	1.05	0.49	0.325
CBOM AFDM (g m^{-2})	6.08	7.14	8.92	6.62	7.38	5.34	1.39	4.13	10.88	6.93	6.93	3.58	6.03	1.27	0.57	0.298
Total AFDM (g m^{-2})	7.08	8.22	10.50	7.84	8.46	5.90	1.96	4.56	11.55	7.51	7.91	3.75	6.80	1.54	0.54	0.308
Algae																
Total biofilm AFDM (g m^{-2})	16.5	6.8 *	38.7	368.5 *	41.4	47.7	33.8	31.0	34.1	50.4	32.9	9.7	100.9	150.6	1.04	0.179
Sand biofilm AFDM (g m^{-2})	n/a	n/a	133.5	368.5 *	45.5	46.5	51.1	46.5	35.9	60.9	66.5	45.1	130.6	158.8	1.12	0.172
Rock biofilm AFDM (g m^{-2})	16.5	6.8	n/a	n/a	4.1	2.1	16.4	15.4	18.4	8.2	13.9	6.6	8.1	5.5	2.32	0.051
Total chlorophyll *a* (mg m^{-2})	56.5	6.9 *	36.2	33.8	19.8	9.4 *	37.6	10.2 *	13.5	8.1 *	32.7	16.9	13.7	11.3	2.17	0.048
Sand chlorophyll *a* (mg m^{-2})	n/a	n/a	31.7	33.8	21.5	11.1 *	36.4	9.0 *	13.6	7.3 *	25.8	10.2	15.3	12.4	1.69	0.095
Rock chlorophyll *a* (mg m^{-2})	56.5	6.9 *	n/a	n/a	5.2	2.8	38.9	11.6 *	12.7	11.3	28.3	23.7	8.1	4.2	1.76	0.089

Appendix B.6. Total abundance of macroinvertbrates in 5 riffle Surber samples (total area = 0.45 m^2) at each site. Functional feeding groups (FFGs) were assigned based on Merritt and Cummins (1998).

			Clark		Copper Sandy		Lawrence		Picketts Mill		Westbrook	
		FFG	open	forest	open	forest	open	forest	open	forest	open	forest
OTHER												
Annelida: Nematoda			2	7	11	18	7	38	0	4	4	20
Annelida: Oligochaeta			122	422	647	131	53	40	744	760	131	158
Crustacea: Amphipoda			0	0	2	0	0	0	0	0	0	0
Crustacea: Cladocera			0	0	909	613	2	0	0	0	0	0
Crustacea: Copepoda			7	0	49	44	7	7	0	0	0	0
Mollusca: Corbiculidae: *Corbicula* spp.			24	2	100	36	53	11	0	2	27	20
Mollusca: Patellidae (limpet)			0	0	4	4	0	0	0	0	0	0
Arthropoda: Acarina: Hydracarina			9	2	38	20	13	29	11	11	31	4
Arthropoda: Arachnida			0	2	0	4	0	0	2	0	2	0
Arthropoda: Insecta: Hymenoptera: Formicidae			0	0	4	0	0	0	0	2	0	0
Arthropoda: Insecta: Collembola			4	4	2	0	0	0	2	0	0	4
Arthropoda: Insecta: Hemiptera: Aphididae			0	2	40	7	0	0	0	0	0	2
Arthropoda: Insecta: Thysanoptera			0	0	2	0	0	0	0	0	2	0
Arthropoda: Insecta: Hemiptera (terrestrial)			0	0	7	4	0	0	0	0	0	0
ODONATA												
Aeshnidae	*Boyeria* spp.	Sc	0	0	0	0	0	0	2	0	0	0
Calopterygidae	*Calopteryx* spp.	G	0	0	2	0	0	0	0	0	0	0
Gomphidae	*Gomphus* spp.	G	0	0	0	0	0	0	4	2	0	0
	Progomphus spp.	Sc	0	0	2	4	0	0	0	0	0	0
		G										
EPHEMEROPTERA		Sc										
Baetidae	*Baetis* spp.		67	53	73	18	78	29	29	20	267	84
Caenidae	*Caenis* spp.		0	0	2	0	0	0	0	0	0	0
Ephemerellidae	*Ephemerella* spp.	P	0	0	0	0	11	13	31	29	4	11
	Eurylophella spp.	P	0	0	0	0	0	2	0	0	0	0
	Epeorus spp.	P	2	0	0	0	0	0	0	0	0	0
Heptageneiudae	*Stenonema* spp.	P	4	4	4	18	0	13	20	78	0	4
MEGALOPTERA			0	0	0	0	0	0	0	0	0	0
Corydalidae	*Corydalus* spp.	P	0	0	4	0	2	0	0	0	0	0
COLEOPTERA												
Curculionidae	*Listronotus* spp.	Sh	0	0	0	0	2	0	0	0	0	0
Elmidae	*Ancyronyx* spp.	Sc	0	0	0	0	0	0	4	0	0	0
	Macronychus spp.	Sc	0	0	0	2	0	0	0	0	0	0
	Optioservus spp.	Sc	13	4	2	0	16	11	0	16	0	0
	Oulimnius spp.	Sc	2	0	0	0	2	0	2	0	7	2
	Stenelemis spp.	Sc	2	0	2	0	0	0	0	4	0	0
Hydraenidae	*Ochthebius* spp.	G?	0	2	0	0	0	0	0	0	7	0
Psphenidae	*Psphenus* spp.	Sc	0	0	0	0	0	2	0	0	2	0
HEMIPTERA			0	0	0	0	0	0	0	0	0	0
Veliidae	*Microvelia* spp.	P	0	0	0	0	0	0	0	0	0	4
PLECOPTERA												
Chloroperlidae	*Suwallia* spp.	P	0	9	0	0	0	0	7	4	2	0
Nemouridae	*Amphinemura* spp.	Sh	0	0	0	0	2	7	0	4	2	0
Perliidae	*Perlesta* spp.	Sh	0	0	0	0	0	7	0	0	4	0
Perlodidae	unknown	P	0	0	0	0	0	0	2	0	0	0

Appendix B.6. Continued.

			Clark		Copper Sandy		Lawrence		Picketts Mill		Westbrook	
		FFG	open	forest	open	forest	open	forest	open	forest	open	forest
TRICHOPTERA												
Calamoceratidae	*Anisocentropus* spp.	Sh	0	0	4	0	0	0	0	0	0	0
Hydropsychidae	*Cheumatopsyche* spp.	F	253	20	80	2	0	11	36	53	16	4
	Hydropsyche spp.	F	191	11	4	0	0	16	4	9	7	2
Philopotamidae	*Chimarra* spp.	F	44	4	0	0	0	0	2	9	0	0
Polycentropodidae	*Cyrnellus* spp.	F	0	0	0	0	0	0	2	0	0	0
	Polycentropus spp.	F	0	0	0	0	0	0	0	0	2	0
Psychomiidae	*Psychomyia* spp.	G	0	0	0	0	0	0	0	0	2	0
Pupae	unknown		9	0	4	0	0	2	0	0	2	0
DIPTERA												
Ceratopogonidae	Ceratopogoninae	P	0	0	0	0	9	0	4	7	2	2
Chironomidae	Non-Tanypodinae	G	273	469	987	251	242	102	320	329	1062	636
	Tanypodinae	P	0	0	0	0	0	0	0	2	0	4
Empididae	*Chelifera* spp.	P	0	0	0	4	0	0	0	0	0	0
	Hemerodromia spp.	P	11	2	0	0	0	4	2	4	2	0
	Unknown	P	0	0	0	0	0	0	0	0	0	0
Simuliidae	*Prosimulium* spp.	F	0	0	0	0	13	9	0	0	0	0
	Simulium spp.	F	44	36	91	18	18	31	7	13	62	13
Tipulidae	*Antocha* spp.	G	4	0	2	0	0	7	2	2	0	0
	Hexatoma spp.	P	0	2	0	0	0	2	0	2	0	0
	Molophilus spp.	Sh	0	0	0	0	0	0	2	0	0	0
	Tipula spp.	Sh	0	2	0	2	2	0	9	0	0	0
Pupae	unknown		29	31	160	33	18	27	31	38	196	136
	Hemerodromia spp.		7	0	22	0	0	0	2	27	0	4
	Simulidae		0	0	0	0	0	0	0	2	0	0
	Antocha spp.		4	0	2	0	0	0	0	0	0	0
LEPIDOPTERA												
Cosmopterigidae	unknown	Sh	0	0	2	0	0	0	0	0	0	0
Unknown		Sh	0	0	0	0	0	0	0	0	0	2

Appendix B.7. Density of terrestrial invertebrate orders collected in a 24-hour period in the summer in 10 floating pan traps (surface area = 0.344 m^2 each) at each site.

	Clark		Copper Sandy		Lawrence		Picketts Mill		Westbrook		OPEN		FOREST			
	open	forest	open	forest	open	forest	open	forest	open	forest	mean	st. dev.	mean	st. dev.	t	p
Total density (no. m^{-2})	90.7	73.5	409.6	79.9	518.0	156.7	212.2	104.3	142.4	123.8	274.6	182.1	107.7	33.9	2.23	0.045
Total biomass (g m^{-2})	0.03	0.33	0.07	0.03	0.04	0.02	0.03	0.04	0.06	0.15	0.05	0.02	0.11	0.13	1.09	0.168
Density common invertebrates (> 1 m^{-2})																
Arachnida	2.3	10.2	15.4	10.5	10.5	11.3	11.0	6.5	6.1	6.4	9.1	5.0	9.0	2.4	0.04	0.485
Coleoptera	5.2	5.5	4.4	1.7	7.0	3.2	8.7	7.8	10.5	12.5	7.2	2.5	6.1	4.2	0.98	0.192
Collembola	3.2	2.9	67.7	4.7	192.4	6.4	118.2	24.9	23.3	43.0	81.0	76.4	16.4	17.3	1.76	0.076
Diptera	62.2	33.4	287.2	33.1	252.9	103.2	50.7	37.8	83.4	43.9	147.3	113.3	50.3	29.9	2.10	0.052
Hemiptera	4.4	4.9	3.5	6.1	4.9	11.3	5.8	10.7	7.8	4.4	5.3	1.7	7.5	3.3	1.27	0.137
Homoptera	0.6	1.7	11.6	9.9	2.0	7.6	2.3	1.0	0.3	2.3	3.4	4.7	4.5	4.0	0.87	0.217
Hymenoptera	6.7	12.5	8.4	4.9	37.8	9.0	6.8	8.1	7.6	7.0	13.4	13.6	8.3	2.8	0.84	0.223
Density uncommon invertebrates (< 1 m^{-2})																
Acari	0.0	0.0	0.0	0.6	0.6	0.0	0.0	0.3	0.0	0.0	0.1	0.3	0.2	0.3	0.33	0.378
Diplopoda	0.0	0.0	0.0	0.0	0.3	0.0	0.6	0.0	0.0	0.0	0.2	0.3	0.0	0.0	1.47	0.108
Ephemeroptera	0.6	0.0	0.9	0.0	2.6	0.3	1.6	0.3	1.2	0.9	1.4	0.8	0.3	0.4	3.02	0.019
Lepidoptera	0.3	0.3	1.7	1.2	0.3	0.6	0.3	1.6	0.0	0.0	0.5	0.7	0.7	0.7	0.65	0.275
Odonata	0.9	0.3	0.9	1.5	0.0	0.6	0.3	0.0	0.0	0.3	0.4	0.4	0.5	0.6	0.46	0.335
Orthoptera	1.2	0.0	0.9	0.0	3.2	0.6	2.9	0.3	0.9	0.3	1.8	1.2	0.2	0.2	3.61	0.011
Plecoptera	0.0	0.0	0.9	0.6	0.0	0.0	0.0	0.0	0.0	0.3	0.2	0.4	0.2	0.3	0.00	0.500
Pscoptera	1.2	0.0	1.7	1.5	2.0	1.5	0.3	2.3	0.0	0.6	1.1	0.9	1.1	0.9	0.18	0.433
Thysanoptera	0.6	0.3	0.0	2.0	0.3	0.3	0.3	0.6	0.0	1.2	0.2	0.2	0.9	0.7	1.52	0.101
Thysanura	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.3	0.0	0.0	0.1	0.1	0.1	0.1	0.00	0.500
Trichoptera	1.5	1.5	4.4	1.7	1.2	0.9	1.9	1.9	1.5	0.9	2.1	1.3	1.4	0.5	1.42	0.114

Appendix B.8. Percent abundance of food items within stomachs of *Eurycea cirrigera*, the southern two-lined salamander, averaged for each stream reach.

	Clark		Lawrence		Picketts Mill		OPEN		FOREST			
	open	forest	open	forest	open	forest	mean	st. dev.	mean	st. dev.	t	*p*
Insecta												
Collembola	5.0	0.0	0.0	0.0	0.0	0.0	1.7	2.9	0	0.0	1.00	0.211
Coleoptera	2.5	3.4	0.0	0.0	3.5	1.3	2.0	1.8	1.566	1.7	0.46	0.346
Tricoptera	5.0	3.4	0.0	28.6	0.0	0.0	1.7	2.9	10.67	15.6	0.92	0.227
Ephemeroptera	5.0	0.0	66.7	28.6	0.0	6.3	23.9	37.1	11.61	15.0	0.92	0.227
Chironomidae	17.5	44.8	33.3	42.9	77.2	75.0	42.7	30.9	54.23	18.0	1.35	0.155
Dipteran pupae	5.0	6.9	0.0	0.0	0.0	1.3	1.7	2.9	2.716	3.7	1.88	0.100
Hymenoptera	7.5	6.9	0.0	0.0	1.8	0.0	3.1	3.9	2.299	4.0	1.53	0.133
Other Invertebrates												
Cladocera	22.5	10.3	0.0	0.0	0.0	0.0	7.5	13.0	3.448	6.0	1.00	0.211
Ostracod	0.0	6.9	0.0	0.0	1.8	0.0	0.6	1.0	2.299	4.0	0.65	0.291
Copepoda	0.0	6.9	0.0	0.0	14.0	12.5	4.7	8.1	6.466	6.3	0.69	0.281
Mite	0.0	0.0	0.0	0.0	0.0	1.3	0.0	0.0	0.417	0.7	1.00	0.211
Oligocheatea	22.5	3.4	0.0	0.0	0.0	2.5	7.5	13.0	1.983	1.8	0.81	0.251
Plant material	7.5	6.9	0.0	0.0	1.8	0.0	3.1	3.9	2.299	4.0	1.53	0.133

Appendix B.9. Fish assemblage measures calculated for each stream reach.

	Clark		Copper Sandy		Lawrence		Picketts Mill		Westbrook		OPEN		FOR
	open	forest	open	forest	open	forest	open	forest	open	forest	mean	st. dev.	mean
Overall measures													
Species richness	16	16	18	14	11	10	17	16	18	18	16	2.9	14.8
Density (no. m^{-2})	8.0	4.1	13.0	7.6	6.4	5.6	9.0	3.7	8.6	3.7	9.0	2.4	4.9
Abundance[a]	2.9	2.8	2.7	2.6	2.6	2.5	2.9	2.7	3.1	2.6	2.8	0.2	2.6
Rel. abundance 1 dominant spp.	0.52	0.32	0.30	0.45	0.41	0.19	0.36	0.25	0.29	0.24	0.37	0.10	0.29
Rel. abundance 2 dominant spp.	0.69	0.52	0.48	0.61	0.60	0.36	0.55	0.42	0.47	0.24	0.56	0.09	0.43
Endemic & cospopolitan species													
Endemic richness	2	2	1	0	1	1	3	3	3	2	2	1	1.6
Endemic density	1.14	0.32	0.25	0.00	0.72	0.94	2.25	1.72	0.91	1.29	1.05	0.74	0.85
Cosmopolitan richness	9	8	10	10	7	6	8	7	9	10	8.6	1.1	8.2
Cosmopolitan density	6.06564	3.20831	9.59404	5.93294	2.62393	2.56163	4.47298	1.3385	3.32945	1.11657	5.21721	2.8	2.83159
E/C (richness)	0.22	0.25	0.10	0.00	0.14	0.17	0.38	0.43	0.33	0.20	0.23	0.12	0.21
E/C (abundance)	0.12	0.07	0.02	0.00	0.18	0.32	0.38	0.94	0.37	0.73	0.21	0.16	0.41
E/E+C (abundance)	0.11	0.07	0.02	0.00	0.15	0.25	0.27	0.48	0.27	0.42	0.16	0.11	0.24
Tolerants													
Relative abundance	0.79	0.81	0.70	0.79	0.45	0.58	0.54	0.46	0.37	0.42	0.57	0.17	0.61
Density	5.89	3.22	8.65	5.55	3.41	3.09	4.87	1.49	4.10	1.60	5.38	2.04	2.99
Habitat generalists													
Relative abundance	0.54	0.39	0.52	0.35	0.26	0.44	0.47	0.33	0.36	0.31	0.43	0.12	0.37
Density	4.22	1.16	6.30	2.67	2.50	2.44	4.43	1.12	4.03	1.31	4.30	1.35	1.74
Pool/pool-run habitats													
Relative abundance	0.369	0.550	0.324	0.610	0.738	0.557	0.412	0.455	0.641	0.659	0.497	0.182	0.566
Rel. abund.:% pool/pool-run area	0.004	0.013	0.004	0.008	0.008	0.006	0.006	0.006	0.007	0.008	0.006	0.001	0.008
Density	2.65	2.64	3.93	4.28	3.92	3.13	2.94	1.45	4.46	2.02	3.58	0.75	2.70
Riffle-run habitats													
Relative abundance	0.093	0.055	0.158	0.042	0.000	0.000	0.123	0.213	0.004	0.032	0.075	0.071	0.068
Rel. abund.: % riffle area	0.005	0.001	0.006	0.002	0.000	0.000	0.005	0.008	0.001	0.002	0.003	0.003	0.003
Density	1.10	0.29	2.76	0.68	0.00	0.00	1.65	1.11	0.13	0.33	1.13	1.14	0.48
Community composition													
Darter & sculpin richness	2	2	3	1	1	1	3	3	3	3	2.4	0.9	2.0
Darter & sculpin rel. abundance	0.10	0.06	0.02	0.00	0.00	0.00	0.12	0.21	0.02	0.09	0.05	0.06	0.07
Darter & sculpin density	1.17	0.32	0.69	0.42	0.18	0.35	1.91	1.20	0.43	0.64	0.88	0.68	0.59
Centrarchid richness	6	7	7	5	5	4	5	4	5	5	5.6	0.9	5.0
Centrarchid rel. abundance	0.29	0.49	0.25	0.52	0.25	0.21	0.09	0.14	0.03	0.15	0.18	0.12	0.30
Centrarchid density	1.94	2.34	2.98	3.64	1.34	1.00	0.50	0.43	0.16	0.37	1.38	1.13	1.56
Insectivorous cyprinid richness	2	2	1	0	1	1	4	4	6	5	2.8	2.2	2.4
Insectivorous cyprinid rel. abund.[b]	0.09	0.15	0.06	0.00	0.27	0.41	0.31	0.49	0.70	0.72	0.29	0.26	0.35
Insectivorous cyprinid density	0.05	0.13	0.06	0.00	0.72	0.94	0.74	0.75	3.07	1.63	0.93	1.24	0.69
Herbivore rel. abundance	0.52	0.33	0.30	0.07	0.06	0.18	0.36	0.25	0.18	0.08	0.28	0.17	0.18
Herbivore density	4.08	0.86	3.37	0.60	0.76	1.08	3.49	0.83	2.28	0.31	2.80	1.31	0.74

[a] Log (x+1) transformed

[b] Arcsin squareroot transformed

Appendix B.10. Assemblage measures calculated for fishes collected in riffle and riffle-run habitats for each stream reach.

	Clark		Copper Sandy		Lawrence		Picketts Mill		Westbrook		OPEN		FOREST	
	open	forest	open	forest	open	forest	open	forest	open	forest	mean	st. dev.	mean	st. dev.
Overall Measures														
Species richness	7	13	8	8	4	7	13	9	12	9	8.8	3.70	9.2	2.28
Density (no. m^{-2})	2.74	1.12	2.57	1.17	1.25	0.75	5.82	1.42	3.89	1.47	3.25	1.71	1.19	0.29
Abundance[a]	2.24	2.48	1.89	1.64	1.28	1.20	2.58	2.09	2.20	1.88	2.04	0.49	1.86	0.48
Relative Abundance														
Darters & sculpin	0.31	0.10	0.03	0.00	0.00	0.00	0.15	0.43	0.03	0.27	0.10	0.13	0.16	0.19
Centrarchids	0.03	0.25	0.04	0.36	0.16	0.13	0.02	0.00	0.01	0.00	0.05	0.06	0.15	0.16
Insectivorous cyprinids[b]	0.00	0.06	0.00	0.00	0.00	0.00	0.34	0.30	0.60	0.53	0.19	0.27	0.18	0.23
Hypentelium etowanum	0.01	0.04	0.10	0.16	0.47	0.38	0.08	0.07	0.06	0.13	0.14	0.19	0.16	0.13
Campostoma oligolepis	0.64	0.55	0.34	0.14	0.26	0.13	0.48	0.23	0.25	0.28	0.39	0.16	0.26	0.17
Fundulus stellifer	0.02	0.02	0.05	0.07	0.00	0.19	0.12	0.14	0.33	0.05	0.10	0.13	0.09	0.07
Percina nigrofasciata	0.00	0.00	0.00	0.02	0.11	0.13	0.02	0.03	0.01	0.01	0.03	0.04	0.04	0.05
Density														
Darters & sculpin	0.93	0.25	0.08	0.00	0.00	0.00	1.23	0.80	0.11	0.43	0.47	0.57	0.30	0.33
Darters & sculpin (all)	0.93	0.25	0.08	0.01	0.06	0.09	1.29	0.83	0.18	0.44	0.51	0.57	0.32	0.33
Centrarchids	0.08	0.27	0.09	0.20	0.10	0.09	0.06	0.00	0.02	0.00	0.07	0.03	0.11	0.12
Insectivorous cyprinids[b]	0.00	0.00	0.00	0.00	0.00	0.00	0.49	0.12	1.04	0.48	0.31	0.46	0.12	0.21
Hypentelium etowanum	0.02	0.04	0.24	0.19	0.69	0.21	0.41	0.06	0.28	0.25	0.33	0.24	0.15	0.09
Campostoma oligolepis	1.66	0.52	0.86	0.18	0.40	0.18	2.80	0.26	1.46	0.27	1.44	0.91	0.28	0.14
Fundulus stellifer	0.05	0.02	0.12	0.04	0.00	0.13	0.77	0.14	0.91	0.04	0.37	0.43	0.07	0.06
Percina nigrofasciata	0.00	0.00	0.00	0.01	0.06	0.09	0.06	0.03	0.07	0.01	0.04	0.04	0.03	0.04

[a] Log (x+1) transformed

[b] Arcsin squareroot transformed

Appendix B.11. Assemblage measures calculated for fishes collected in pool and run habitats for each stream reach.

	Clark		Copper Sandy		Lawrence		Picketts Mill		Westbrook		OPEN		FOREST	
	open	forest	open	forest	open	forest	open	forest	open	forest	mean	st. dev.	mean	st. dev.
Overall Measures														
Species richness	16	16	18	14	11	10	17	16	18	18	16	2.92	14.8	3.03
Density (no. m^{-2})	4.76	2.97	10.42	4.50	5.17	4.83	3.20	2.26	4.69	1.83	5.65	2.77	3.28	1.34
Abundance[a]	2.73	2.55	2.61	2.53	2.56	2.48	2.58	2.57	3.09	2.44	2.71	0.22	2.51	0.05
Relative Abundance														
Darters & sculpin	0.04	0.02	0.02	0.00	0.00	0.00	0.09	0.14	0.02	0.03	0.03	0.04	0.04	0.06
Centrarchids	0.37	0.69	0.28	0.54	0.26	0.22	0.15	0.19	0.03	0.18	0.22	0.13	0.36	0.23
Insectivorous cyprinids[b]	0.11	0.19	0.07	0.00	0.28	0.42	0.27	0.55	0.72	0.78	0.29	0.26	0.39	0.30
Hypentelium etowanum	0.02	0.07	0.16	0.17	0.10	0.15	0.04	0.02	0.02	0.05	0.07	0.06	0.09	0.06
Campostoma oligolepis	0.48	0.13	0.29	0.05	0.05	0.18	0.23	0.25	0.18	0.03	0.24	0.16	0.13	0.09
Fundulus stellifer	0.06	0.03	0.07	0.09	0.43	0.19	0.25	0.09	0.28	0.11	0.22	0.16	0.10	0.06
Percina nigrofasciata	0.00	0.00	0.04	0.05	0.02	0.06	0.04	0.02	0.04	0.04	0.03	0.02	0.03	0.02
Density														
Darters & sculpin	0.24	0.06	0.22	0.00	0.00	0.00	0.42	0.31	0.07	0.05	0.19	0.16	0.09	0.13
Centrarchids	1.68	2.07	2.89	2.56	1.24	0.92	0.44	0.43	0.14	0.31	1.28	1.09	1.26	1.01
Insectivorous cyprinids[b]	0.04	0.13	0.06	0.00	0.72	0.94	0.25	0.63	2.03	1.01	0.62	0.83	0.54	0.46
Hypentelium etowanum	0.08	0.22	1.92	0.44	0.50	0.58	0.14	0.04	0.09	0.13	0.54	0.79	0.28	0.22
Campostoma oligolepis	2.18	0.31	2.45	0.24	0.36	0.90	0.69	0.58	0.81	0.05	1.30	0.95	0.41	0.33
Fundulus stellifer	0.50	0.09	0.65	0.38	1.86	1.05	0.84	0.21	1.33	0.11	1.04	0.56	0.37	0.40
Percina nigrofasciata	0.00	0.00	0.00	0.30	0.11	0.26	0.20	0.05	0.18	0.11	0.10	0.10	0.14	0.13

[a] Log (x+1) transformed

[b] Arcsin squareroot transformed

Appendix B.12. Total fish catch in all habitats within each stream reach.

		Clark		Copper Sandy		Lawrence		Picketts Mill		Westbrook	
Scientific name	Common name	open	forest	open	forest	open	forest	open	forest	open	forest
Petromyzontidae											
Ichthyomyzon sp. cf. gagei	southern brook lamprey		7	2	2						
Cyprinidae											
Campostoma oligolepis	largescale stoneroller	367	209	142	24	23	56	271	121	257	30
Cyprinella callistia	Alabama shiner							8	12	158	47
Cyprinella venusta	blacktail shiner	1	10					2	1	44	11
Hybopsis sp. cf. winchelli	clear chub	5	4								
Notropis longirostris	longnose shiner									107	8
Notropis stilbius	silverstripe shiner			2				1	10	104	7
Notropis xaenocephalus	Coosa shiner					27	50	59	88	171	92
Phenacobius catostomus	riffle minnow									1	
Semotilus atromaculatus	creek chub		11	18	25	22	16	3	2		
Catostomidae											
Hypentelium etowanum	Alabama hog sucker	13	37	73	64	46	50	46	15	32	28
Moxostoma duquesnei	black redhorse							42	2	10	16
Moxostoma poecilurum	blacktail redhorse	4									
Ictaluridae											
Ameiurus brunneus	snail bullhead			1	1						
Ameiurus natalis	yellow bullhead	1									
Noturus leptacanthus	speckled madtom			69	16						1
Fundulidae											
Fundulus stellifer	southern studfish	37	18	31	33	157	60	143	52	400	38
Poecilidae											
Gambusia affinis	eastern mosquitofish				1						
Cottidae											
Cottus carolinae zopherus	Coosa banded sculpin			7				6	14		11
Centrarchidae											
Lepomis auritus	redbreast sunfish	36	42	17	21	72	52	30	31	14	22
Lepomis cyanellus	green sunfish	20	107	1	1	20					
Lepomis gulosus	warmouth										1
Lepomis macrochirus	bluegill sunfish	124	132	91	170	2	8	29	34	5	21
Lepomis microlophus	redear sunfish	7	6	2				1			
Lepomis spp.unknown YOY	unknown sunfish									8	
Micropterus coosae	Coosa bass		3	1		2	4	2	4	5	3
Micropterus punctulatus	spotted bass	6	19	3					2	6	8
Micropterus salmoides	largemouth bass	12	8	3	4	1	3	3			
Pomoxis nigromaculatus	black crappie				1						
Percidae											
Etheostoma scotti	Cherokee darter	66	36					87	91	4	
Etheostoma stigmaeum	speckled darter									20	22
Perca flavescens	yellow perch	1									
Percina kathae	Mobile logperch	8	2	3							
Percina nigrofasciata	blackbanded darter			15	19	9	19	25	13	50	12
Total no. ind.		708	651	481	382	381	318	758	492	1396	378
Total no. taxa		16	16	18	14	11	10	17	16	18	18

Appendix B.13. Fish catch from riffle and riffle-run habitats within each stream reach.

		Clark		Copper Sandy		Lawrence		Picketts Mill		Westbrook	
Scientific name	Common name	open	forest	open	forest	open	forest	open	forest	open	forest
Petromyzontidae											
Ichthyomyzon sp. cf. gagei	southern brook lamprey		4								
Cyprinidae											
Campostoma oligolepis	largescale stoneroller	110	165	26	6	5	2	184	28	40	21
Cyprinella callistia	Alabama shiner							3	4	29	8
Cyprinella venusta	blacktail shiner		1					2		1	
Hybopsis sp. cf. winchelli	clear chub										
Notropis longirostris	longnose shiner									12	1
Notropis stilbius	silverstripe shiner										
Notropis xaenocephalus	Coosa shiner							38	7	7	10
Phenacobius catostomus	riffle minnow									1	
Semotilus atromaculatus	creek chub		3		3		1		2		
Catostomidae											
Hypentelium etowanum	Alabama hog sucker	1	13	8	7	9	6	30	8	9	10
Moxostoma duquesnei	black redhorse										
Moxostoma poecilurum	blacktail redhorse										
Ictaluridae											
Ameiurus brunneus	snail bullhead				1						
Ameiurus natalis	yellow bullhead										
Noturus leptacanthus	speckled madtom			34	7						
Fundulidae											
Fundulus stellifer	southern studfish	3	7	4	3		3	47	17	52	4
Poecilidae											
Gambusia affinis	eastern mosquitofish										
Cottidae											
Cottus carolinae zopherus	Coosa banded sculpin			2				6	10		11
Centrarchidae											
Lepomis auritus	redbreast sunfish	2	7	1		3	1	1			
Lepomis cyanellus	green sunfish		26								
Lepomis gulosus	warmouth										
Lepomis macrochirus	bluegill sunfish		24	1	16			6			
Lepomis microlophus	redear sunfish		1					1			
Lepomis spp.unknown YOY	unknown sunfish										
Micropterus coosae	Coosa bass			1			1				
Micropterus punctulatus	spotted bass	1	12							1	
Micropterus salmoides	largemouth bass	2	6					1			
Pomoxis nigromaculatus	black crappie										
Percidae											
Etheostoma scotti	Cherokee darter	54	31					52	43	2	
Etheostoma stigmaeum	speckled darter									2	9
Perca flavescens	yellow perch										
Percina kathae	Mobile logperch										
Percina nigrofasciata	blackbanded darter				1	2	2	9	4	2	1
Total no. ind.		173	300	77	44	19	16	380	123	158	75
Total no. taxa		7	13	8	8	4	7	13	9	12	9

Appendix B.14. Fish catch from pool and run habitats within each stream reach.

		Clark		Copper Sandy		Lawrence		Picketts Mill		Westbrook	
Scientific name	Common name	open	forest	open	forest	open	forest	open	forest	open	forest
Petromyzontidae											
Ichthyomyzon sp. cf. gagei	southern brook lamprey		3	2	2						
Cyprinidae											
Campostoma oligolepis	largescale stoneroller	257	44	116	18	18	54	87	93	217	9
Cyprinella callistia	Alabama shiner							5	8	129	38
Cyprinella venusta	blacktail shiner	1	9						1	43	11
Hybopsis sp. cf. winchelli	clear chub	5	4								
Notropis longirostris	longnose shiner									95	6
Notropis stilbius	silverstripe shiner			2				1	10	104	7
Notropis xaenocephalus	Coosa shiner					27	50	21	81	164	73
Phenacobius catostomus	riffle minnow										
Semotilus atromaculatus	creek chub		8	18	22	22	15	3			
Catostomidae											
Hypentelium etowanum	Alabama hog sucker	12	24	65	57	37	44	16	7	23	15
Moxostoma duquesnei	black redhorse							42	2	10	16
Moxostoma poecilurum	blacktail redhorse	4									
Ictaluridae											
Ameiurus brunneus	snail bullhead			1							
Ameiurus natalis	yellow bullhead	1									
Noturus leptacanthus	speckled madtom			35	9						
Fundulidae											
Fundulus stellifer	southern studfish	34	11	27	30	157	57	96	35	348	29
Poecilidae											
Gambusia affinis	eastern mosquitofish				1						
Cottidae											
Cottus carolinae zopherus	Coosa banded sculpin			5					4		
Centrarchidae											
Lepomis auritus	redbreast sunfish	34	35	16	21	69	51	29	31	14	18
Lepomis cyanellus	green sunfish	20	81	1	1	20					
Lepomis gulosus	warmouth										1
Lepomis macrochirus	bluegill sunfish	124	108	90	154	2	8	23	34	5	21
Lepomis microlophus	redear sunfish	7	5	2							
Lepomis spp.unknown YOY	unknown sunfish									8	
Micropterus coosae	Coosa bass		3			2	3	2	4	5	2
Micropterus punctulatus	spotted bass	5	7	3					2	5	8
Micropterus salmoides	largemouth bass	10	2	3	4	1	3	2			
Pomoxis nigromaculatus	black crappie				1						
Percidae											
Etheostoma scotti	Cherokee darter	12	5					35	48	2	
Etheostoma stigmaeum	speckled darter									18	9
Perca flavescens	yellow perch	1									
Percina kathae	Mobile logperch	8	2	3							
Percina nigrofasciata	blackbanded darter			15	18	7	17	16	9	48	11
Total no. ind.		535	351	404	338	362	302	378	369	1238	274
Total no. taxa		16	16	18	14	11	10	17	16	18	18

APPENDIX C

SUPPLEMENTAL MATERIAL TO CHAPTER 3

Appendix C.1. Location and general site information for 66 study sites. Catchment % impervious surface cover from 2001 land cover data; road crossings per stream length from 1999 Department of Transportation road coverage. Catchment % land cover based on *Landsat* TM satellite imagery for most recent prior date (1992, 1998, 2001).

Site ID	Date	Stream Name	County	Basin Area (km^2)	Catchment % Imperv	Road Xings (no. km^{-1})	Catchment		
							Forest	Urban	Agriculture
19	6/11/1993	Stamp Creek	Bartow	21.6	4.0	0.18	88.2	0.5	2.3
49	7/7/1999	Conn Creek	Cherokee	19.5	4.7	0.04	84.2	0.8	4.4
68	8/13/2002	Buzzard Flapper	Cherokee	9.6	2.8	0.37	84.5	0.8	2.5
73	8/13/2002	Boardtree Creek	Cherokee	6.2	4.3	0.25	72.6	1.3	5.5
87	9/4/2002	Murphy Creek	Cherokee	12.2	1.7	0.32	96.5	1.4	0.4
95	8/7/2002	Hickory Log Creek	Cherokee	17.6	3.4	0.39	82.0	1.6	5.0
107	5/20/1999	Edward Cr.	Cherokee	9.5	3.9	0.27	77.0	1.7	5.3
135	5/22/2002	Canton Creek	Cherokee	10.6	4.1	0.36	78.9	1.8	10.7
140	8/5/2002	Scott Mill Creek	Cherokee	12.1	4.9	0.70	84.4	2.6	4.5
145	5/23/2002	Mill Creek	Cherokee	17.0	4.9	0.30	71.8	3.0	15.1
159	7/10/2001	Burt Creek	Dawson	36.3	2.5	0.15	94.2	3.0	0.8
183	8/5/2002	Toonigh Creek	Cherokee	7.4	5.4	1.05	71.3	3.1	18.0
192	5/25/1995	Jug Creek	Cherokee	11.8	4.8	0.21	84.3	3.3	10.5
197	6/1/1993	Etowah RIver	Cherokee	11.3	4.5	0.52	82.3	3.9	6.6
209	8/5/1999	McCannless Creek	Cherokee	12.0	5.7	0.24	57.7	4.0	31.2
227	10/15/2003	Butler Creek	Cobb	17.1	3.2	0.43	86.0	4.1	3.1
228	8/29/2000	Little Allatoona Creek	Cobb	8.1	4.8	0.35	86.8	4.3	3.9
231	5/20/2002	Proctor Creek	Cobb	7.7	4.8	0.39	71.8	4.3	14.0
234	3/28/1994	Noonday Creek	Cobb	14.6	5.1	0.36	75.7	4.6	16.2
245	10/6/1994	Tributary to Yellow Creek	Dawson	11.7	3.4	0.08	79.6	5.5	9.3
248	8/6/2002	Black Mill Creek	Dawson	12.8	5.1	0.41	75.8	5.6	14.7
253	7/11/2001	Palmer Creek	Dawson	9.6	4.3	0.21	90.6	5.7	0.5
318	5/30/2002	Yellow Creek	Forsyth	13.9	6.7	0.59	62.0	6.4	13.7
339	7/12/2002	Westbrook Creek	Paulding	10.2	7.1	0.19	72.3	6.5	6.0
342	7/9/2002	Lane Creek	Paulding	17.5	6.5	0.17	53.9	8.0	27.3
352	8/15/2002	Bluffy Creek	Paulding	8.8	3.3	0.02	78.6	8.1	1.1
356	8/30/2002	Norton Creek	Pickens	5.4	4.4	0.42	71.1	8.5	14.3
359	8/14/2002	Rock Creek	Pickens	6.5	4.7	0.18	82.3	8.6	0.9
360	7/16/2002	Sharp Mountain Creek	Pickens	11.4	9.7	0.16	73.9	8.6	9.3
362	8/5/2002	Four Mile Creek	Pickens	17.1	9.1	0.29	65.3	9.0	21.8
387	8/6/2002	Polecat Branch	Pickens	5.9	3.9	0.32	87.2	9.4	1.4
394	8/10/2000	Tributary to Amicalola Creek	Dawson	10.9	4.9	0.35	64.6	9.6	17.1
419	7/17/2002	Disharoon Creek	Pickens	5.6	3.4	0.49	73.0	9.7	11.6
420	8/7/2002	Town Creek	Pickens	16.2	7.2	0.33	60.3	10.5	16.2
439	7/5/2000	Tributary to Sharp Mtn. Creek	Pickens	13.8	6.9	0.63	50.3	12.4	29.1

Appendix C.1. Continued.

Site				Basin Area	Catchment	Road Xings	Catchment		
ID	Date	Stream Name	County	(km^2)	% Imperv	(no. km^{-1})	Forest	Urban	Agric
441	8/13/2002	Boston Creek	Bartow	15.5	8.9	0.22	49.5	13.2	29.5
451	7/9/2002	Upper Little Pumpkinvine Creek	Paulding	10.5	8.7	0.14	76.3	13.5	1.0
452	7/10/2002	West Fork Pumpkinvine	Paulding	5.5	8.0	0.16	70.8	13.6	3.1
453	8/2/1999	Avery Creek	Cherokee	26.8	7.4	0.47	48.5	14.0	26.7
454	8/14/2002	Smithwick Creek	Cherokee	9.9	7.1	0.29	55.8	14.0	22.1
455	9/20/2002	Gorman Branch	Cherokee	16.3	9.7	0.00	63.1	14.4	10.4
456	8/11/1999	Bluff Creek	Cherokee	15.7	8.0	0.24	51.3	15.6	24.3
457	7/6/1999	Settingdown Creek	Forsyth	21.0	3.1	0.45	74.0	17.4	0.1
458	8/23/2000	Sweetwater Creek	Cherokee	8.9	3.0	0.36	74.5	18.1	6.5
459	7/7/2000	Knox Creek	Cherokee	7.4	7.0	0.33	69.6	19.1	3.2
472	7/5/2000	Tributary to Etowah River	Cherokee	11.2	8.5	0.39	65.7	20.4	5.6
538	5/7/2001	Puckett Creek	Cherokee	15.2	13.4	1.39	64.4	20.4	6.2
557	3/16/2001	Tributary to Etowah River	Dawson	12.8	14.7	0.31	55.9	20.9	13.6
559	3/22/2001	Mill Creek	Dawson	12.0	9.4	0.32	59.5	21.2	9.2
565	5/30/2001	Bannister Creek	Forsyth	13.9	10.1	0.46	58.9	22.4	9.9
612	7/24/2001	Rose Creek	Cherokee	14.1	9.9	0.67	61.4	22.6	10.6
623	10/10/2001	Clark Creek	Cherokee	9.9	14.5	0.64	54.4	23.5	17.4
624	10/24/2001	Copper Sandy Creek	Fulton	19.0	12.7	0.32	60.9	24.8	8.4
627	7/9/2002	Lawrence Creek	Paulding	17.0	27.0	0.60	50.0	25.2	13.4
629	8/6/2002	Pickets Mill Creek	Paulding	12.6	13.4	0.60	60.8	25.7	7.9
646	7/10/2002	Possum Creek	Paulding	23.0	10.7	0.33	43.5	25.9	23.4
650	3/24/1999	Darnell Creek	Pickens	9.8	14.3	0.34	46.2	47.1	2.1
685	8/8/2002	Richland Creek	Bartow	14.7	15.8	0.25	36.2	48.7	7.7
686	8/14/2002	Chicken Creek	Fulton	19.7	29.8	0.29	34.8	50.4	2.8
692	6/26/2002	Allatoona Creek	Cobb	12.2	18.2	0.36	38.3	52.2	3.1
696	7/16/2002	Ward Creek	Bartow	5.6	23.6	0.15	31.9	54.6	4.1
697	7/22/2002	Tributary to Noonday Creek	Cobb	14.7	22.0	0.68	37.3	57.1	1.7
698	7/22/2002	Little Noonday Creek	Cobb	11.4	21.9	0.45	32.7	61.2	0.2
699	7/26/2002	Tributary to the Little River	Cherokee	11.2	20.2	0.55	34.8	61.4	0.3
700	8/15/2002	East Fork Rubes Creek	Cobb	9.8	31.1	0.81	28.8	65.1	2.0
701	8/15/2002	West Fork Rubes Creek	Cobb	17.2	23.9	0.43	30.4	65.2	0.1
Mean				36.3	29.8	0.37	96.5	61.4	31.2
Min				13.4	12.1	0.00	58.0	25.5	10.6
Max				36.3	29.8	1.39	96.5	61.4	31.2

Appendix C.2. Riparian % land cover data for 30 m and 100 m buffers for 200 m, 1 km, and entire drainage network upstream of the study site.

Site	Riparian, network (100)			Riparian, network (30)			Riparian, 1 km (100)			Riparian, 1 km (30)			Riparian, 200 m (100)			Riparian, 200 m (30)		
ID	Forest	Urban	Agric	Forest	Urban	Agric	Forest	Urban	Agric	Forest	Urban	Agric	Forest	Urban	Agric	Forest	Urban	Agric
19	89.3	0.1	1.9	90.2	0.0	1.4	91.0	0.0	1.9	95.5	0.0	1.5	93.9	0.0	0.0	92.9	0.0	0.0
49	86.4	0.6	3.6	87.1	0.5	3.8	88.4	0.0	3.9	100.0	0.0	0.0	100.0	0.0	0.0	100.0	0.0	0.0
68	88.2	0.5	3.2	90.2	0.3	4.0	83.6	0.0	4.7	77.9	0.0	10.3	86.2	0.0	3.4	75.0	0.0	0.0
73	78.6	0.9	3.5	85.4	0.0	1.4	88.2	0.0	0.0	98.6	0.0	0.0	80.3	0.0	0.0	100.0	0.0	0.0
87	97.7	0.2	0.5	98.6	0.2	0.5	88.9	0.0	2.1	91.0	0.0	3.0	98.3	0.0	1.7	93.3	0.0	6.7
95	87.1	1.3	2.8	91.8	0.6	1.8	77.9	6.9	2.8	90.9	6.1	1.5	73.7	14.0	5.3	85.7	7.1	0.0
107	82.9	0.9	0.9	93.1	0.7	0.3	93.0	0.0	0.0	100.0	0.0	0.0	94.6	0.0	0.0	100.0	0.0	0.0
135	81.8	0.5	11.2	83.6	0.1	9.8	52.6	0.0	44.9	53.6	0.0	46.4	40.0	0.0	58.3	78.6	0.0	21.4
140	84.6	2.0	3.7	86.7	1.5	1.8	95.4	0.0	0.0	98.5	0.0	0.0	96.4	0.0	0.0	100.0	0.0	0.0
145	72.4	2.0	17.4	72.8	1.4	17.5	98.3	0.0	0.9	96.9	0.0	1.6	94.4	0.0	3.7	85.7	0.0	7.1
159	95.0	1.9	0.9	96.3	0.9	0.7	96.9	0.0	0.0	95.5	0.0	0.0	95.1	0.0	0.0	93.3	0.0	0.0
183	77.9	0.7	14.0	82.0	0.1	9.7	11.2	3.0	83.2	19.4	0.0	76.1	15.8	3.5	73.7	50.0	0.0	28.6
192	88.8	1.5	7.4	87.7	1.5	7.0	40.6	0.0	59.0	25.4	0.0	74.6	9.4	0.0	90.6	11.8	0.0	88.2
197	89.1	0.9	4.8	90.9	0.7	3.6	71.7	0.0	21.2	82.4	0.0	17.6	58.6	0.0	39.7	80.0	0.0	20.0
209	65.7	2.3	24.7	71.1	1.3	17.8	78.3	0.9	0.0	92.4	0.0	0.0	63.0	0.0	0.0	92.9	0.0	0.0
227	89.6	1.9	4.0	91.2	1.1	4.1	84.8	2.2	9.8	94.0	0.0	1.5	89.7	0.0	0.0	80.0	0.0	0.0
228	91.1	2.7	2.5	94.2	1.7	1.3	96.9	0.0	0.0	92.5	0.0	0.0	94.5	0.0	0.0	87.5	0.0	0.0
231	76.7	3.3	11.2	83.0	3.2	7.3	68.6	1.3	24.9	82.6	2.9	4.3	62.1	5.2	22.4	60.0	13.3	6.7
234	82.0	2.3	11.0	82.3	1.8	8.6	80.4	0.4	16.3	88.6	0.0	8.6	79.1	1.5	17.9	86.7	0.0	13.3
245	88.7	1.0	6.6	94.0	0.0	3.7	98.7	0.0	0.0	100.0	0.0	0.0	98.2	0.0	0.0	100.0	0.0	0.0
248	80.5	2.7	13.0	83.8	1.9	10.4	25.2	9.3	49.1	30.4	7.2	50.7	0.0	13.0	87.0	0.0	0.0	100.0
253	92.2	3.5	0.4	94.1	2.1	0.3	100.0	0.0	0.0	100.0	0.0	0.0	100.0	0.0	0.0	100.0	0.0	0.0
318	73.6	3.3	8.8	78.5	1.8	7.5	58.8	1.8	34.2	59.4	2.9	30.4	16.4	1.6	77.0	12.5	0.0	81.3
339	76.1	3.4	4.3	78.9	1.2	2.5	77.8	2.1	11.9	83.8	1.5	5.9	47.6	4.8	30.2	53.3	6.7	20.0
342	62.1	5.7	18.5	67.6	5.8	10.3	67.0	3.6	14.7	76.8	5.8	0.0	85.7	1.8	10.7	92.9	0.0	0.0
352	85.2	3.8	0.7	88.4	3.3	0.6	98.3	0.0	0.0	100.0	0.0	0.0	93.4	0.0	0.0	100.0	0.0	0.0
356	74.3	5.3	15.1	77.1	5.4	11.0	86.8	0.0	12.8	74.6	0.0	25.4	61.0	0.0	39.0	46.7	0.0	53.3
359	82.0	8.5	0.7	85.3	9.0	0.0	91.1	0.0	1.3	88.2	0.0	0.0	86.4	0.0	0.0	86.7	0.0	0.0
360	78.2	6.0	9.9	81.3	4.8	7.6	74.4	1.7	17.5	92.5	0.0	1.5	55.6	3.2	27.0	75.0	0.0	6.3
362	74.3	6.4	14.0	75.9	6.0	10.2	47.7	2.1	48.1	49.3	0.0	49.3	9.7	3.2	83.9	7.1	0.0	92.9
387	93.5	4.2	0.5	95.0	2.6	0.0	93.9	0.0	0.9	95.7	0.0	0.0	100.0	0.0	0.0	100.0	0.0	0.0
394	74.1	4.6	13.0	81.2	3.4	8.0	72.4	10.3	7.8	87.5	9.7	0.0	47.5	37.3	6.8	56.3	43.8	0.0
419	81.3	1.2	5.6	77.3	4.8	7.8	90.0	0.0	3.9	88.1	0.0	6.0	91.1	0.0	5.4	100.0	0.0	0.0
420	72.8	6.9	8.8	82.8	6.0	3.1	89.4	6.6	0.4	94.0	3.0	0.0	100.0	0.0	0.0	100.0	0.0	0.0
439	62.4	6.9	19.1	65.0	5.9	11.6	74.7	3.9	12.4	74.6	1.5	17.9	56.9	10.3	24.1	42.9	0.0	50.0

Appendix C.2. Continued.

Site	Riparian, network (100)			Riparian, network (30)			Riparian, 1 km (100)			Riparian, 1 km (30)			Riparian, 200 m (100)			Riparian, 200 m (30)		
ID	Forest	Urban	Agric	Forest	Urban	Agric	Forest	Urban	Agric	Forest	Urban	Agric	Forest	Urban	Agric	Forest	Urban	Agric
441	62.3	10.3	18.7	70.1	8.8	11.4	68.2	2.9	24.3	71.8	2.8	19.7	94.9	5.1	0.0	100.0	0.0	0.0
451	84.6	6.5	1.0	91.8	2.9	0.8	96.6	0.0	0.0	92.5	0.0	0.0	100.0	0.0	0.0	100.0	0.0	0.0
452	79.1	9.5	1.4	82.8	5.7	0.2	96.6	0.0	2.5	100.0	0.0	0.0	90.6	0.0	9.4	100.0	0.0	0.0
453	55.5	10.1	21.0	59.6	8.6	15.5	23.8	0.4	66.3	25.4	0.0	71.6	12.1	0.0	81.8	14.3	0.0	78.6
454	64.6	10.1	16.2	70.6	8.3	10.7	55.5	2.1	27.7	58.8	5.9	23.5	48.3	5.0	21.7	62.5	6.3	0.0
455	71.9	10.1	6.1	77.4	7.0	4.5	95.0	1.4	0.5	97.1	0.0	0.0	100.0	0.0	0.0	100.0	0.0	0.0
456	65.5	8.4	16.8	75.2	6.0	9.7	64.6	5.1	23.2	69.0	1.4	22.5	33.9	12.9	37.1	60.0	6.7	13.3
457	72.4	13.7	0.2	70.5	10.5	0.3	81.8	16.1	0.4	84.1	13.0	0.0	75.0	16.1	1.8	87.5	0.0	0.0
458	76.8	15.2	6.4	80.5	11.9	4.2	92.5	0.0	4.4	88.6	0.0	4.3	69.2	0.0	15.4	46.7	0.0	20.0
459	76.5	14.3	2.8	78.1	15.5	2.0	22.8	69.7	0.0	1.5	98.5	0.0	15.8	82.5	0.0	0.0	100.0	0.0
472	73.3	11.5	5.6	78.4	7.2	4.3	41.4	17.3	24.1	44.1	5.9	30.9	19.0	7.9	52.4	0.0	5.9	64.7
538	68.1	15.9	6.3	71.5	11.3	5.9	83.0	1.8	8.9	82.6	1.4	10.1	59.3	5.1	15.3	37.5	6.3	37.5
557	64.7	15.5	7.4	70.6	10.0	4.9	77.3	5.8	11.2	84.3	4.3	4.3	62.9	0.0	30.6	73.3	0.0	20.0
559	70.1	13.6	6.2	74.5	13.4	3.6	65.0	2.1	0.0	60.6	0.0	0.0	72.4	0.0	0.0	50.0	0.0	0.0
565	67.6	17.9	5.9	76.7	12.2	2.5	66.5	8.1	16.5	65.8	6.8	19.2	90.3	4.8	3.2	93.8	0.0	0.0
612	68.9	13.3	10.0	72.2	10.8	7.4	53.3	27.3	9.7	62.1	24.2	9.1	28.1	49.1	0.0	35.7	42.9	0.0
623	61.7	19.2	13.3	65.3	16.5	12.7	65.0	11.7	7.6	76.1	10.4	0.0	65.4	7.7	26.9	100.0	0.0	0.0
624	66.2	23.7	4.7	74.0	17.0	3.3	70.0	15.2	10.4	68.7	11.9	11.9	77.6	13.8	1.7	86.7	6.7	0.0
627	56.0	19.5	10.4	57.7	16.4	7.9	77.8	1.3	7.8	90.9	0.0	6.1	77.0	0.0	3.3	100.0	0.0	0.0
629	69.9	20.3	4.3	76.5	14.2	2.8	69.1	11.7	12.1	73.8	3.1	9.2	50.0	10.0	26.7	26.7	13.3	33.3
646	53.4	19.9	17.1	58.7	15.7	13.0	32.8	26.9	32.4	50.7	17.4	17.4	36.5	20.6	28.6	26.7	40.0	6.7
650	51.3	40.0	2.6	57.3	32.0	2.7	72.5	24.6	0.8	87.3	9.9	0.0	62.7	37.3	0.0	71.4	28.6	0.0
685	47.4	37.4	6.7	55.7	31.8	4.7	62.5	8.3	4.6	64.7	7.4	1.5	70.6	7.8	5.9	40.0	20.0	6.7
686	43.6	42.1	1.2	54.6	31.7	1.0	71.7	21.3	1.3	88.6	10.0	0.0	71.2	25.4	0.0	78.6	21.4	0.0
692	42.5	49.7	0.8	50.4	43.2	0.2	26.5	0.4	0.0	39.1	0.0	0.0	4.2	0.0	0.0	5.9	0.0	0.0
696	38.7	49.1	2.6	48.3	41.8	1.7	42.4	28.4	4.2	72.1	13.2	0.0	61.3	4.8	6.5	100.0	0.0	0.0
697	42.3	52.7	0.9	51.4	44.0	0.6	71.6	21.0	0.0	88.4	5.8	0.0	66.1	30.4	0.0	93.8	6.3	0.0
698	42.1	50.4	0.2	55.0	37.2	0.1	50.4	46.6	0.0	70.1	23.9	0.0	26.7	71.7	0.0	28.6	64.3	0.0
699	38.8	56.9	0.1	50.6	44.9	0.1	77.8	19.8	0.0	94.0	1.5	0.0	86.6	9.0	0.0	86.7	0.0	0.0
700	40.4	53.1	1.0	51.0	42.3	0.6	54.1	38.8	0.4	75.0	22.2	1.4	57.4	36.8	1.5	82.4	11.8	5.9
701	38.8	56.2	0.1	48.5	44.4	0.1	59.2	13.3	0.4	73.5	7.4	0.0	67.7	13.8	1.5	68.8	18.8	0.0
Mean	97.7	56.9	24.7	98.6	44.9	17.8	100.0	69.7	83.2	100.0	98.5	76.1	100.0	82.5	90.6	100.0	100.0	100.0
Min	65.4	20.2	7.6	70.9	16.2	5.3	69.1	12.2	12.2	74.8	9.6	10.6	62.3	13.8	17.0	65.1	12.2	15.3
Max	97.7	56.9	24.7	98.6	44.9	17.8	100.0	69.7	83.2	100.0	98.5	76.1	100.0	82.5	90.6	100.0	100.0	100.0

Appendix C.3. Abundance, richness, and estimated richness (based on first order jackknife estimator) data for fish assemblage metrics at the 66 study sites. Numbers in parentheses indicate total number of species in category.

	Total		Endemic (7)			Cosmopolitan (23)			Endemic:Cosmopolitan			Sensitive (8)			Len Tolerant (13)		
ID	Abund	Rich	Abund	Rich	Est Rich	Abund	Rich	Est Rich	Abund	Rich	Est Rich	Abund	Rich	Est Rich	Abund	Rich	Est Rich
19	145	12	71	4	4	27	4	5	2.6	1.0	0.8	50	2	2	5	2	3
49	257	11	124	3	4	51	3	3	2.4	1.0	1.3	1	1	1	0	0	0
68	190	9	38	1	1	97	5	5	0.4	0.2	0.2	4	1	1	43	2	2
73	162	12	0	0	0	117	9	10	0.0	0.0	0.0	20	2	2	48	4	5
87	462	20	200	3	3	154	8	9	1.3	0.4	0.3	7	2	2	69	4	5
95	627	12	333	5	5	218	5	5	1.5	1.0	1.0	38	2	2	60	2	2
107	178	21	34	3	3	68	11	15	0.5	0.3	0.2	27	4	5	13	5	8
135	235	14	42	2	2	162	8	10	0.3	0.3	0.2	12	2	2	124	3	4
140	200	13	83	3	3	90	6	8	0.9	0.5	0.4	3	1	1	10	2	3
145	196	10	2	1	1	132	6	6	0.0	0.2	0.2	0	0	0	101	3	3
159	258	18	64	6	7	98	6	8	0.7	1.0	0.9	39	5	8	19	3	4
183	237	7	28	1	1	155	4	4	0.2	0.3	0.3	0	0	0	90	2	2
192	68	7	21	3	3	32	3	3	0.7	1.0	1.0	15	1	1	2	1	1
197	318	16	186	3	3	81	8	11	2.3	0.4	0.3	2	2	2	30	3	4
209	373	21	116	4	6	176	11	12	0.7	0.4	0.5	42	4	4	27	5	6
227	153	10	35	1	1	96	7	9	0.4	0.1	0.1	1	1	1	63	4	5
228	115	14	2	1	1	107	11	13	0.0	0.1	0.1	5	1	1	73	6	8
231	342	15	14	1	1	310	10	13	0.0	0.1	0.1	0	0	0	171	7	9
234	46	8	0	0	0	36	7	9	0.0	0.0	0.0	0	0	0	20	5	7
245	64	7	37	3	4	11	2	2	3.4	1.5	2.0	1	1	1	4	1	1
248	342	16	65	3	4	66	6	7	1.0	0.5	0.6	5	2	2	18	2	3
253	208	14	77	3	4	19	5	8	4.1	0.6	0.5	2	2	2	2	2	2
318	131	14	0	0	0	80	11	17	0.0	0.0	0.0	9	1	1	62	7	12
339	119	13	7	2	3	77	7	7	0.1	0.3	0.4	11	2	2	46	3	3
342	66	11	0	0	0	53	8	11	0.0	0.0	0.0	0	0	0	32	5	8
352	225	17	104	4	4	44	7	8	2.4	0.6	0.5	69	3	3	19	3	4
356	379	19	225	5	5	107	9	10	2.1	0.6	0.5	104	5	7	40	4	4
359	337	13	122	4	4	125	6	6	1.0	0.7	0.7	39	3	3	62	2	2
360	416	18	265	3	3	77	9	12	3.4	0.3	0.3	29	2	3	17	5	7
362	151	6	75	2	2	49	2	2	1.5	1.0	1.0	0	0	0	47	1	1
387	490	14	129	4	4	163	6	7	0.8	0.7	0.6	132	2	2	49	3	4
394	156	12	115	4	5	11	4	5	10.5	1.0	1.0	129	4	4	1	1	1
419	397	9	186	3	3	54	4	4	3.4	0.8	0.8	31	1	1	31	3	3
420	424	20	162	5	6	148	9	11	1.1	0.6	0.5	57	4	6	92	4	5
439	181	6	0	0	0	154	4	5	0.0	0.0	0.0	0	0	0	5	2	3

Appendix C.3. Continued.

	Total		Endemic (7)			Cosmopolitan (23)			Endemic:Cosmopolitan			Sensitive (8)			Len Tolerant (13)		
ID	Abund	Rich	Abund	Rich	Est Rich	Abund	Rich	Est Rich	Abund	Rich	Est Rich	Abund	Rich	Est Rich	Abund	Rich	Est Rich
441	73	8	5	1	1	62	6	6	0.1	0.2	0.2	0	0	0	35	3	3
451	112	14	14	4	4	43	6	7	0.3	0.7	0.6	18	3	4	28	3	3
452	114	16	10	3	5	95	9	10	0.1	0.3	0.5	9	1	1	50	5	6
453	176	11	0	0	0	153	9	9	0.0	0.0	0.0	0	0	0	131	7	7
454	212	11	9	2	2	112	6	6	0.1	0.3	0.3	0	0	0	51	3	3
455	213	15	137	5	6	26	6	6	5.3	0.8	1.0	44	5	6	7	1	1
456	242	12	121	3	3	55	6	9	2.2	0.5	0.3	1	1	1	2	2	2
457	97	10	0	0	0	74	9	12	0.0	0.0	0.0	5	1	1	33	6	9
458	139	16	1	1	1	125	11	13	0.0	0.1	0.1	17	1	1	62	6	7
459	58	10	0	0	0	45	6	6	0.0	0.0	0.0	1	1	1	18	3	3
472	270	16	0	0	0	127	10	11	0.0	0.0	0.0	7	1	1	51	5	5
538	196	18	50	2	3	85	7	8	0.6	0.3	0.4	28	2	3	30	3	4
557	257	12	0	0	0	30	6	7	0.0	0.0	0.0	1	1	1	14	2	2
559	1115	22	52	3	3	166	10	11	0.3	0.3	0.3	84	6	6	39	4	5
565	195	17	3	3	3	37	7	8	0.1	0.4	0.4	20	4	6	4	2	3
612	232	10	19	2	2	196	6	6	0.1	0.3	0.3	0	0	0	79	4	4
623	648	15	37	1	1	534	8	8	0.1	0.1	0.1	0	0	3	295	5	5
624	481	18	7	1	1	362	11	13	0.0	0.1	0.1	70	2		114	5	6
627	70	9	12	1	1	33	5	5	0.4	0.2	0.2	0	0	0	16	2	2
629	315	16	19	2	2	208	9	11	0.1	0.2	0.2	19	3	4	113	4	5
646	114	13	15	3	4	45	4	5	0.3	0.8	0.8	5	2	3	36	3	4
650	116	10	76	6	7	13	2	2	5.8	3.0	3.5	53	3	4	0	0	0
685	364	18	13	3	4	342	13	18	0.0	0.2	0.2	1	1	1	247	10	4
686	342	10	0	0	0	341	9	10	0.0	0.0	0.0	0	0	0	341	9	10
692	175	15	27	3	3	103	7	8	0.3	0.4	0.4	14	3	4	79	3	3
696	133	12	53	4	4	62	6	6	0.9	0.7	0.7	17	2	2	22	2	2
697	113	12	10	1	1	73	7	9	0.1	0.1	0.1	0	0	0	61	5	7
698	245	12	113	2	3	86	5	5	1.3	0.4	0.6	0	0	0	25	3	3
699	130	15	54	3	3	57	7	8	0.9	0.4	0.4	0	0	0	34	3	3
700	131	12	27	2	2	64	6	6	0.4	0.3	0.3	0	0	0	33	3	3
701	83	11	8	2	2	63	6	7	0.1	0.3	0.3	0	0	0	21	3	4
Mean	58	58	58	2	3	110	7	8	1.1	1.1	0.4	20	2	2	54	3	4
Min	0	0	0	0	0	11	2	2	0.0	0.0	0.0	0	0	0	0	0	0
Max	333	333	333	6	7	534	13	18	10.5	10.5	3.5	132	6	8	341	10	12

Appendix C.4. Fish catch for each site (site number across top).

Scientific name	Common name	19	49	68	73	87	95	107	135	140	145	159	183	192	197
Petromyzontidae															
Ichthyomyzon sp. cf. gagei	southern brook lamprey	0	0	0	0	2	0	0	0	0	0	0	0	0	10
Ichthyomyzon sp.	lamprey (unknown)	0	4	0	0	2	0	11	0	0	0	0	0	0	10
Cyprinidae															
Campostoma oligolepis	largescale stoneroller	13	23	16	11	61	69	11	18	52	10	80	38	15	23
Cyprinella callistia	Alabama shiner	16	0	0	0	4	0	1	0	0	0	1	0	0	0
Cyprinella trichroistia	tricolor shiner	34	0	0	0	0	17	0	0	0	0	1	0	0	0
Cyprinella venusta	blacktail shiner	0	0	0	0	0	0	0	0	0	29	0	0	0	0
Luxilus zonistius	bandfin chub	0	0	0	1	0	0	0	0	0	0	1	0	15	0
Nocomis leptocephalus	bluehead chub	0	9	0	0	3	0	17	1	0	0	1	0	0	0
Notemigonus chrysoleucas	golden shiner	0	0	0	0	0	0	1	0	0	0	0	0	0	0
Notropis chrosomus	rainbow shiner	0	0	0	0	0	21	2	0	0	0	1	0	15	0
Notropis lutipinnis	yellowfin shiner	0	40	0	32	15	0	31	0	20	0	0	0	0	0
Notropis stilbius	silverstripe shiner	0	0	0	0	1	0	1	0	0	0	0	0	0	0
Notropis xaenocephalus	Coosa shiner	21	11	0	0	183	146	0	24	12	2	1	0	0	45
Semotilus atromaculatus	creek chub	0	6	34	31	13	87	5	7	24	3	4	27	15	8
Catostomidae															
Hypentelium etowanum	Alabama hog sucker	13	27	31	12	44	62	5	22	0	30	73	48	0	18
Minytrema melanops	spotted sucker	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Moxostoma duquesnei	black redhorse	0	0	0	0	0	0	9	2	0	0	0	0	0	1
Moxostoma erythrurum	golden redhorse	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ictaluridae															
Ameiurus brunneus	snail bullhead	0	0	4	2	0	0	0	10	0	0	1	0	0	0
Ameiurus natalis	yellow bullhead	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ameiurus nebulosus	brown bullhead	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Noturus leptacanthus	speckled madtom	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Fundulidae															
Fundulus stellifer	southern studfish	0	0	3	0	14	0	10	4	2	0	0	6	0	0
Poecilidae															
Gambusia affinis	eastern mosquitofish	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Gambusia holbrooki	western mosquitofish	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Gambusia sp.	mosquitofish (unknown)	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cottidae															
Cottus carolinae zopherus	Coosa banded sculpin	13	112	0	0	9	4	16	0	50	0	25	28	2	123
Centrarchidae															
Lepomis auritus	redbreast sunfish	4	0	22	2	9	58	1	61	9	33	2	84	0	8
Lepomis cyanellus	green sunfish	0	0	0	1	1	0	4	0	0	0	16	0	2	1
Lepomis gulosus	warmouth	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lepomis macrochirus	bluegill sunfish	1	0	21	43	57	0	6	62	1	66	1	6	0	21
Lepomis microlophus	redear sunfish	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Micropterus coosae	Coosa bass	3	0	21	0	23	14	0	4	2	3	4	0	0	3
Micropterus punctulatus	spotted bass	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Micropterus salmoides	largemouth bass	0	0	0	2	2	2	1	1	0	2	0	0	0	0
Percidae															
Etheostoma etowahae	Etowah darter	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Etheostoma scotti	Cherokee darter	3	0	38	0	8	145	16	18	21	0	1	0	4	18
Etheostoma stigmaeum	speckled darter	0	0	0	18	3	0	15	0	3	0	0	0	0	1
Percina kathae	Mobile logperch	15	2	0	0	0	0	0	0	3	0	0	0	0	10
Percina nigrofasciata	blackbanded darter	9	22	0	7	8	2	14	1	1	18	10	0	0	18
Percina palmaris	bronze darter	0	1	0	0	0	0	0	0	0	0	35	0	0	0
Total no. ind.		145	257	190	162	462	627	177	235	200	196	258	237	68	318
Total no. taxa		12	11	9	12	20	12	20	14	13	10	18	7	7	16

Appendix C.4. Continued.

	209	227	228	231	234	245	248	253	318	339	342	352	356	359	360	362	387	394
Ichthyomyzon sp. cf. gagei	0	0	0	0	0	0	2	0	0	0	0	1	1	0	2	0	0	0
Ichthyomyzon sp.	0	0	0	0	0	0	2	1	0	0	3	1	1	0	2	0	0	0
Campostoma oligolepis	19	30	13	116	3	0	15	10	1	23	12	6	39	17	30	0	53	5
Cyprinella callistia	0	0	0	0	0	0	0	1	0	9	0	28	1	33	28	0	120	17
Cyprinella trichroistia	2	0	0	0	0	0	0	0	0	0	0	38	93	0	0	0	0	83
Cyprinella venusta	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Luxilus zonistius	0	0	0	0	0	0	64	0	0	0	0	0	0	0	0	0	0	0
Nocomis leptocephalus	0	0	0	0	0	0	43	61	10	0	0	0	0	0	0	0	0	0
Notemigonus chrysoleucas	0	0	1	0	0	0	0	0	1	0	0	0	0	0	2	0	0	0
Notropis chrosomus	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	12	0
Notropis lutipinnis	0	0	0	0	0	0	76	1	16	0	0	0	0	0	0	0	0	0
Notropis stilbius	6	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3
Notropis xaenocephalus	1	0	0	0	0	0	0	0	0	6	0	43	35	86	24	0	24	5
Semotilus atromaculatus	0	0	5	11	0	7	24	1	3	0	7	3	12	13	23	2	53	0
Hypentelium etowanum	31	17	5	8	10	14	22	38	25	2	2	17	41	26	37	14	70	9
Minytrema melanops	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0
Moxostoma duquesnei	4	1	0	0	0	0	0	0	0	2	0	0	1	4	0	0	0	0
Moxostoma erythrurum	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ameiurus brunneus	0	0	0	0	0	0	0	0	9	0	0	0	0	0	0	0	0	3
Ameiurus natalis	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ameiurus nebulosus	0	0	0	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Noturus leptacanthus	5	0	0	0	0	0	0	0	0	0	0	3	6	0	1	0	0	0
Fundulus stellifer	1	5	0	1	0	0	0	0	0	17	8	24	0	0	0	0	0	0
Gambusia affinis	0	0	0	1	1	0	0	0	0	0	1	0	0	0	0	0	0	0
Gambusia holbrooki	2	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0
Gambusia sp.	2	0	0	1	1	0	0	0	1	0	1	0	0	0	0	0	0	0
Cottus carolinae zopherus	112	0	0	0	0	34	62	67	0	0	0	16	49	2	195	49	32	0
Lepomis auritus	10	14	3	32	2	0	0	1	25	20	25	13	13	32	9	0	13	0
Lepomis cyanellus	1	29	23	107	10	0	0	0	1	0	1	1	23	0	1	0	0	0
Lepomis gulosus	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0
Lepomis macrochirus	12	19	40	27	6	4	17	1	32	24	4	5	2	30	4	47	35	1
Lepomis microlophus	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Micropterus coosae	3	0	0	0	0	2	2	10	0	7	0	6	3	31	4	13	5	1
Micropterus punctulatus	0	0	3	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Micropterus salmoides	0	0	5	0	0	0	1	0	0	2	0	0	2	0	1	0	1	0
Etheostoma etowahae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Etheostoma scotti	1	35	2	14	0	2	1	9	0	1	0	7	45	32	46	26	61	0
Etheostoma stigmaeum	27	0	5	0	0	0	3	0	0	0	0	0	0	0	0	0	0	0
Percina kathae	39	0	1	6	0	0	0	0	0	0	0	0	0	0	1	0	3	0
Percina nigrofasciata	90	2	8	14	13	0	6	6	5	4	2	13	9	29	6	0	8	2
Percina palmaris	0	0	0	0	0	1	2	1	0	0	0	0	3	0	0	0	0	26
	373	153	115	342	46	64	342	208	131	119	66	225	379	337	416	151	490	156
	21	10	14	15	8	7	16	14	14	13	11	17	19	13	18	6	14	12

Appendix C.4. Continued.

	419	420	439	441	451	452	453	454	455	456	457	458	459	472	538	557	559	565
Ichthyomyzon sp. cf. gagei	0	0	0	0	0	0	0	0	0	0	0	4	4	2	6	26	21	7
Ichthyomyzon sp.	0	0	0	0	0	1	0	0	1	0	0	4	4	2	6	26	21	7
Campostoma oligolepis	0	10	140	14	7	6	13	38	2	18	0	27	5	27	12	2	40	3
Cyprinella callistia	0	53	0	0	15	0	0	0	5	0	0	0	1	0	1	1	30	1
Cyprinella trichroistia	0	1	0	0	2	0	0	0	34	0	0	0	0	0	0	0	9	0
Cyprinella venusta	0	0	0	0	0	0	0	0	0	0	0	4	0	0	10	0	3	0
Luxilus zonistius	0	0	0	0	0	0	0	0	0	0	0	0	0	0	12	0	26	0
Nocomis leptocephalus	0	0	0	0	0	0	0	0	0	11	0	0	0	30	8	61	220	25
Notemigonus chrysoleucas	0	0	0	0	0	0	0	2	0	0	0	6	0	4	0	0	0	0
Notropis chrosomus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Notropis lutipinnis	0	0	0	0	0	0	0	0	0	12	0	0	0	91	5	89	415	95
Notropis stilbius	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0
Notropis xaenocephalus	0	3	0	0	6	8	0	0	65	54	0	0	0	0	0	0	0	1
Semotilus atromaculatus	23	28	9	10	0	18	0	20	9	10	14	5	8	15	6	2	25	2
Hypentelium etowanum	154	51	21	6	8	4	22	17	20	43	23	1	4	5	7	24	159	18
Minytrema melanops	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0
Moxostoma duquesnei	0	0	0	0	0	9	0	0	2	0	0	0	0	0	0	0	0	0
Moxostoma erythrurum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	6	0
Ameiurus brunneus	0	2	0	0	0	0	0	0	0	0	5	0	0	0	0	0	2	4
Ameiurus natalis	0	0	0	0	0	0	2	0	0	0	1	0	0	0	0	0	0	0
Ameiurus nebulosus	0	1	0	0	0	0	2	0	0	0	1	0	0	0	0	0	0	0
Noturus leptacanthus	0	1	0	0	1	0	0	0	2	1	0	0	0	0	0	0	0	0
Fundulus stellifer	0	1	0	0	23	1	1	72	0	0	0	0	0	13	0	0	0	0
Gambusia affinis	0	0	0	0	0	0	39	0	0	0	0	0	0	0	0	0	0	0
Gambusia holbrooki	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Gambusia sp.	0	0	0	0	0	0	39	0	0	0	0	0	0	0	0	0	0	0
Cottus carolinae zopherus	146	128	0	5	4	1	0	2	28	11	0	0	0	0	49	0	35	1
Lepomis auritus	12	67	1	19	20	14	32	23	7	0	5	17	7	4	1	0	1	0
Lepomis cyanellus	0	17	4	0	5	7	0	0	0	1	0	21	9	10	2	2	5	3
Lepomis gulosus	3	0	0	0	0	4	0	0	0	0	1	2	0	2	0	0	0	0
Lepomis macrochirus	16	7	0	14	3	24	3	26	0	1	20	15	2	31	27	12	31	1
Lepomis microlophus	0	0	0	0	0	1	0	0	0	0	0	1	0	0	0	0	0	0
Micropterus coosae	3	5	6	0	9	3	0	2	24	0	0	0	0	0	0	0	0	2
Micropterus punctulatus	0	0	0	0	0	0	0	0	0	0	0	13	14	1	0	0	0	0
Micropterus salmoides	0	0	0	2	0	0	14	0	0	0	5	0	0	0	0	0	2	0
Etheostoma etowahae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Etheostoma scotti	9	26	0	0	2	1	0	7	9	56	0	1	0	0	1	0	0	0
Etheostoma stigmaeum	0	0	0	0	0	0	0	0	0	0	0	17	0	7	27	0	29	14
Percina kathae	0	3	0	0	0	0	0	0	0	0	0	0	0	0	6	0	0	0
Percina nigrofasciata	0	15	0	3	7	12	9	3	4	24	22	1	0	26	10	11	25	10
Percina palmaris	31	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	8	1
	397	420	181	73	112	114	176	212	213	242	97	139	58	270	196	257	1115	195
	9	19	6	8	14	16	11	11	15	12	10	16	10	16	18	12	22	17

Appendix C.4. Continued.

	612	623	624	627	629	646	650	685	686	692	696	697	698	699	700	701
Ichthyomyzon sp. cf. gagei	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ichthyomyzon sp.	0	7	2	0	0	1	0	0	1	0	0	0	0	2	0	0
Campostoma oligolepis	100	209	142	6	76	0	10	90	0	13	20	10	45	12	5	2
Cyprinella callistia	0	0	0	0	9	4	14	0	0	11	0	0	0	0	0	0
Cyprinella trichroistia	0	0	0	0	0	0	38	0	0	0	0	0	0	0	0	0
Cyprinella venusta	2	10	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Luxilus zonistius	0	0	0	0	0	0	0	0	0	0	0	4	7	0	0	0
Nocomis leptocephalus	0	0	0	0	0	0	0	0	0	0	0	1	4	3	0	0
Notemigonus chrysoleucas	0	0	0	0	0	0	0	0	47	0	0	0	0	0	0	0
Notropis chrosomus	0	0	0	0	0	0	0	0	0	0	14	0	0	0	0	0
Notropis lutipinnis	0	0	0	0	0	20	0	0	0	0	0	0	0	0	2	0
Notropis stilbius	0	0	2	0	1	0	0	0	0	0	0	0	0	0	0	0
Notropis xaenocephalus	0	0	0	12	14	0	22	9	0	3	10	10	112	32	22	4
Semotilus atromaculatus	17	11	18	7	3	0	0	0	0	0	10	0	0	7	11	35
Hypentelium etowanum	15	37	73	10	9	10	13	7	0	11	16	11	16	6	27	6
Minytrema melanops	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Moxostoma duquesnei	0	0	0	0	9	0	0	0	0	0	3	0	0	0	0	0
Moxostoma erythrurum	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0
Ameiurus brunneus	0	0	1	0	0	0	0	1	0	0	0	0	0	0	0	0
Ameiurus natalis	0	0	0	0	0	0	0	3	5	0	0	0	0	0	0	0
Ameiurus nebulosus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Noturus leptacanthus	0	0	69	0	0	0	0	0	0	0	0	0	0	0	0	0
Fundulus stellifer	0	18	31	12	64	16	0	0	0	16	0	14	12	4	6	5
Gambusia affinis	0	0	0	0	0	0	0	1	17	0	0	1	0	0	0	0
Gambusia holbrooki	0	0	0	0	0	0	0	42	0	0	0	0	0	0	0	0
Gambusia sp.	0	0	0	0	0	0	0	42	17	0	0	1	0	0	0	0
Cottus carolinae zopherus	8	0	7	0	0	7	7	1	0	18	16	0	1	18	5	4
Lepomis auritus	24	42	17	14	45	8	0	16	85	25	13	23	15	26	21	8
Lepomis cyanellus	49	107	1	2	2	0	0	16	0	3	0	7	8	4	2	0
Lepomis gulosus	0	0	0	0	0	0	0	1	22	0	0	0	0	0	0	0
Lepomis macrochirus	3	132	91	0	65	27	0	124	142	51	9	29	2	4	10	12
Lepomis microlophus	0	6	2	0	0	0	0	1	5	0	0	0	0	0	0	1
Micropterus coosae	0	3	1	3	5	3	0	2	0	1	2	0	7	4	5	1
Micropterus punctulatus	0	19	3	0	0	0	0	0	0	0	0	0	0	1	0	0
Micropterus salmoides	3	8	3	0	1	1	0	1	1	0	0	0	0	0	0	0
Etheostoma etowahae	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0
Etheostoma scotti	11	37	0	0	5	7	7	3	0	6	13	0	0	4	0	0
Etheostoma stigmaeum	0	0	0	0	1	0	0	0	0	2	0	0	0	0	0	0
Percina kathae	0	2	3	0	0	0	0	0	0	6	0	0	0	0	0	0
Percina nigrofasciata	0	0	15	4	6	9	3	4	0	8	7	2	16	3	15	5
Percina palmaris	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0
	232	648	481	70	315	114	116	364	342	175	133	113	245	130	131	83
	10	15	18	9	16	13	10	18	10	15	12	12	12	15	12	11

APPENDIX D

SUPPLEMENTAL MATERIAL TO CHAPTER 4

Appendix D.1. Site information used to calculate hydrologic data. Discharge for 0.5-yr RI flood was extrapolated from formula in Stamey and Hess (1993). Hydraulic depths were calculated using HEC-RAS®, and adjusted with the bankfull depth.

Site No.	Stream name	Manning's n	Bankfull Depth (mm)	Discharge 0.5-yr RI flood (ft^3 s^{-1})	Mean Daily Stage (year)	Daily Stage (year)	50% Adj Hydr Depth (mm); Q0.5	75% Adj Hydr Depth (mm); Q0.5	Adj Hydr Depth (mm); Q0.5
11	West Fork Pumpkinvine	0.03	2134	9.9	153	102	447	620	792
18	Town Creek	0.045	1402	10.6	161	102	367	500	632
24	Polecat Branch	0.04	1768	9.0	180	71	451	641	831
26	Murphy Creek	0.042	1676	7.5	286	223	578	756	933
27	Hickory Log Creek	0.04	1585	8.2	147	80	620	890	1160
31	Gorman Branch	0.038	1951	9.0	347	237	577	747	917
33	Boston Creek	0.03	2134	9.7	99	47	637	932	1227
38	Black Mill Creek	0.038	2713	11.0	159	62	432	617	802
44	Board Tree Creek	0.035	2774	11.6	314	224	714	959	1204
45	Upper Smithwick Creek	0.033	2134	10.8	351	217	587	772	957
46	Scott Mill Creek	0.045	1981	8.9	218	166	501	669	836
48	Toonigh Creek	0.033	2560	10.3	126	0	465	698	930
52	Copper Sandy Creek	0.033	2652	10.7	473	294	599	752	904
54	Trib. Sweat Mountain	0.038	2134	7.0	476	381	726	899	1071
55	East Fork Rubes Creek	0.033	1920	8.5	269	94	614	874	1134
56	West Fork Rubes Creek	0.03	2591	8.5	273	125	250	312	375
57	Lower Noonday Creek	0.03	2195	11.4	285	152	722	1007	1292
58	Upper Noonday Creek	0.03	1676	7.7	274	211	451	571	691
60	Clark Creek	0.045	1676	11.2	364	51	406	584	761
61	Proctor Creek	0.033	1920	12.6	144	93	488	685	883
62	Butler Creek	0.042	2286	10.2	99	0	420	630	840
63	Upper Allatoona Creek	0.042	2347	12.2	106	14	479	712	944
65	Picketts Mill	0.035	2438	9.0	293	193	503	658	813
66	Possum Creek	0.035	2438	11.1	159	86	536	761	986
67	Westbrook Creek	0.033	3109	8.5	158	50	435	628	820
71	Lawrence Creek	0.033	1585	8.6	75	0	410	615	820
72	Lane Creek	0.03	2042	8.0	382	242	657	865	1072
75	Ward Creek	0.038	1707	11.4	147	33	303	438	573
76	Upper Little Pumpkinvine	0.038	2530	11.2	159	44	644	944	1244
80	Buzzard Flapper	0.04	1676	7.8	331	185	620	838	1055
Mean		0.036	2124	9.7	234	126	521	719	917
Minimum		0.030	1402	7.0	75	0	250	312	375
Maximum		0.045	3109	12.6	476	381	726	1007	1292

Appendix D.2. Baseflow data for 7 January 2003 through 14 April 2003 (spring I).

Site No.	Minimum Daily Stage	Min 7-Day Mean/ Mean	Min 7-Day Max/ Mean	MagLow <25/ Mean	MagLow <10/ Mean	MagLow <5/ Mean	DurLow <25	DurLow <10	DurLow <5
11									
18									
24	118	0.57	0.74	1.03	0.89	0.77	209	183	101
26									
27	80	0.71	0.74	0.68	0.65	0.63	224	66	31
31	283	0.71	0.74	0.97	0.87	0.85	317	207	45
33	60	0.78	0.79	0.68	0.64	0.63	1003	211	27
38	104	0.55	0.58	0.83	0.73	0.71	321	110	42
44									
45	217	0.63	0.66	0.71	0.68	0.67	228	60	39
46	167	0.82	0.83	0.82	0.81	0.80	192	50	41
48									
52	295	0.76	0.79	0.73	0.70	0.68	328	94	36
54									
55	177	0.72	0.75	0.75	0.72	0.70	214	143	50
56									
57	212	0.80	0.81	0.83	0.81	0.79	182	62	50
58									
60									
61									
62	70	0.42	0.50	1.21	0.92	0.89	321	75	59
63									
65									
66									
67	60	0.61	0.66	0.51	0.47	0.45	215	61	42
71									
72	323	0.82	0.83	0.91	0.88	0.87	274	96	63
75									
76									
80	216	0.82	0.87	0.77	0.74	0.72	170	87	32

Appendix D.3. Baseflow data for 14 April 2003 through 14 May 2003 (spring II).

Site No.	Minimum Daily Stage	Min 7-Day Mean/ Mean	Min 7-Day Max/ Mean	MagLow <25/ Mean	MagLow <10/ Mean	MagLow <5/ Mean	DurLow <25	DurLow <10	DurLow <5
11									
18	114	0.72	0.73	0.76	0.75	0.74	84	23	22
24	124	0.68	0.82	0.79	0.74	0.72	72	33	24
26									
27	95	0.67	0.70	0.68	0.67	0.66	71	29	29
31	366	0.84	0.85	1.09	1.08	1.07	69	31	16
33	70	0.69	0.69	0.71	0.71	0.71	336	336	336
38	133	0.63	0.65	0.90	0.88	0.86	72	53	25
44									
45	348	0.82	0.83	1.03	1.01	1.00	85	63	32
46	174	0.74	0.75	0.84	0.82	0.81	72	76	22
48									
52									
54									
55									
56									
57	190	0.67	0.73	0.81	0.74	0.68	83	47	33
58									
60									
61									
62									
63									
65									
66									
67	90	0.61	0.67	0.61	0.58	0.57	57	24	17
71	64	0.61	0.87	1.17	1.01	0.94	60	24	17
72	312	0.79	0.84	0.86	0.84	0.83	59	47	18
75									
76									
80									

Appendix D.4. Baseflow data for 15 May 2003 through 7 August 2003 (summer). Baseflow principle components axes (PCAs) calculated from 9 baseflow variables.

Site No.	Minimum Daily Stage	Min 7-Day Mean/ Mean	Min 7-Day Max/ Mean	MagLow <25/ Mean	MagLow <10/ Mean	MagLow <5/ Mean	DurLow <25	DurLow <10	DurLow <5	Baseflow PCA 1	Baseflow PCA 2
11											
18	114	0.70	0.73	0.81	0.79	0.77	181	124	57	-0.07	0.44
24	88	0.68	0.69	0.56	0.54	0.53	233	57	36	-1.99	-0.48
26	255	0.89	0.91	0.93	0.91	0.91	241	140	78	2.00	2.08
27	106	0.75	0.78	1.00	0.97	0.95	228	82	53	1.43	0.16
31	281	0.90	0.91	0.83	0.82	0.81	113	50	33	1.63	-1.73
33	48	0.56	0.63	0.77	0.69	0.65	118	55	39	-1.41	-1.82
38	78	0.47	0.51	0.69	0.61	0.56	130	84	67	-2.64	-0.45
44											
45	286	0.79	0.81	0.90	0.86	0.84	230	176	47	1.57	1.80
46	184	0.76	0.78	0.89	0.87	0.86	172	43	41	0.99	-1.17
48											
52	373	0.69	0.75	0.93	0.86	0.82	193	151	100	0.94	2.53
54	432	0.84	0.85	0.95	0.93	0.92	175	140	24	2.84	0.33
55	221	0.64	0.73	0.98	0.92	0.89	106	66	41	1.17	-1.49
56											
57	152	0.61	0.67	0.68	0.64	0.61	169	130	58	-1.37	0.55
58	218	0.78	0.80	0.85	0.84	0.83	112	52	45	0.94	-1.53
60											
61											
62											
63											
65											
66											
67											
71	9	0.15	0.30	0.38	0.28	0.20	207	94	92	-6.70	1.18
72	299	0.76	0.78	0.87	0.84	0.82	232	66	50	1.10	0.29
75											
76	111	0.55	0.63	0.88	0.82	0.78	197	54	46	-0.43	-0.70
80											

Appendix D.5. Baseflow data for 15 August 2003 through 4 November 2003 (autumn). Baseflow principle components axes (PCAs) calculated from 9 baseflow variables.

Site No.	Minimum Daily Stage	Min 7-Day Mean/ Mean	Min 7-Day Max/ Mean	MagLow <25/ Mean	MagLow <10/ Mean	MagLow <5/ Mean	DurLow <25	DurLow <10	DurLow <5	Baseflow PCA 1	Baseflow PCA 2
11	102	0.71	0.76	0.81	0.77	0.73	78	44	34	1.09	-1.70
18	110	0.88	0.92	0.77	0.75	0.74	210	70	44	1.11	0.24
24	71	0.60	0.74	0.55	0.50	0.45	132	70	50	-0.72	-1.21
26	262	0.94	0.95	0.95	0.93	0.93	116	48	29	2.80	-0.33
27	102	0.88	0.91	0.79	0.76	0.74	127	120	93	0.73	-0.07
31											
33	63	0.77	0.80	0.73	0.69	0.68	70	42	31	0.83	-1.96
38	66	0.65	0.69	0.52	0.47	0.44	215	111	31	-1.09	-0.10
44	224	0.91	0.93	0.74	0.73	0.73	160	63	57	1.42	0.11
45	248	0.90	0.92	0.73	0.72	0.72	130	54	28	1.64	-0.39
46	176	0.90	0.92	0.85	0.84	0.83	137	56	44	1.88	-0.37
48											
52	371	0.85	0.91	0.86	0.82	0.81	170	130	41	1.83	1.19
54	405	0.93	0.93	0.87	0.86	0.86	285	55	49	2.40	2.15
55	94	0.55	0.57	0.40	0.37	0.37	304	86	28	-1.77	0.59
56											
57	204	0.75	0.75	0.73	0.72	0.72	183	53	34	0.95	-0.12
58	213	0.86	0.89	0.84	0.81	0.79	283	143	98	0.81	2.28
60											
61											
62	0	0.00	0.00	0.00	0.00	0.00	188	188	188	-6.55	-0.35
63											
65	193	0.72	0.74	0.72	0.69	0.68	204	52	32	0.75	0.02
66	96	0.79	0.80	0.64	0.62	0.62	197	43	43	0.35	-0.39
67	55	0.72	0.72	0.35	0.35	0.35	242	242	242	-3.41	2.26
71	0	0.00	0.00	0.00	0.00	0.00	186	186	186	-6.53	-0.40
72	242	0.87	0.89	0.71	0.69	0.67	125	62	48	1.25	-0.36
75											
76	86	0.72	0.77	0.66	0.62	0.60	140	57	31	0.22	-1.09
80											

Appendix D.6. Baseflow data for 4 November 2003 through 28 January 2004 (winter).

Site No.	Minimum Daily Stage	Min 7-Day Mean/ Mean	Min 7-Day Max/ Mean	MagLow <25/ Mean	MagLow <10/ Mean	MagLow <5/ Mean	DurLow <25	DurLow <10	DurLow <5
11									
18	130	0.81	0.84	0.88	0.86	0.85	171	89	40
24									
26	223	0.83	0.86	0.86	0.83	0.82	255	58	28
27	89	0.67	0.70	0.74	0.68	0.66	231	167	50
31	237	0.89	0.92	0.73	0.72	0.70	86	51	33
33	47	0.49	0.53	0.61	0.57	0.54	132	48	32
38	62	0.55	0.62	0.48	0.43	0.42	186	122	51
44	310	0.86	0.87	1.06	1.04	1.03	235	165	41
45									
46	166	0.84	0.87	0.80	0.79	0.78	77	44	20
48									
52	309	0.73	0.79	0.73	0.71	0.70	139	58	29
54									
55	104	0.54	0.70	0.60	0.53	0.46	191	94	87
56									
57	197	0.75	0.78	0.73	0.71	0.70	206	68	58
58	211	0.83	0.86	0.86	0.84	0.83	153	67	24
60									
61									
62	23	0.31	0.32	0.32	0.29	0.27	109	67	43
63									
65	225	0.73	0.79	0.84	0.81	0.80	216	56	32
66	86	0.50	0.55	0.65	0.60	0.58	168	118	61
67	50	0.24	0.26	0.47	0.38	0.35	285	149	43
71	0	0.00	0.00	0.00	0.00	0.00	243	243	243
72	310	0.82	0.84	0.88	0.85	0.84	228	58	37
75									
76	44	0.41	0.50	0.42	0.36	0.33	177	95	33
80	205	0.67	0.72	0.77	0.72	0.69	149	79	49

Appendix D.7. Stormflow data for 7 January 2003 through 14 April 2003 (spring I).

Site No.	Freq Q>0.5	75% Freq Q>0.5	50% Freq Q>0.5	Mag Q>0.5/ Mean	75% Mag Q>0.5/ Mean	50% Mag Q>0.5/ Mean	Dur Q>0.5	75% Dur Q>0.5	50% Dur Q>0.5	Area Q>0.5	75% Area Q>0.5	50% Area Q>0.5	RateRise 5cm	RateRise 10cm	RateRise 20cm	RateFall 5cm	RateFall 10cm	RateFall 20cm
11																		
18																		
24	1	1	1	5.7	5.2	3.6	2	3	9	380	894	1738	2	2	2	3	2	1
26																		
27	1	2	2	10.1	8.1	7.6	4	10	12	1281	3048	5976	13	10	6	11	7	4
31	2	3	4	5.0	4.5	3.1	19	23	51	15467	18945	24794	15	11	7	11	9	5
33	1	1	1	0.0	11.1	9.9	0	5	8	0	825	2737	11	7	3	10	3	0
38	3	4	5	7.9	6.5	4.6	18	29	66	8143	12253	20205	23	16	9	22	9	2
44																		
45	3	3	5	4.4	3.5	2.7	19	34	66	10939	15936	24740	20	13	10	18	8	2
46	1	2	2	4.9	4.3	3.8	5	9	13	1163	2324	4149	16	8	4	10	5	2
48																		
52	1	2	4	2.3	1.8	1.5	7	35	172	1413	3571	17589	17	5	3	8	2	1
54																		
55	1	2	3	6.0	5.3	4.3	13	18	29	6297	9942	15586	29	13	2	14	6	2
56																		
57	2	3	7	4.3	3.9	2.9	18	24	51	7195	10702	16309	25	16	6	19	9	3
58																		
60																		
61																		
62	2	2	2	12.0	11.1	9.0	12	15	23	4188	7092	10774	22	11	4	18	9	5
63																		
65																		
66																		
67	1	2	2	6.9	6.0	5.0	8	13	20	2186	4121	7301	11	8	6	14	7	2
71																		
72	1	1	2	3.2	2.7	2.0	2	6	52	299	1002	4979	8	6	3	4	3	0
75																		
76																		
80	1	2	3	3.9	3.4	2.5	5	8	22	1460	2800	5491	15	9	2	10	4	2

Appendix D.8. Stormflow data for 14 April 2003 through 14 May 2003 (spring II).

Site No.	Freq Q>0.5	75% Freq Q>0.5	50% Freq Q>0.5	Mag Q>0.5/ Mean	75% Mag Q>0.5/ Mean	50% Mag Q>0.5/ Mean	Dur Q>0.5	75% Dur Q>0.5	50% Dur Q>0.5	Area Q>0.5	75% Area Q>0.5	50% Area Q>0.5	RateRise 5cm	RateRise 10cm	RateRise 20cm	RateFall 5cm	RateFall 10cm	RateFall 20cm
11																		
18	0	1	1	0.0	0.0	2.7	0	0	3	0	0	188	2	1	0	3	0	0
24	1	2	2	5.6	4.4	3.3	2	6	29	346	946	4171	8	5	2	6	3	2
26																		
27	1	1	1	10.4	9.0	8.2	2	4	5	739	1706	2880	10	4	2	5	3	2
31	1	1	4	4.9	3.9	2.4	11	18	64	8761	10949	17026	15	8	3	8	6	3
33	1	1	1	12.1	12.0	10.3	1	4	7	10	1013	2641	8	4	2	9	4	0
38	1	2	2	7.6	6.2	4.3	9	16	44	3714	5932	11287	18	8	3	16	7	3
44																		
45	1	3	4	3.5	2.8	2.3	13	37	79	3719	7971	18369	20	14	5	22	7	1
46	1	1	2	5.3	4.8	4.1	3	4	6	956	1490	2312	11	7	2	6	2	2
48																		
52																		
54																		
55																		
56																		
57	3	3	4	4.2	3.6	2.8	27	43	83	9807	15734	26219	21	16	10	29	19	5
58																		
60																		
61																		
62																		
63																		
65																		
66																		
67	1	1	2	6.6	6.1	4.4	6	8	20	1385	2761	5198	18	10	5	13	4	4
71	1	1	1	11.3	9.9	7.7	1	4	13	29	521	2183	7	3	2	7	1	0
72	1	1	2	3.3	2.8	1.9	2	6	46	352	1127	4015	10	4	4	7	2	1
75																		
76																		
80																		

Appendix D.9. Stormflow data for 15 May 2003 through 7 August 2003 (summer). Stormflow principle components axes (PCAs) calculated from 18 stormflow variables.

Site No.	Freq Q>0.5	75% Freq Q>0.5	50% Freq Q>0.5	Mag Q>0.5/ Mean	75% Mag Q>0.5/ Mean	50% Mag Q>0.5/ Mean	Dur Q>0.5	75% Dur Q>0.5	50% Dur Q>0.5	Area Q>0.5	75% Area Q>0.5	50% Area Q>0.5	RateRise 5cm	RateRise 10cm	RateRise 20cm	RateFall 5cm	RateFall 10cm	RateFall 20cm	Storm-flow PCA 1	Storm-flow PCA 2
11																				
18	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	5	0	0	-4.70	-1.21
24	3	1	6	0.0	3.8	3.3	0	5	11	0	205	1558	23	9	4	13	6	1	-2.60	-0.66
26	3	3	3	3.7	3.3	2.8	4	8	14	505	1460	3305	10	7	5	11	5	2	-2.52	-0.23
27	2	2	3	11.7	10.2	8.5	6	9	13	3299	5398	8146	19	10	8	13	7	5	-1.36	2.67
31	1	1	4	3.3	3.1	2.5	7	9	18	1670	3028	4979	19	7	2	11	5	1	-2.62	-0.39
33	3	2	3	14.6	12.8	11.0	6	12	18	1573	3900	8044	28	14	7	27	11	4	-0.95	3.43
38	3	2	6	7.8	6.7	5.1	29	43	79	12717	19304	29869	30	20	10	32	20	4	2.55	0.91
44																				
45	5	2	6	3.7	2.9	2.3	9	22	55	3063	5436	11863	39	17	8	26	9	1	-0.38	-0.84
46	4	3	7	4.5	4.0	3.2	5	10	23	746	2058	4499	30	13	8	17	8	2	-1.22	-0.26
48																				
52	7	3	7	2.1	1.7	1.4	6	80	758	597	4167	65446	23	10	2	6	2	1	-0.25	-4.36
54	5	4	9	2.5	2.2	1.9	8	25	68	1128	3821	11120	46	27	8	33	8	1	0.24	-1.44
55	5	5	8	4.9	4.4	3.4	23	40	92	4291	12257	28516	57	43	20	71	32	2	4.12	-0.44
56																				
57	11	9	12	4.1	3.4	2.6	29	52	114	9759	16181	29292	66	43	23	79	31	9	7.18	-0.63
58	11	10	13	3.4	2.9	2.3	19	32	70	4344	7338	13264	57	36	17	58	25	7	4.67	-0.97
60																				
61																				
62																				
63																				
65																				
66																				
67																				
71	6	4	6	15.9	13.0	9.2	14	25	60	5262	9000	17055	38	27	16	41	15	8	2.24	3.33
72	2	1	6	3.8	3.2	2.1	7	13	76	2789	4660	11169	22	7	3	9	3	0	-2.18	-0.81
75																				
76	2	2	3	10.6	9.0	6.7	3	5	11	1310	2444	4631	22	10	3	12	6	3	-2.22	1.89
80																				

Appendix D.10. Stormflow data for 15 August 2003 through 4 November 2003 (autumn). Stormflow principle components axes (PCAs) calculated from 18 stormflow variables.

Site No.	Freq Q>0.5	75% Freq Q>0.5	50% Freq Q>0.5	Mag Q>0.5/ Mean	75% Mag Q>0.5/ Mean	50% Mag Q>0.5/ Mean	Dur Q>0.5	75% Dur Q>0.5	50% Dur Q>0.5	Area Q>0.5	75% Area Q>0.5	50% Area Q>0.5	RateRise 5cm	RateRise 10cm	RateRise 20cm	RateFall 5cm	RateFall 10cm	RateFall 20cm	Storm-flow PCA 1	Storm-flow PCA 2
11	0	1	1	0.0	0.0	3.2	0	0	2	0	0	83	6	2	1	1	1	0	-0.66	1.17
18	0	0	0	0.0	0.0	0.0	0	0	0	0	0	0	1	1	0	2	0	0	-2.40	-0.80
24	0	1	2	0.0	0.0	2.8	0	0	1	0	0	56	6	3	2	4	2	0	0.08	1.90
26	0	0	0	0.0	0.0	0.0	0	0	0	0	0	0	0	0	0	0	0	0	-2.75	-1.16
27	0	0	0	0.0	0.0	0.0	0	0	0	0	0	0	1	0	0	0	0	0	-2.67	-1.06
31																				
33	0	0	0	0.0	0.0	0.0	0	0	0	0	0	0	1	0	0	0	0	0	-2.67	-1.06
38	0	0	2	0.0	0.0	3.3	0	0	5	0	0	451	6	4	1	4	1	0	0.09	1.52
44	0	0	0	0.0	0.0	0.0	0	0	0	0	0	0	2	1	0	0	0	0	-2.48	-0.82
45	0	2	2	0.0	2.3	2.1	0	4	9	0	203	1270	6	4	3	7	2	0	3.31	1.50
46	0	0	1	0.0	0.0	0.0	0	0	0	0	0	0	5	2	0	1	1	0	-1.70	-0.12
48																				
52	0	0	0	0.0	0.0	0.0	0	0	0	0	0	0	3	0	0	1	0	0	-2.45	-0.78
54	1	2	2	2.3	2.1	1.9	1	2	5	15	245	866	6	4	3	7	3	0	4.18	0.72
55	1	2	2	4.2	3.8	3.2	1	3	6	9	440	1519	11	6	3	11	6	0	7.11	1.60
56																				
57	1	2	3	3.6	3.1	2.4	2	5	12	403	1037	2363	8	6	2	9	5	1	10.28	-3.98
58	1	1	2	2.7	2.4	2.1	1	3	6	50	300	733	5	3	1	3	1	0	3.24	-0.87
60																				
61																				
62	0	1	1	0.0	0.0	6.0	0	0	1	0	0	174	7	5	4	7	2	0	1.09	3.62
63																				
65	0	0	3	0.0	0.0	1.8	0	0	1	0	0	23	7	4	0	5	0	0	-0.50	1.29
66	0	0	0	0.0	0.0	0.0	0	0	0	0	0	0	7	2	0	2	0	0	-1.84	-0.05
67	0	0	0	0.0	0.0	0.0	0	0	0	0	0	0	2	0	0	0	0	0	-2.60	-0.95
71	0	0	0	0.0	0.0	0.0	0	0	0	0	0	0	1	0	0	0	0	0	-2.67	-1.06
72	0	0	0	0.0	0.0	0.0	0	0	0	0	0	0	4	1	0	5	1	0	-1.80	-0.24
75																				
76	0	0	0	0.0	0.0	0.0	0	0	0	0	0	0	3	1	1	1	0	0	-2.17	-0.39
80																				

Appendix D.11. Stormflow data for 4 November 2003 through 28 January 2004 (winter).

Site No.	Freq Q>0.5	75% Freq Q>0.5	50% Freq Q>0.5	Mag Q>0.5/ Mean	75% Mag Q>0.5/ Mean	50% Mag Q>0.5/ Mean	Dur Q>0.5	75% Dur Q>0.5	50% Dur Q>0.5	Area Q>0.5	75% Area Q>0.5	50% Area Q>0.5	RateRise 5cm	RateRise 10cm	RateRise 20cm	RateFall 5cm	RateFall 10cm	RateFall 20cm
11																		
18	0	0	1	0.0	0.0	2.5	0	0	5	0	0	173	2	0	0	1	0	0
24																		
26	1	1	1	3.6	3.2	2.8	1	3	6	89	488	1262	6	4	2	5	2	0
27	1	1	1	7.9	7.3	6.3	1	2	4	4	353	1219	10	5	2	6	3	1
31	0	0	1	0.0	0.0	1.9	0	0	5	0	0	339	3	0	0	2	1	1
33	0	0	0	0.0	0.0	0.0	0	0	0	0	0	0	7	0	0	3	0	0
38	0	0	2	0.0	0.0	3.2	0	0	15	0	0	1115	3	0	0	1	1	0
44	0	1	1	0.0	3.2	2.7	0	2	8	0	79	1100	6	2	0	4	1	0
45																		
46	1	1	1	4.7	4.0	3.4	1	3	5	184	606	1200	9	4	2	7	4	1
48																		
52	0	1	2	0.0	1.7	1.4	0	21	176	0	814	12789	4	0	0	2	0	0
54																		
55	1	2	4	4.9	4.2	3.4	7	17	40	1346	4464	11986	34	14	3	25	9	1
56																		
57	1	4	5	3.9	3.1	2.5	6	16	38	1741	3528	7918	32	10	3	12	6	3
58	4	4	7	3.2	2.8	2.4	14	24	41	2446	4654	8461	24	11	6	23	7	3
60																		
61																		
62	2	3	4	10.7	9.7	7.5	8	11	21	1747	3625	6801	29	15	6	22	11	5
63																		
65	4	5	5	4.0	3.6	3.0	23	32	51	7994	12403	18739	31	11	8	25	12	3
66	3	3	4	7.8	6.9	5.7	17	29	48	4272	9485	17825	29	16	5	20	9	2
67	1	3	4	6.9	4.6	3.5	5	45	482	1350	4465	59884	16	7	2	13	3	2
71	1	1	2	12.0	11.0	7.2	2	3	18	159	643	2426	16	5	3	8	2	2
72	0	1	2	0.0	2.4	1.9	0	2	30	0	85	2192	9	2	0	2	0	0
75																		
76	1	1	2	8.2	7.6	5.6	2	3	9	129	781	2196	18	8	2	10	3	2
80	1	1	2	3.1	2.7	2.2	1	4	13	33	469	2066	7	3	0	3	2	0

Appendix D.12. Fish richness and estimated richness (based on 3 consecutive 50-m reaches) calculated using CAPTURE® for 30 study sites. Fish assemblage metrics include endemics (END), cosmopolitans (COS), fluvial specialists (FLU), lentic tolerants (LEN), sensitives (SEN) and ratios of these variables.

Site No.	Stream name	Total Richness	Total Est Rich	END Est Rich	COS Est Rich	END/COS (Est Rich)	FLU Est Rich	LEN Est Rich	FLU/LEN (Est Rich)	SEN Est Rich
11	West Fork Pumpkinvine	18	19	1	16	0.06	12	8	1.5	2
18	Town Creek	18	21	6	11	0.55	16	5	3.2	5
24	Polecat Branch	12	17	3	8	0.38	12	4	3.0	3
26	Murphy Creek	18	19	3	11	0.27	17	4	4.3	6
27	Hickory Log Creek	14	15	6	8	0.75	11	5	2.2	4
31	Gorman Branch	19	21	4	11	0.36	17	4	4.3	7
33	Boston Creek	10	13	3	6	0.50	11	3	3.7	1
38	Black Mill Creek	20	25	1	17	0.06	21	5	4.2	4
44	Board Tree Creek	19	20	1	13	0.08	17	5	3.4	4
45	Upper Smithwick Creek	13	15	2	9	0.22	10	5	2.0	3
46	Scott Mill Creek	20	23	3	11	0.27	17	5	3.4	3
48	Toonigh Creek	7	8	1	5	0.20	5	3	1.7	0
52	Copper Sandy Creek	17	21	1	12	0.08	13	6	2.2	4
54	Trib. Sweat Mountain	18	20	3	12	0.25	14	6	2.3	3
55	East Fork Rubes Creek	13	14	2	9	0.22	9	6	1.5	0
56	West Fork Rubes Creek	12	13	4	6	0.67	10	3	3.3	1
57	Lower Noonday Creek	15	16	2	8	0.25	11	6	1.8	1
58	Upper Noonday Creek	12	16	0	13	0.00	5	11	0.5	0
60	Clark Creek	15	17	2	10	0.20	13	4	3.3	2
61	Proctor Creek	16	17	1	11	0.09	7	11	0.6	1
62	Butler Creek	10	12	1	7	0.14	7	5	1.4	3
63	Upper Allatoona Creek	19	20	5	12	0.42	14	8	1.8	4
65	Picketts Mill	18	20	4	9	0.44	17	3	5.7	5
66	Possum Creek	18	22	3	11	0.27	17	5	3.4	5
67	Westbrook Creek	19	20	2	13	0.15	16	5	3.2	3
71	Lawrence Creek	11	13	1	8	0.13	8	5	1.6	0
72	Lane Creek	11	12	0	8	0.00	6	6	1.0	0
75	Ward Creek	20	24	3	15	0.20	15	8	1.9	2
76	Upper Little Pumpkinvine	16	17	4	7	0.57	13	4	3.3	3
80	Buzzard Flapper	11	13	1	7	0.14	9	4	2.3	3
Mean		15.3	17.4	2.4	10.1	0.26	12.3	5.4	2.6	2.7
Minimum		7.0	8.0	0.0	5.0	0.00	5.0	3.0	0.5	0.0
Maximum		20.0	25.0	6.0	17.0	0.75	21.0	11.0	5.7	7.0

Appendix D.13. Fish abundance and estimated abundance (based on 3-pass removal) calculated using CAPTURE® for 30 study sites. Fish assemblage metrics include endemics (END), cosmopolitans (COS), fluvial specialists (FLU), lentic tolerants (LEN), sensitives (SEN) and ratios of these variables.

Site No.	Stream name	Total Abundance	Total Est Abund	END Est Abund	COS Est Abund	END/COS Abund	FLU Est Abund	LEN Est Abund	FLU/LEN Abund	SEN Est Abund
11	West Fork Pumpkinvine	89	115.6	4.0	97.6	0.04	40.2	75.4	0.5	6.0
18	Town Creek	132	135.1	46.1	54.6	0.84	91.5	43.6	2.1	15.4
24	Polecat Branch	62	66.3	3.0	18.6	0.16	49.7	16.6	3.0	37.7
26	Murphy Creek	232	243.0	123.4	51.9	2.38	221.0	22.0	10.0	29.0
27	Hickory Log Creek	133	177.0	157.0	12.0	13.08	175.0	2.0	87.5	3.0
31	Gorman Branch	435	499.7	334.7	64.5	5.19	469.1	30.6	15.3	95.6
33	Boston Creek	37	37.2	8.0	20.2	0.40	34.2	3.0	11.4	0.0
38	Black Mill Creek	373	394.5	82.8	149.2	0.55	361.8	32.7	11.1	20.0
44	Board Tree Creek	106	120.4	2.0	81.8	0.02	110.4	10.0	11.0	18.0
45	Upper Smithwick Creek	72	77.4	5.0	33.0	0.15	62.4	15.0	4.2	0.0
46	Scott Mill Creek	217	241.6	160.8	65.9	2.44	223.7	28.9	7.7	1.0
48	Toonigh Creek	124	152.9	50.0	52.6	0.95	117.3	35.6	3.3	0.0
52	Copper Sandy Creek	285	361.8	30.0	193.0	0.16	272.6	89.2	3.1	19.0
54	Trib. Sweat Mountain	116	125.9	62.0	33.3	1.86	94.6	31.3	3.0	1.0
55	East Fork Rubes Creek	176	196.7	23.0	139.7	0.16	111.4	85.4	1.3	0.0
56	West Fork Rubes Creek	99	120.0	11.0	87.0	0.13	81.1	38.9	2.1	2.0
57	Lower Noonday Creek	142	151.7	11.0	71.7	0.15	113.8	37.9	3.0	3.0
58	Upper Noonday Creek	96	118.5	0.0	62.6	0.00	59.0	59.6	1.0	0.0
60	Clark Creek	62	70.5	17.0	38.5	0.44	53.5	17.0	3.1	0.0
61	Proctor Creek	143	206.6	1.0	203.6	0.00	10.0	196.6	0.1	0.0
62	Butler Creek	65	66.6	7.0	49.6	0.14	23.0	43.6	0.5	1.0
63	Upper Allatoona Creek	372	512.2	51.2	430.9	0.12	101.4	411.9	0.2	21.2
65	Picketts Mill	211	381.8	42.0	70.9	0.59	348.9	34.9	10.0	9.0
66	Possum Creek	133	142.9	34.6	79.8	0.43	83.0	59.8	1.4	12.4
67	Westbrook Creek	110	111.2	3.0	65.3	0.05	84.0	27.2	3.1	14.0
71	Lawrence Creek	102	112.4	1.0	63.1	0.02	67.5	44.9	1.5	0.0
72	Lane Creek	106	148.5	0.0	139.5	0.00	84.7	64.8	1.3	0.0
75	Ward Creek	139	141.4	22.0	92.4	0.24	83.3	58.1	1.4	6.0
76	Upper Little Pumpkinvine	112	161.5	85.5	37.0	2.31	139.5	22.0	6.3	0.0
80	Buzzard Flapper	122	157.4	21.2	95.4	0.22	88.9	68.5	1.3	1.0
Mean		153.4	184.9	46.6	88.5	1.11	128.5	56.9	7.0	10.5
Minimum		37.0	37.2	0.0	12.0	0.00	10.0	2.0	0.1	0.0
Maximum		435.0	512.2	334.7	430.9	13.08	469.1	411.9	87.5	95.6

Appendix D.14. Fish species catch for first pass of 50-m reach for each stream (site ID across top).

Species	11	18	24	26	27	31	33	38	44	45	46	48	52	54	55	56	57	58	60	61	62	63	65	66	67	71	72	75	76
Petromyzontidae																													
Ichthyomyzon sp.	0	0	0	3	0	18	0	2	0	0	0	0	1	2	0	0	2	0	0	0	0	0	0	0	0	0	1	1	3
Cyprinidae																													
Campostoma oligolepis	1	4	2	3	3	0	7	64	10	3	10	0	14	1	22	5	4	3	8	0	4	7	1	5	0	8	15	2	10
Cyprinella callistia	0	9	21	7	0	0	0	0	2	0	1	0	1	0	0	0	0	0	0	0	0	10	1	3	2	0	0	0	0
Cyprinella trichoristia	0	0	0	0	2	30	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cyprinella venusta	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	2	0	0	0	0	7	0	0	0	0
Hybopsis lineapunctata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Hybopsis sp. cf. *H. winchelli*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	1	0	0	0	0	0	0	0
Luxilus zonistius	0	0	0	0	0	0	0	7	0	0	0	0	0	0	0	0	3	0	0	0	0	0	0	0	0	0	0	0	0
Nocomis leptocephalus	0	0	0	0	0	0	0	23	0	0	0	0	0	5	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Notemigonus crysoleucas	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Notropis chrosomus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Notropis longirostris	0	0	0	0	0	0	0	0	5	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	9	0	0	0	0
Notropis lutipinnis	0	0	0	0	0	0	0	40	9	0	7	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Notropis stilbius	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	7	0	0	0	0	0	0
Notropis xaenocephalus	0	0	0	44	11	95	0	0	0	0	7	0	0	19	0	5	8	0	0	0	0	0	2	11	0	0	0	12	8
Phenacobius catostomus	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Pimephales vigilax	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Semotilus atromaculatus	0	0	0	4	5	1	3	0	2	4	7	8	0	0	4	9	0	0	2	0	0	0	0	1	0	0	0	0	0
Catostomidae																													
Minytrema melanops	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	10	0
Hypentelium etowanum	6	4	2	10	4	12	6	18	11	4	2	15	20	1	8	13	0	4	4	0	7	1	4	7	7	17	3	4	6
Moxostoma duquesnei	0	1	0	0	0	0	0	7	0	0	0	0	0	0	0	0	3	0	0	0	1	0	4	1	3	0	0	4	0
Moxostoma erythrurum	0	0	0	0	0	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4	0	0	1	0
Moxostoma poecilurum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1	0
Ictaluridae																													
Ameiurus brunneus	0	0	2	0	0	0	0	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ameiurus natalis	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ameiurus nebulosus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0
Ictalurus punctatus	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Noturus leptacanthus	0	0	0	0	0	4	0	0	0	0	0	0	6	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0
Salmonidae																													
Onchorhynchus mykiss	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Fundulidae																													
Fundulus stellifer	0	4	0	4	0	1	0	0	0	2	2	10	35	5	12	3	41	13	3	0	1	2	36	0	11	8	3	0	6

Appendix D.14. Continued.

Species	11	18	24	26	27	31	33	38	44	45	46	48	52	54	55	56	57	58	60	61	62	63	65	66	67	71	72	75	76
Poecilidae																													
Gambusia affinis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	3	0	0	0	0	0	0	0	0	0
Gambusia holbrooki	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0
Gambusia holbrooki x affinis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0
Cottidae																													
Cottus sp. cf. *C. carolinae*	0	23	0	22	27	4	1	53	0	1	50	21	7	22	4	0	2	0	0	0	0	23	0	5	1	0	0	6	0
Centrarchidae																													
Ambloplites ariommus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lepomis auritus	5	20	6	4	1	11	1	3	2	4	2	9	9	4	10	3	11	10	1	0	6	22	7	13	11	15	16	12	6
Lepomis cyanellus	0	5	0	0	0	1	0	1	0	0	0	0	0	0	0	0	3	3	2	24	7	8	0	1	0	0	0	0	2
Lepomis gulosus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	1	0	0	0	0	0	0	0
Lepomis macrochirus	15	0	0	5	0	2	0	16	4	2	4	0	19	6	27	9	6	17	0	28	6	90	11	15	10	4	8	20	3
Lepomis megalotis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lepomis microlophus	1	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lepomis punctatus	0	0	0	0	0	0	0	0	0	0	0	0	1	1	5	0	0	0	0	0	0	0	0	0	0	0	0	2	1
Lepomis punctatus x miniatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lepomis macrochirus x auritus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
Micropterus coosae	0	0	0	4	1	2	0	3	0	0	0	0	1	1	1	0	3	0	0	0	0	1	2	1	1	0	0	3	2
Micropterus punctulatus	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
Micropterus salmoides	1	0	0	0	0	0	0	0	0	0	0	0	1	0	1	1	0	0	2	2	0	3	0	1	1	0	0	1	0
Pomoxis annularis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
Pomoxis nigromaculatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Percidae																													
Etheostoma etowahae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Etheostoma jordani	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Etheostoma scotti	0	10	2	8	19	4	1	0	0	0	11	0	0	1	0	0	0	0	8	0	3	5	2	3	0	0	0	0	0
Etheostoma stigmaeum	3	0	0	8	0	10	0	4	7	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	0	0	0
Perca flavescens	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
Percina kathae	0	0	0	5	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1	0	0	0	1	0
Percina nigrofasciata	6	2	0	6	0	1	0	10	9	2	2	0	20	0	0	1	6	0	0	4	0	3	15	4	4	2	3	7	4
Percina palmaris	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Percina sp. cf. *P. macrocephala*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Appendix D.15. Fish species catch for second pass of 50-m reach for each stream (site ID across top).

Species	11	18	24	26	27	31	33	38	44	45	46	48	52	54	55	56	57	58	60	61	62	63	65	66	67	71	72	75	76
Petromyzontidae																													
Ichthyomyzon sp.	1	0	0	3	0	19	0	3	0	0	1	0	4	0	0	0	4	0	0	0	0	0	0	0	0	0	0	5	4
Cyprinidae																													
Campostoma oligolepis	0	1	0	0	0	0	2	9	0	1	8	1	1	0	13	6	4	0	4	0	0	2	0	0	0	2	11	0	0
Cyprinella callistia	0	2	9	5	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	2	1	4	2	0	0	0	0
Cyprinella trichoristia	0	0	0	0	0	23	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cyprinella venusta	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	0	0	0
Hybopsis lineapunctata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Hybopsis sp. cf. *H. winchelli*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0
Luxilus zonistius	0	0	0	0	0	0	0	9	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nocomis leptocephalus	0	0	0	0	0	0	0	6	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Notemigonus crysoleucas	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Notropis chrosomus	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
Notropis longirostris	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	8	0	0	0	0
Notropis lutipinnis	0	0	0	0	0	0	0	8	5	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Notropis stilbius	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3	0	0	0	0	0	0
Notropis xaenocephalus	0	0	0	18	3	81	0	0	0	0	3	0	0	3	1	3	0	0	0	0	0	0	24	2	0	0	0	1	18
Phenacobius catostomus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Pimephales vigilax	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Semotilus atromaculatus	0	0	0	0	2	2	1	0	1	2	3	2	0	1	0	6	0	0	3	0	0	0	1	1	0	0	0	0	0
Catostomidae																													
Minytrema melanops	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	4	0
Hypentelium etowanum	0	1	1	3	2	6	1	4	3	4	0	6	10	2	3	3	1	2	0	0	2	2	3	2	1	7	0	3	2
Moxostoma duquesnei	0	0	0	0	0	0	0	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0
Moxostoma erythrurum	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	0	1	0
Moxostoma poecilurum	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0
Ictaluridae																													
Ameiurus brunneus	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ameiurus natalis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ameiurus nebulosus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0
Ictalurus punctatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Noturus leptacanthus	0	0	0	0	0	0	0	0	0	0	0	0	2	1	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
Salmonidae																													
Onchorhynchus mykiss	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Fundulidae																													
Fundulus stellifer	1	3	0	6	0	1	0	0	0	18	4	9	13	6	3	2	7	12	0	0	0	3	2	0	7	3	0	0	3

Appendix D.15. Continued.

Species	11	18	24	26	27	31	33	38	44	45	46	48	52	54	55	56	57	58	60	61	62	63	65	66	67	71	72	75	76
Poecilidae																													
Gambusia affinis	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	1	0	0
Gambusia holbrooki	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3	0	0	0	0	0	0	2	0	0
Gambusia holbrooki x affinis	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3	0	0	0	0	0	0	1	0	0
Cottidae																													
Cottus sp. cf. *C. carolinae*	2	8	0	11	21	4	5	9	0	3	25	12	1	6	6	0	1	0	0	0	0	9	2	4	1	0	0	1	2
Centrarchidae																													
Ambloplites ariommus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lepomis auritus	0	6	1	3	1	7	0	1	1	8	4	7	4	0	6	3	2	3	1	0	4	16	2	3	0	9	4	4	1
Lepomis cyanellus	2	1	0	0	0	1	1	0	0	0	0	0	0	0	0	0	2	4	3	21	3	8	0	0	0	0	1	4	2
Lepomis gulosus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	0	0	0	0	0	0	0	0
Lepomis macrochirus	6	0	0	2	0	0	0	6	1	0	11	0	11	7	15	10	2	4	2	16	4	70	5	8	2	3	6	7	0
Lepomis megalotis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lepomis microlophus	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
Lepomis punctatus	0	0	2	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Lepomis punctatus x miniatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lepomis macrochirus x auritus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Micropterus coosae	0	0	2	2	0	0	0	0	0	0	1	0	0	0	0	0	0	1	1	0	0	0	4	5	0	0	0	0	1
Micropterus punctulatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Micropterus salmoides	2	0	0	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1	1	0	0
Pomoxis annularis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Pomoxis nigromaculatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
Percidae																													
Etheostoma etowahae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Etheostoma jordani	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Etheostoma scotti	0	1	1	6	11	2	0	0	0	0	10	0	0	1	0	0	0	0	3	1	4	3	4	4	0	0	0	0	2
Etheostoma stigmaeum	2	0	0	2	0	9	0	2	2	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
Perca flavescens	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
Percina kathae	0	1	0	2	0	1	0	0	0	0	1	0	0	0	0	0	0	0	1	0	0	1	0	0	0	0	0	2	0
Percina nigrofasciata	5	0	0	0	0	1	0	5	7	1	1	0	5	0	0	0	2	0	0	2	1	2	7	0	1	3	3	3	5
Percina palmaris	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Percina sp. cf. *P. macrocephala*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Appendix D.16. Fish species catch for third pass of 50-m reach for each stream (site ID across top).

Species	11	18	24	26	27	31	33	38	44	45	46	48	52	54	55	56	57	58	60	61	62	63	65	66	67	71	72	75
Petromyzontidae																												
Ichthyomyzon sp.	1	0	0	2	0	26	0	4	2	0	0	0	2	1	0	0	4	0	0	0	0	0	0	0	0	0	2	4
Cyprinidae																												
Campostoma oligolepis	1	0	0	0	0	0	1	7	1	1	3	0	10	0	9	3	1	0	3	0	0	1	0	1	0	1	10	0
Cyprinella callistia	0	3	4	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4	1	1	0	0	0	0
Cyprinella trichoristia	0	0	0	0	0	6	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cyprinella venusta	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	0	0
Hybopsis lineapunctata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Hybopsis sp. cf. *H. winchelli*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0
Luxilus zonistius	0	0	0	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nocomis leptocephalus	0	0	0	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Notemigonus crysoleucas	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Notropis chrosomus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Notropis longirostris	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0
Notropis lutipinnis	0	0	0	0	0	0	0	10	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Notropis stilbius	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Notropis xaenocephalus	0	0	0	3	2	27	0	0	0	0	1	0	0	4	0	0	0	0	0	0	0	0	2	1	0	1	0	1
Phenacobius catostomus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Pimephales vigilax	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Semotilus atromaculatus	0	1	0	1	0	0	2	0	0	4	1	4	0	0	0	5	0	0	0	0	0	0	0	1	0	1	0	0
Catostomidae																												
Minytrema melanops	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Hypentelium etowanum	1	2	2	4	1	4	1	3	0	3	2	3	12	0	1	1	0	1	0	0	0	1	3	1	0	4	1	0
Moxostoma duquesnei	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1	0	0	0
Moxostoma erythrurum	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Moxostoma poecilurum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ictaluridae																												
Ameiurus brunneus	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ameiurus natalis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
Ameiurus nebulosus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
Ictalurus punctatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Noturus leptacanthus	0	0	0	0	0	0	0	0	0	0	0	0	9	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Salmonidae																												
Onchorhynchus mykiss	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Fundulidae																												
Fundulus stellifer	3	0	0	0	0	0	0	0	0	3	1	3	10	2	1	0	4	7	1	0	0	0	32	2	2	4	0	0

Appendix D.16. Continued.

Species	11	18	24	26	27	31	33	38	44	45	46	48	52	54	55	56	57	58	60	61	62	63	65	66	67	71	72	75
Poecilidae																												
Gambusia affinis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Gambusia holbrooki	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Gambusia holbrooki x affinis	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cottidae																												
Cottus sp. cf. *C. carolinae*	2	2	0	3	16	0	1	15	1	1	17	8	22	4	12	1	0	0	0	0	0	6	0	2	1	0	0	1
Centrarchidae																												
Ambloplites ariommus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lepomis auritus	0	6	4	1	0	4	0	1	0	1	1	6	3	3	3	1	1	6	3	0	2	10	4	5	0	3	2	2
Lepomis cyanellus	0	1	0	0	0	0	0	0	0	0	0	0	1	0	0	0	2	1	2	16	8	7	0	0	0	1	0	0
Lepomis gulosus	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lepomis macrochirus	11	2	0	5	0	1	0	3	2	0	5	0	11	3	8	4	7	2	0	6	2	39	2	6	3	3	6	2
Lepomis megalotis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lepomis microlophus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lepomis punctatus	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lepomis punctatus x miniatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lepomis macrochirus x auritus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Micropterus coosae	0	2	0	1	0	5	0	0	0	0	0	0	0	1	0	0	0	0	1	0	0	0	0	1	0	1	0	0
Micropterus punctulatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Micropterus salmoides	1	0	0	0	0	0	0	0	0	0	0	0	2	0	1	0	1	0	0	0	0	1	0	0	0	0	1	0
Pomoxis annularis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Pomoxis nigromaculatus	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Percidae																												
Etheostoma etowahae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Etheostoma jordani	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Etheostoma scotti	0	1	0	3	0	1	0	0	0	0	6	0	0	1	0	0	0	0	0	0	0	1	6	1	0	0	0	0
Etheostoma stigmaeum	1	0	0	4	0	1	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Perca flavescens	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Percina kathae	0	1	0	2	0	1	1	0	0	0	1	0	0	0	0	0	0	0	2	0	0	1	1	0	0	0	0	0
Percina nigrofasciata	1	2	0	0	0	0	0	1	6	0	0	0	11	1	0	0	5	0	0	1	0	1	4	0	1	1	5	3
Percina palmaris	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Percina sp. cf. *P. macrocephala*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Appendix D.17. Fish species catch for lower 50-m reach of each stream (site ID across top).

Species	11	18	24	26	27	31	33	38	44	45	46	48	52	54	55	56	57	58	60	61	62	63	65	66	67	71	72	75
Petromyzontidae																												
Ichthyomyzon sp.	0	0	0	0	0	18	0	2	0	0	0	0	1	5	0	0	2	0	0	0	0	0	0	0	0	0	3	1
Cyprinidae																												
Campostoma oligolepis	1	4	42	3	2	0	5	64	10	8	8	0	14	0	22	2	4	3	11	0	12	7	0	2	0	3	5	2
Cyprinella callistia	0	23	24	2	0	0	0	0	2	0	0	0	1	0	0	0	0	0	0	0	0	10	0	8	36	0	0	0
Cyprinella trichoristia	0	1	0	0	9	30	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cyprinella venusta	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	25	0	0	0
Hybopsis lineapunctata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Hybopsis sp. cf. *H. winchelli*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	5	0	0	1	0	0	0	0	0	0
Luxilus zonistius	0	0	0	0	0	0	0	7	0	0	0	0	0	0	0	0	3	0	0	0	0	0	0	0	0	0	0	0
Nocomis leptocephalus	0	0	0	0	0	0	0	23	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Notemigonus crysoleucas	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Notropis chrosomus	0	0	0	0	8	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0
Notropis longirostris	0	0	0	0	0	0	0	0	5	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	15	0	0	0
Notropis lutipinnis	0	0	0	0	0	0	0	40	9	0	5	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Notropis stilbius	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Notropis xaenocephalus	0	1	0	79	9	95	0	0	0	0	6	0	0	14	0	2	8	0	0	0	0	0	0	28	7	11	0	12
Phenacobius catostomus	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Pimephales vigilax	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Semotilus atromaculatus	0	2	0	1	9	1	7	0	2	0	7	8	0	0	4	14	0	0	8	0	0	0	0	0	0	3	1	0
Catostomidae																												
Minytrema melanops	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	10
Hypentelium etowanum	0	7	15	13	10	12	0	18	11	4	3	15	20	1	8	9	0	4	7	0	3	1	6	1	0	20	2	4
Moxostoma duquesnei	2	0	0	6	0	0	0	7	0	0	0	0	0	0	0	0	3	0	0	0	0	0	7	0	0	0	0	4
Moxostoma erythrurum	1	0	0	0	0	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	1
Moxostoma poecilurum	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Ictaluridae																												
Ameiurus brunneus	0	0	4	0	0	0	0	1	0	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ameiurus natalis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
Ameiurus nebulosus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0
Ictalurus punctatus	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Noturus leptacanthus	0	0	0	0	0	4	0	0	0	0	0	0	6	1	0	0	0	0	0	0	0	2	0	2	0	0	0	0
Salmonidae																												
Onchorhynchus mykiss	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Fundulidae																												
Fundulus stellifer	0	0	0	1	0	1	0	0	0	5	1	10	35	4	12	2	41	13	1	0	1	2	15	0	20	18	5	0

Appendix D.17. Continued.

Species	11	18	24	26	27	31	33	38	44	45	46	48	52	54	55	56	57	58	60	61	62	63	65	66	67	71	72	75
Poecilidae																												
Gambusia affinis	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	3	0	0	0	0	0	0	0	0
Gambusia holbrooki	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0
Gambusia holbrooki x affinis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0
Cottidae																												
Cottus sp. cf. *C. carolinae*	1	79	0	16	34	4	5	53	0	17	29	21	7	0	4	0	2	0	0	0	0	23	4	6	3	0	0	6
Centrarchidae																												
Ambloplites ariommus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lepomis auritus	4	1	42	7	7	11	2	3	2	7	5	9	9	13	10	1	11	10	10	0	2	22	4	4	0	12	11	12
Lepomis cyanellus	0	1	0	0	0	1	0	1	0	0	0	0	0	2	0	0	3	3	3	24	7	8	0	1	1	5	4	0
Lepomis gulosus	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	1	0	0	0	0	0	0
Lepomis macrochirus	27	1	2	10	3	2	0	16	4	1	0	0	19	18	27	11	6	17	4	28	7	90	7	14	9	3	7	20
Lepomis megalotis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lepomis microlophus	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lepomis punctatus	0	0	0	0	1	0	0	0	0	1	0	0	1	0	5	0	0	0	0	0	0	0	0	0	0	0	0	2
Lepomis punctatus x miniatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lepomis macrochirus x auritus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Micropterus coosae	0	0	0	17	3	2	0	3	0	1	0	0	1	0	1	0	3	0	0	0	0	1	3	2	1	0	0	3
Micropterus punctulatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3	0	0	0	0	0	0	0	0	0
Micropterus salmoides	2	0	0	0	0	0	0	0	0	1	1	0	1	0	1	1	0	0	2	2	0	3	1	0	1	1	0	1
Pomoxis annularis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Pomoxis nigromaculatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Percidae																												
Etheostoma etowahae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Etheostoma jordani	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Etheostoma scotti	0	6	1	10	12	4	0	0	0	7	4	0	0	0	0	0	0	0	5	0	15	5	7	9	0	0	0	0
Etheostoma stigmaeum	4	0	0	7	0	10	0	4	7	0	1	0	0	0	0	0	0	0	1	0	0	0	0	0	1	0	0	0
Perca flavescens	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0
Percina kathae	0	1	1	0	0	2	0	0	0	0	3	0	0	0	0	0	0	0	1	0	0	1	0	0	0	0	0	1
Percina nigrofasciata	5	5	0	4	2	1	0	10	9	2	2	0	20	1	0	5	6	0	1	4	0	3	4	0	5	4	7	7
Percina palmaris	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Percina sp. cf. *P. macrocephala*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Appendix D.18. Fish species catch for middle 50-m reach of each stream (site ID across top).

Species	11	18	24	26	27	31	33	38	44	45	46	48	52	54	55	56	57	58	60	61	62	63	65	66	67	71	72	75
Petromyzontidae																												
Ichthyomyzon sp.	0	0	0	3	0	7	0	7	0	0	0	0	4	2	0	0	0	0	0	0	0	0	0	0	0	0	1	0
Cyprinidae																												
Campostoma oligolepis	1	4	1	3	3	1	7	9	7	5	10	7	1	1	4	2	6	0	8	0	4	10	1	5	11	8	7	0
Cyprinella callistia	0	9	38	7	0	6	0	0	2	0	1	0	0	0	0	0	0	0	0	0	0	10	1	3	22	0	0	0
Cyprinella trichoristia	0	0	0	0	2	25	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cyprinella venusta	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	22	0	0	0
Hybopsis lineapunctata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0
Hybopsis sp. cf. *H. winchelli*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Luxilus zonistius	0	0	0	0	0	0	0	14	0	0	0	0	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0	0
Nocomis leptocephalus	0	0	0	0	0	0	0	11	3	0	0	0	0	5	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Notemigonus crysoleucas	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Notropis chrosomus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0
Notropis longirostris	0	0	0	0	0	0	0	0	13	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	11	0	0	0
Notropis lutipinnis	0	0	0	0	0	0	0	27	4	0	7	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Notropis stilbius	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	7	0	0	0	0	0
Notropis xaenocephalus	0	0	0	44	11	23	0	0	0	0	7	0	0	19	4	3	68	0	0	0	0	0	2	11	5	0	0	11
Phenacobius catostomus	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Pimephales vigilax	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Semotilus atromaculatus	0	0	1	4	5	3	3	2	1	3	7	5	0	0	1	41	0	0	2	0	0	0	0	1	1	0	2	1
Catostomidae																												
Minytrema melanops	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2
Hypentelium etowanum	6	4	4	10	4	5	6	12	17	4	2	19	10	1	5	24	8	4	4	0	7	0	4	7	8	17	3	1
Moxostoma duquesnei	0	1	0	0	0	0	0	3	0	0	0	0	0	0	0	0	0	0	0	0	1	0	4	1	2	0	0	0
Moxostoma erythrurum	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Moxostoma poecilurum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ictaluridae																												
Ameiurus brunneus	0	0	1	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ameiurus natalis	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ameiurus nebulosus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3	0	0	0	0	0	0	0	0
Ictalurus punctatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
Noturus leptacanthus	0	0	0	0	0	6	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Salmonidae																												
Onchorhynchus mykiss	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Fundulidae																												
Fundulus stellifer	0	4	0	4	0	0	0	0	0	20	2	0	13	5	1	0	31	0	3	1	1	2	36	0	13	8	1	0

Appendix D.18. Continued.

Species	11	18	24	26	27	31	33	38	44	45	46	48	52	54	55	56	57	58	60	61	62	63	65	66	67	71	72	75
Poecilidae																												
Gambusia affinis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0
Gambusia holbrooki	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0
Gambusia holbrooki x affinis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0
Cottidae																												
Cottus sp. cf. *C. carolinae*	0	23	1	22	27	25	1	13	0	2	50	66	1	22	4	1	1	0	0	0	0	8	0	5	2	0	0	25
Centrarchidae																												
Ambloplites ariommus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lepomis auritus	5	20	4	4	1	0	1	11	15	5	2	11	4	4	14	7	3	2	1	8	6	4	7	13	10	15	21	7
Lepomis cyanellus	0	5	0	0	0	0	0	1	0	0	0	0	0	0	1	0	0	1	2	22	7	4	0	1	0	0	1	0
Lepomis gulosus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3	0	0	0	0	0	0	5	0
Lepomis macrochirus	15	0	0	5	0	0	0	27	13	4	4	0	11	6	26	26	6	19	0	52	6	48	11	15	18	4	12	6
Lepomis megalotis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lepomis microlophus	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	7	0	0	0	0	0	0	0	0
Lepomis punctatus	0	0	0	0	0	0	0	0	0	0	0	0	0	1	2	0	0	0	0	0	0	0	0	0	0	0	0	2
Lepomis punctatus x miniatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lepomis macrochirus x auritus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0
Micropterus coosae	0	0	2	4	1	1	0	0	0	0	0	0	0	1	1	1	1	1	0	0	0	1	2	1	0	0	0	2
Micropterus punctulatus	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0
Micropterus salmoides	1	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	2	1	0	0	0	1	0	0	0	0
Pomoxis annularis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Pomoxis nigromaculatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Percidae																												
Etheostoma etowahae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Etheostoma jordani	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Etheostoma scotti	0	10	5	8	19	8	1	0	0	2	11	0	0	1	0	0	0	0	8	3	3	0	2	3	0	0	0	2
Etheostoma stigmaeum	3	0	0	8	0	1	0	4	6	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	0	0
Perca flavescens	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Percina kathae	0	0	0	5	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1	0	0	1	1	0	0	0	0
Percina nigrofasciata	6	2	0	6	0	4	0	8	8	1	2	0	5	0	2	3	5	0	0	9	0	5	15	4	2	2	3	2
Percina palmaris	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Percina sp. cf. *P. macrocephala*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Appendix D.19. Fish species catch for upper 50-m reach of each stream (site ID across top).

Species	11	18	24	26	27	31	33	38	44	45	46	48	52	54	55	56	57	58	60	61	62	63	65	66	67	71	72	75	76
Petromyzontidae																													
Ichthyomyzon sp.	0	0	0	1	0	6	0	3	2	0	0	0	0	1	0	0	1	0	0	0	0	0	0	0	0	0	1	0	3
Cyprinidae																													
Campostoma oligolepis	6	2	2	1	2	6	8	2	12	3	75	4	2	4	4	5	11	0	16	0	14	18	17	4	0	1	15	2	10
Cyprinella callistia	0	9	21	2	0	11	0	0	1	0	1	0	0	0	0	0	0	0	0	0	0	15	7	2	2	0	0	0	0
Cyprinella trichoristia	0	0	0	0	4	124	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cyprinella venusta	10	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	5	0	7	0	0	0	0
Hybopsis lineapunctata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Hybopsis sp. cf. *H. winchelli*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	34	0	0	0	0	0	0	0	0	0	0
Luxilus zonistius	0	0	0	0	0	0	0	10	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nocomis leptocephalus	0	0	0	0	0	0	0	8	1	0	0	0	1	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Notemigonus crysoleucas	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Notropis chrosomus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Notropis longirostris	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	9	0	0	0	0
Notropis lutipinnis	0	0	0	0	0	0	0	28	0	0	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Notropis stilbius	0	0	0	0	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0	0	0	7	3	0	0	0	0	0	0
Notropis xaenocephalus	0	0	0	4	8	65	0	0	0	0	1	0	0	29	31	5	0	0	0	0	0	0	57	4	0	6	0	4	8
Phenacobius catostomus	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Pimephales vigilax	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Semotilus atromaculatus	0	1	0	0	5	0	1	0	0	4	1	7	0	3	8	9	0	0	2	0	0	0	0	2	0	2	0	0	0
Catostomidae																													
Minytrema melanops	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0
Hypentelium etowanum	6	5	2	17	16	11	10	4	11	4	1	4	8	4	7	13	0	0	10	1	5	3	17	5	7	14	3	7	6
Moxostoma duquesnei	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	14	0	3	0	0	10	0
Moxostoma erythrurum	1	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	4	0	0	1	0
Moxostoma poecilurum	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0
Ictaluridae																													
Ameiurus brunneus	0	0	2	1	0	0	0	0	2	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ameiurus natalis	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0
Ameiurus nebulosus	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	21	0	0	0	0	0	0	0	0	0
Ictalurus punctatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	0	0	0	0	0	0
Noturus leptacanthus	0	0	0	0	0	2	0	0	0	0	0	0	7	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Salmonidae																													
Onchorhynchus mykiss	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Fundulidae																													
Fundulus stellifer	1	0	0	3	0	1	0	0	0	2	1	5	5	9	1	3	2	3	2	4	2	1	0	3	11	1	3	0	6

Appendix D.19. Continued.

Species	11	18	24	26	27	31	33	38	44	45	46	48	52	54	55	56	57	58	60	61	62	63	65	66	67	71	72	75	76
Poecilidae																													
Gambusia affinis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Gambusia holbrooki	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	13	0	0	0	0	0	0	1	0	0
Gambusia holbrooki x affinis	1	0	0	0	19	0	0	0	0	0	0	0	0	0	0	0	0	0	0	9	0	0	0	0	0	0	0	0	0
Cottidae																													
Cottus sp. cf. *C. carolinae*	2	35	0	16	19	29	10	15	0	1	32	32	3	5	13	0	0	0	0	0	0	7	8	2	1	0	0	32	0
Centrarchidae																													
Ambloplites ariommus	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lepomis auritus	7	5	6	6	5	4	0	1	14	4	0	7	2	6	6	3	6	3	24	14	0	3	9	4	11	6	16	4	6
Lepomis cyanellus	0	2	0	0	0	0	0	0	2	0	1	0	0	0	1	0	2	8	6	22	4	1	0	0	0	0	0	2	2
Lepomis gulosus	2	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	1	0	1	0	1	0	0	0	0	0	0	0
Lepomis macrochirus	23	2	0	26	1	0	0	7	13	2	12	1	8	17	1	9	4	17	13	90	0	46	10	7	10	1	8	9	3
Lepomis megalotis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lepomis microlophus	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0
Lepomis punctatus	0	0	0	0	1	0	0	0	0	0	2	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1
Lepomis punctatus x miniatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lepomis macrochirus x auritus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Micropterus coosae	0	0	0	5	1	7	0	0	0	0	2	0	0	2	1	0	1	0	0	0	0	0	1	3	1	2	0	3	2
Micropterus punctulatus	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
Micropterus salmoides	1	1	0	0	0	0	2	0	0	0	0	0	0	0	0	1	1	1	1	0	0	1	2	0	1	0	0	0	0
Pomoxis annularis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Pomoxis nigromaculatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Percidae																													
Etheostoma etowahae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Etheostoma jordani	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Etheostoma scotti	0	0	2	9	22	9	0	0	0	0	13	0	0	1	0	0	0	0	11	2	13	2	5	1	0	0	0	3	0
Etheostoma stigmaeum	5	0	0	2	0	3	0	6	1	0	2	0	0	0	0	0	0	0	1	0	0	0	0	0	1	0	0	0	0
Perca flavescens	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Percina kathae	0	1	0	0	0	2	1	0	1	0	1	0	0	1	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
Percina nigrofasciata	2	4	0	5	3	5	3	14	10	2	3	0	7	0	0	1	1	0	0	1	1	4	4	2	4	10	3	9	4
Percina palmaris	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Percina sp. cf. *P. macrocephala*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Appendix D.20. Total fish catch within first pass of three 50-m reaches of each stream (site ID across top).

Species	11	18	24	26	27	31	33	38	44	45	46	48	52	54	55	56	57	58	60	61	62	63	65	66	67	71	72	75
Petromyzontidae																												
Ichthyomyzon sp.	0	0	0	4	0	31	0	12	2	0	0	0	5	8	0	0	3	0	0	0	0	0	0	0	0	0	5	1
Cyprinidae																												
Campostoma oligolepis	8	10	45	7	7	7	20	75	29	16	93	11	17	5	30	9	21	3	35	0	30	35	18	11	11	12	27	4
Cyprinella callistia	0	41	83	11	0	17	0	0	5	0	2	0	1	0	0	0	0	0	0	0	0	35	8	13	60	0	0	0
Cyprinella trichoristia	0	1	0	0	15	179	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cyprinella venusta	11	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	2	0	0	5	0	54	0	0	0
Hybopsis lineapunctata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Hybopsis sp. cf. *H. winchelli*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	41	0	0	1	0	0	0	0	0	0
Luxilus zonistius	0	0	0	0	0	0	0	31	0	0	0	0	0	0	0	0	7	0	0	0	0	0	0	0	0	0	0	0
Nocomis leptocephalus	0	0	0	0	0	0	0	42	4	0	1	0	1	7	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Notemigonus crysoleucas	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Notropis chrosomus	0	0	0	0	8	0	0	0	0	0	0	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0	0	0
Notropis longirostris	0	0	0	0	0	0	0	0	19	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	35	0	0	0
Notropis lutipinnis	0	0	0	0	0	0	0	95	13	0	15	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Notropis stilbius	0	0	0	0	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0	0	0	7	10	0	0	0	0	0
Notropis xaenocephalus	0	1	0	127	28	183	0	0	0	0	14	0	0	62	35	10	76	0	0	0	0	0	59	43	12	17	0	27
Phenacobius catostomus	0	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Pimephales vigilax	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Semotilus atromaculatus	0	3	1	5	19	4	11	2	3	7	15	20	0	3	13	64	0	0	12	0	0	0	0	3	1	5	3	1
Catostomidae																												
Minytrema melanops	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	14
Hypentelium etowanum	12	16	21	40	30	28	16	34	39	12	6	38	38	6	20	46	8	8	21	1	15	4	27	13	15	51	8	12
Moxostoma duquesnei	3	1	0	6	0	0	0	10	0	0	0	0	0	0	0	0	3	0	0	0	1	0	25	1	5	0	0	14
Moxostoma erythrurum	2	0	0	0	0	1	0	1	4	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	4	0	0	2
Moxostoma poecilurum	5	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1
Ictaluridae																												
Ameiurus brunneus	0	0	7	1	0	0	0	1	4	1	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ameiurus natalis	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1	0	0	0	0	0	0	0
Ameiurus nebulosus	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	26	0	0	0	0	0	0	0	0
Ictalurus punctatus	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	2	0	1	0	0	0	0	0	0
Noturus leptacanthus	0	0	0	0	0	12	0	0	0	0	0	0	15	1	0	0	0	0	0	0	0	2	0	2	0	0	0	0
Salmonidae																												
Onchorhynchus mykiss	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Fundulidae																												
Fundulus stellifer	1	4	0	8	0	2	0	0	0	27	4	15	53	18	14	5	74	16	6	5	4	5	51	3	44	27	9	0

Appendix D.20. Continued.

Species	11	18	24	26	27	31	33	38	44	45	46	48	52	54	55	56	57	58	60	61	62	63	65	66	67	71	72	75
Poecilidae																												
Gambusia affinis	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	3	0	0	0	0	0	0	0	0
Gambusia holbrooki	0	0	0	0	0	0	0	0	0	0	0	0	3	0	0	0	0	0	0	17	0	0	0	0	0	0	1	0
Gambusia holbrooki x affinis	1	0	0	0	19	0	0	0	0	0	0	0	0	0	0	0	0	0	0	13	0	0	0	0	0	0	0	0
Cottidae																												
Cottus sp. cf. *C. carolinae*	3	137	1	54	80	58	16	81	0	20	111	119	11	27	21	1	3	0	0	0	0	38	12	13	6	0	0	63
Centrarchidae																												
Ambloplites ariommus	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lepomis auritus	16	26	52	17	13	15	3	15	31	16	7	27	15	23	30	11	20	15	35	22	8	29	20	21	21	33	48	23
Lepomis cyanellus	0	8	0	0	0	1	0	2	2	0	1	0	0	2	2	0	5	12	11	68	18	13	0	2	1	5	5	2
Lepomis gulosus	3	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	2	0	4	0	2	0	0	0	0	5	0
Lepomis macrochirus	65	3	2	41	4	2	0	50	30	7	16	1	38	41	54	46	16	53	17	170	13	184	28	36	37	8	27	35
Lepomis megalotis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lepomis microlophus	5	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	1	0	7	0	0	0	0	0	0	0	0
Lepomis punctatus	0	0	0	0	2	0	0	0	0	1	2	0	1	3	7	0	0	0	0	0	0	0	0	0	0	0	0	5
Lepomis punctatus x miniatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lepomis macrochirus x auritus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0
Micropterus coosae	0	0	2	26	5	10	0	3	0	1	2	0	1	3	3	1	5	1	0	0	0	2	6	6	2	2	0	8
Micropterus punctulatus	0	0	0	0	0	0	1	1	1	0	0	0	0	0	0	0	0	0	5	0	0	0	0	0	0	0	0	0
Micropterus salmoides	4	1	0	0	0	0	2	0	1	2	1	0	1	0	1	2	1	1	5	3	0	4	3	1	2	1	0	1
Pomoxis annularis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Pomoxis nigromaculatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Percidae																												
Etheostoma etowahae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Etheostoma jordani	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Etheostoma scotti	0	16	8	27	53	21	1	0	0	9	28	0	0	2	0	0	0	0	24	5	31	7	14	13	0	0	0	5
Etheostoma stigmaeum	12	0	0	17	0	14	0	14	14	0	3	0	0	0	0	0	0	0	2	0	0	0	1	0	3	0	0	0
Perca flavescens	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0
Percina kathae	0	2	1	5	0	4	1	0	2	0	4	0	0	1	0	0	0	0	1	1	0	2	1	1	0	0	0	1
Percina nigrofasciata	13	11	0	15	5	10	3	32	27	5	7	0	32	1	2	9	12	0	1	14	1	12	23	6	11	16	13	18
Percina palmaris	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Percina sp. cf. *P. macrocephala*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

APPENDIX E

SUPPLEMENTAL MATERIAL TO CHAPTER 5

Appendix E.1. Land cover data for 30 study sites.

Site	Basin Area	Road Xings	Impound	Catchment land cover (2001)					Land cover change (1992-2001)				Riparian
ID	(km^2)	(no. km^{-1})	(no. km^{-1})	Imperv	Urban	Forest	Agric	Water	Urban	Forest	Agric	Water	% Forest
11	10.5	0.16	0.49	6.3	4.5	62.5	13.2	0.6	0.7	-2.3	-1.0	0.0	68.3
18	15.4	0.58	0.39	13.0	20.2	57.6	16.3	0.3	2.6	5.2	-1.5	0.0	100.0
24	12.1	0.26	0.90	10.1	22.0	59.5	12.3	1.3	5.5	4.4	-4.1	0.0	55.1
26	9.5	0.31	0.49	5.0	4.7	69.0	16.1	0.5	1.7	-2.5	-3.1	0.1	82.4
27	10.7	0.39	0.51	4.9	9.7	64.2	17.3	0.6	1.4	-6.4	-1.6	0.2	92.8
31	12.1	0.00	0.07	1.7	1.4	96.4	0.4	0.0	0.2	0.0	-0.1	0.0	90.3
33	13.5	0.22	0.12	3.3	4.0	86.2	3.0	0.3	0.5	-3.5	0.3	0.0	75.8
38	16.4	0.44	0.34	7.2	9.0	52.6	13.8	0.2	4.0	-6.2	-5.7	0.0	93.9
44	17.5	0.27	0.64	6.5	8.0	53.9	27.3	1.2	1.7	-8.2	-2.0	0.3	78.8
45	15.8	0.31	0.65	8.0	15.6	51.2	24.4	0.9	4.9	-6.9	-3.0	0.2	61.2
46	12.0	0.70	1.11	9.4	21.2	59.6	9.2	0.6	3.1	-7.6	-2.4	0.1	70.1
48	16.2	1.03	0.28	15.4	48.6	37.1	7.1	0.4	7.8	-8.9	-3.6	0.1	65.3
52	15.6	0.29	0.70	13.9	18.7	57.2	13.0	2.0	10.0	-11.2	-3.2	0.1	94.4
54	8.5	0.58	0.52	13.7	44.9	46.8	2.1	1.5	7.0	-8.3	-1.5	0.1	60.9
55	11.2	0.77	0.47	20.4	57.5	32.2	0.2	0.7	7.5	-5.7	-2.3	0.0	96.9
56	11.2	0.39	0.70	21.8	60.7	32.7	0.2	1.1	11.3	-9.3	-4.8	0.1	71.0
57	17.0	0.45	0.55	23.7	64.5	30.3	0.1	0.9	6.7	-3.4	-3.9	-0.1	69.7
58	9.8	0.73	0.53	31.0	65.1	29.0	1.9	0.5	5.1	-2.0	-2.6	0.0	74.3
60	14.5	0.60	0.54	17.5	49.7	39.9	3.7	0.7	13.3	-12.6	-5.0	0.0	45.7
61	19.9	0.39	0.50	29.8	50.2	34.9	2.7	0.5	24.1	-25.2	-5.8	0.0	86.4
62	14.6	0.43	0.78	22.0	57.0	37.4	1.7	0.4	10.8	-9.5	-2.6	0.1	86.8
63	19.0	0.35	0.68	12.6	24.6	60.8	8.5	0.8	7.8	-5.7	-3.3	0.1	62.1
65	12.5	0.59	0.67	13.3	25.7	60.8	7.8	0.4	8.0	-7.6	-0.1	0.0	64.7
66	16.4	0.33	0.54	9.7	14.3	63.1	10.4	1.4	7.1	-10.1	-1.0	0.1	96.9
67	11.2	0.19	0.31	8.5	20.3	65.7	5.6	1.1	4.4	-9.7	1.9	-0.1	39.1
71	11.4	0.60	0.28	9.7	8.5	74.1	9.3	1.1	2.7	-6.9	1.6	0.3	89.7
72	10.3	0.17	0.69	7.1	6.5	72.3	6.0	1.7	3.7	-11.1	0.2	0.3	86.2
75	17.0	0.15	0.36	4.9	3.0	71.9	15.1	0.9	1.4	-6.9	1.4	0.0	97.0
76	16.6	0.12	0.14	3.5	1.7	81.8	5.1	0.1	1.4	-10.6	1.5	0.1	84.1
80	9.8	0.41	0.37	7.2	14.1	55.8	22.1	0.9	1.5	-3.9	-2.8	0.1	50.7
Mean	13.6	0.4	0.5	12.0	25.2	56.6	9.2	0.8	5.6	-6.8	-2.0	0.1	76.3
Min	8.5	0.0	0.1	1.7	1.4	29.0	0.1	0.0	0.2	-25.2	-5.8	-0.1	39.1
Max	19.9	1.0	1.1	31.0	65.1	96.4	27.3	2.0	24.1	5.2	1.9	0.3	100.0

Appendix E.2. Geomorphic data for 30 study sites. Sediment principle components axis (PCA) calculated from 10 geomorphic variables (excluding slope).

Site ID	Slope	St.dev. Velocity ($m\ s^{-1}$)	St.dev. Particle Size (phi)	St.dev. Depth (ft)	Rugosity	% Riffle Habitat	% Fines in Riffles	St.dev. % Fines	Froude #	Bed Mobility	Turbidity (NTU)	Sediment PCA
11	0.002	0.18	1.43	0.50	1.12	10.3	24.5	9.5	0.44	7.9	18.0	2.0
18	0.009	0.19	2.18	0.28	1.07	32.4	3.1	2.9	0.59	1.1	8.0	-2.1
24	0.007	0.22	4.58	0.27	1.08	29.4	5.9	2.7	0.6	1.4	9.2	-2.6
26	0.004	0.54	2.71	0.54	1.05	23.6	7.0	5.0	0.41	1.7	5.0	-1.2
27	0.003	0.52	2.29	0.31	1.11	37.5	1.5	1.8	0.36	0.9	3.9	-2.4
31	0.007	0.24	3.17	0.53	1.09	27.7	6.9	11.4	0.6	1.7	2.2	-1.4
33	0.001	0.15	1.55	0.39	1.10	1.5	54.5	8.4	0.31	10.4	4.8	2.2
38	0.003	0.18	2.60	0.90	1.11	19.0	18.6	11.8	0.44	2.2	4.5	0.5
44	0.003	0.41	2.00	0.26	1.04	17.8	28.2	4.3	0.48	4.1	20.3	0.2
45	0.003	0.57	2.25	0.44	1.15	5.3	28.5	11.1	0.43	2.4	7.5	1.3
46	0.008	0.89	1.62	0.40	1.09	62.3	2.4	0.9	0.55	0.9	5.7	-2.9
48	0.003	0.71	2.82	0.44	1.08	44.9	12.7	7.6	0.47	1.5	6.3	-1.4
52	0.006	0.59	0.64	0.53	1.09	39.4	30.3	20.9	0.69	28.0	15.5	1.5
54	0.005	0.53	3.02	0.28	1.07	57.8	13.9	4.8	0.54	1.9	4.7	-2.1
55	0.002	0.16	2.05	0.60	1.06	17.8	21.7	9.8	0.38	3.8	3.3	0.4
56	0.002	0.41	1.36	0.23	1.04	20.7	43.9	2.7	0.46	6.8	6.6	0.2
57	0.001	0.52	2.13	0.52	1.08	20.7	16.3	7.6	0.32	3.5	5.1	0.2
58	0.003	0.53	3.15	0.43	1.07	12.8	6.9	4.5	0.53	1.5	5.2	-1.4
60	0.005	0.60	2.32	0.46	1.10	14.5	26.0	9.2	0.44	0.8	4.5	-0.7
61	0.003	0.03	1.29	0.60	1.16	0.0	89.0	3.5	0.45	12.1	10.7	3.7
62	0.006	0.80	3.21	1.07	1.14	31.0	3.3	1.9	0.54	1.6	3.9	-1.3
63	0.004	0.14	2.46	0.52	1.13	9.9	4.0	1.6	0.45	1.0	5.3	-1.1
65	0.004	0.70	2.81	0.95	1.14	17.1	43.5	24.3	0.52	2.3	10.3	1.8
66	0.003	0.47	2.74	0.43	1.09	20.4	29.6	3.0	0.46	3.3	27.2	0.4
67	0.003	0.56	2.61	0.39	1.08	29.4	59.5	8.6	0.48	2.8	11.8	0.6
71	0.002	0.75	1.85	0.69	1.13	5.2	49.3	26.1	0.43	3.6	13.2	2.9
72	0.005	0.56	0.90	0.65	1.08	26.0	37.9	21.4	0.66	18.3	8.0	1.7
75	0.003	0.67	2.47	0.52	1.12	29.1	12.2	1.8	0.39	0.7	7.8	-1.7
76	0.001	0.49	0.74	0.57	1.10	14.2	62.5	10.2	0.3	18.2	8.9	2.7
80	0.005	2.24	3.14	0.50	1.09	5.3	16.3	7.3	0.49	1.8	13.6	-0.2
Mean	0.004	0.52	2.27	0.51	1.10	22.8	25.3	8.2	0.5	4.9	8.7	0.0
Min	0.001	0.03	0.64	0.23	1.04	0.0	1.5	0.9	0.3	0.7	2.2	-2.9
Max	0.009	2.24	4.58	1.07	1.16	62.3	89.0	26.1	0.7	28.0	27.2	3.7

Appendix E.3. Fish estimated richness (based on 3 consecutive 50-m reaches) and abundance (based on 3-pass removal) calculated using CAPTURE® for 30 study sites. Fish assemblage metrics include sensitives (SEN), fluvial specialists (FLU), lentic tolerants (LEN), and *Etheostoma scotti*.

Site ID	Total Est Rich	Total Est Abund	SEN Est Rich	SEN Est Abud	FLU Est Rich	FLU Est Abund	LEN Est Rich	LEN Est Abund	COS Est Rich	COS Est Abund	*E. scotti* Est. Abund
11	19	115.6	2	6.0	12	40.2	8	75.4	16	97.6	0.0
18	21	135.1	5	27.4	16	91.5	5	43.6	11	54.6	12.0
24	17	66.3	3	40.7	12	49.7	4	16.6	8	18.6	3.0
26	19	243.0	6	48.7	17	221.0	4	22.0	11	51.9	19.7
27	15	177.0	4	33.1	11	175.0	5	2.0	8	12.0	30.1
31	21	499.7	7	102.6	17	469.1	4	30.6	11	64.5	7.0
33	13	37.2	1	1.0	11	34.2	3	3.0	6	20.2	1.0
38	25	394.5	4	20.0	21	361.8	5	32.7	17	149.2	0.0
44	20	120.4	4	18.0	17	110.4	5	10.0	13	81.8	0.0
45	15	77.4	3	0.0	10	62.4	5	15.0	9	33.0	0.0
46	23	241.6	3	40.8	17	223.7	5	28.9	11	65.9	39.8
48	8	152.9	0	0.0	5	117.3	3	35.6	5	52.6	0.0
52	21	361.8	4	19.0	13	272.6	6	89.2	12	193.0	0.0
54	20	125.9	3	4.0	14	94.6	6	31.3	12	33.3	3.0
55	14	196.7	0	0.0	9	111.4	6	85.4	9	139.7	0.0
56	13	120.0	1	2.0	10	81.1	3	38.9	6	87.0	0.0
57	16	151.7	1	3.0	11	113.8	6	37.9	8	71.7	0.0
58	16	118.5	0	0.0	5	59.0	11	59.6	13	62.6	0.0
60	17	70.5	2	11.0	13	53.5	4	17.0	10	38.5	11.0
61	17	206.6	1	1.0	7	10.0	11	196.6	11	203.6	1.0
62	12	66.6	3	8.0	7	23.0	5	43.6	7	49.6	7.0
63	20	512.2	4	30.2	14	101.4	8	411.9	12	430.9	9.0
65	20	381.8	5	21.0	17	348.9	3	34.9	9	70.9	12.0
66	22	142.9	5	20.9	17	83.0	5	59.8	11	79.8	8.4
67	20	111.2	3	14.0	16	84.0	5	27.2	13	65.3	0.0
71	13	112.4	0	0.0	8	67.5	5	44.9	8	63.1	0.0
72	12	148.5	0	0.0	6	84.7	6	64.8	8	139.5	0.0
75	24	141.4	2	6.0	15	83.3	8	58.1	15	92.4	0.0
76	17	161.5	3	2.0	13	139.5	4	22.0	7	37.0	2.0
80	13	157.4	3	22.2	9	88.9	4	68.5	7	95.4	21.2
Mean	17.4	184.9	2.7	16.7	12.3	128.5	5.4	56.9	10.1	88.5	6.2
Min	8.0	37.2	0.0	0.0	5.0	10.0	3.0	2.0	5.0	12.0	0.0
Max	25.0	512.2	7.0	102.6	21.0	469.1	11.0	411.9	17.0	430.9	39.8