

Urbanization Effects on Fishes and Habitat Quality in a Southern Piedmont River Basin

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Abstract.—We quantified the relationships among urban land cover, fishes, and habitat quality to determine how fish assemblages respond to urbanization and if a habitat index can be used as an indirect measure of urban effects on stream ecosystems. We sampled 30 wadeable streams along an urban gradient (5–37% urban land cover) in the Etowah River basin, Georgia. Fish assemblages, sampled by electrofishing standardized stream reaches, were assessed using species richness, density, and species composition metrics. Habitat quality was scored using the Rapid Habitat Assessment Protocol (RHAP) of the U.S. Environmental Protection Agency. Urban land cover (including total, high-, and low-density urban) was estimated for the drainage basin above each reach. A previous study of these sites indicated that stream slope and basin area were strongly related to local variation in assemblage structure. We used multiple linear regression (MLR) analysis to account for this variation and isolate the urban effect on fishes. The MLR models indicated that urbanization lowered species richness and density and led to predictable changes in species composition. Darters and sculpin, cyprinids, and endemics declined along the urban gradient whereas centrarchids persisted and became the dominant group. The RHAP was not a suitable indicator of urban effects because RHAP-urban relationships were confounded by an overriding influence of stream slope on RHAP scores, and urban-related changes in fish assemblage structure preceded gross changes in stream habitat quality. Regression analysis indicated that urban effects on fishes accrue rapidly (<10 years) and are detectable at low levels (~5–10% urbanization). We predict that the decline of endemics and other species will continue and centrarchid-dominated streams will become more common as development proceeds within the Etowah basin.

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Introduction

Protecting stream resources from human impacts increasingly depends on understanding the linkages between urban land use and stream systems. Relatively low levels of basin urbanization (e.g., 10–20%) cause major changes in stream hydrology, geomorphology, water quality, and stream communities (Baer and Pringle 2000; Paul and Meyer 2001). Urban land cover, or associated variables such as impervious surface area, is linked to declines in fish richness, diversity, density, and biomass as well as changes in population structure of fishes and trophic structure of assemblages (Klein 1979; Scott et al. 1986; Lenat and Crawford 1994; Weaver and Garman 1994; Yoder and Rankin 1996). Urbanization is also associated with declines in biotic integrity, with increases in tolerant and exotic taxa, and with decline or extirpation of sensitive species (Wang et al. 1997; Boet et al. 1999; Onorato et al. 2000; Wang et al. 2000; Wolter et al. 2000).

In this study, we investigated the effects of urbanization on stream fishes in a southern Piedmont drainage characterized by exceptional species endemism as well as by local variation in assemblage composition. The southern Piedmont is among the most rapidly developing areas of the United States (USDA 2000), and increasing urbanization will likely impact fish assemblages. Although biotic integrity is known to decrease with urban sprawl (e.g., Steedman 1988; Wang et al. 1997), it is less clear if specific taxa (such as endemic species) vary predictably with increasing urban land cover. In addition, Walters et al. (2003a) found that reach-level variation in assemblage structure was strongly linked to stream slope and associated benthic habitat variables in this Piedmont system. This led us to investigate whether an effect of increasing urban land cover on fishes would be detectable given spatial variation in fish assemblages driven by geomorphology. Spatial variation is a common feature of stream communities (Allan 1995); however, most prior studies of urbanization effects on fishes have been limited to gradients in single streams or comparisons between urban and reference drainage basins (Paul and Meyer 2001). Studies of urban gradients and fishes in multiple basins are rare and have mostly focused on agricultural-dominated areas of the upper Midwest (e.g., Steedman 1988; Wang et al. 1997, 2000). Comparable studies are lacking for the southern Piedmont.

Linking land use change with stream communities and habitat is a critical step in aquatic resource management (Jacobson et al. 2001). If habitat quality changes predictably with urban land cover and fish

assemblages shift in response to habitat quality, then habitat quality assessment can be used as an indirect measure of urban effects on stream biota. However, studies in the Midwest have found that habitat quality is a suitable indicator of agricultural impacts but poorly reflects urban effects on stream biota (Roth et al. 1996; Wang et al. 1997, 2001). If urbanization effects on biota precede gross structural changes to streams, then we could expect to find streams in urbanizing catchments with altered biotic assemblages that also appear to have high habitat quality. Conversely, strong correlations among fish assemblage structure, urban land cover, and stream habitat quality would support the use of habitat indices in streams draining urbanizing catchments.

Temporal components of land cover change on streams have received less attention than spatial components (Allan and Johnson 1997; Harding et al. 1998). Streams clearly respond to urbanization, but the timing of the response is poorly understood and basic questions remain unanswered. It is unknown whether fishes respond to disturbance during the initial development phase (e.g., increased sediment loading) or to the chronic, long-term effects of increased urban cover (e.g., altered hydrology and poor water quality). If urban effects accrue rapidly, then fish assemblages in recently urbanized catchments should exhibit changes in structure comparable to those with a longer history of equivalent levels of urbanization. Understanding how quickly streams respond is important for managing aquatic resources in urbanizing landscapes.

Urbanizing rural catchments in the southern Piedmont present a unique setting to examine the complex spatial and temporal aspects of urban impacts on aquatic systems. This study addresses the following questions: (1) How do fish assemblages change along an urban gradient?, (2) Does a widely-used measure of habitat quality reflect urbanization effects on fish assemblages?, and (3) How quickly do fishes respond to urban development?

Study Area

This study was conducted in wadeable tributaries in the Piedmont province of the Etowah River basin, which forms part of the upper Coosa River drainage. Portions of the Etowah basin lie in the Blue Ridge, Ridge and Valley, and Piedmont physiographic provinces of the southern Appalachian highlands (Figure 1). The southern Appalachian highlands are a center of stream fish biodiversity and endemism (Williams et

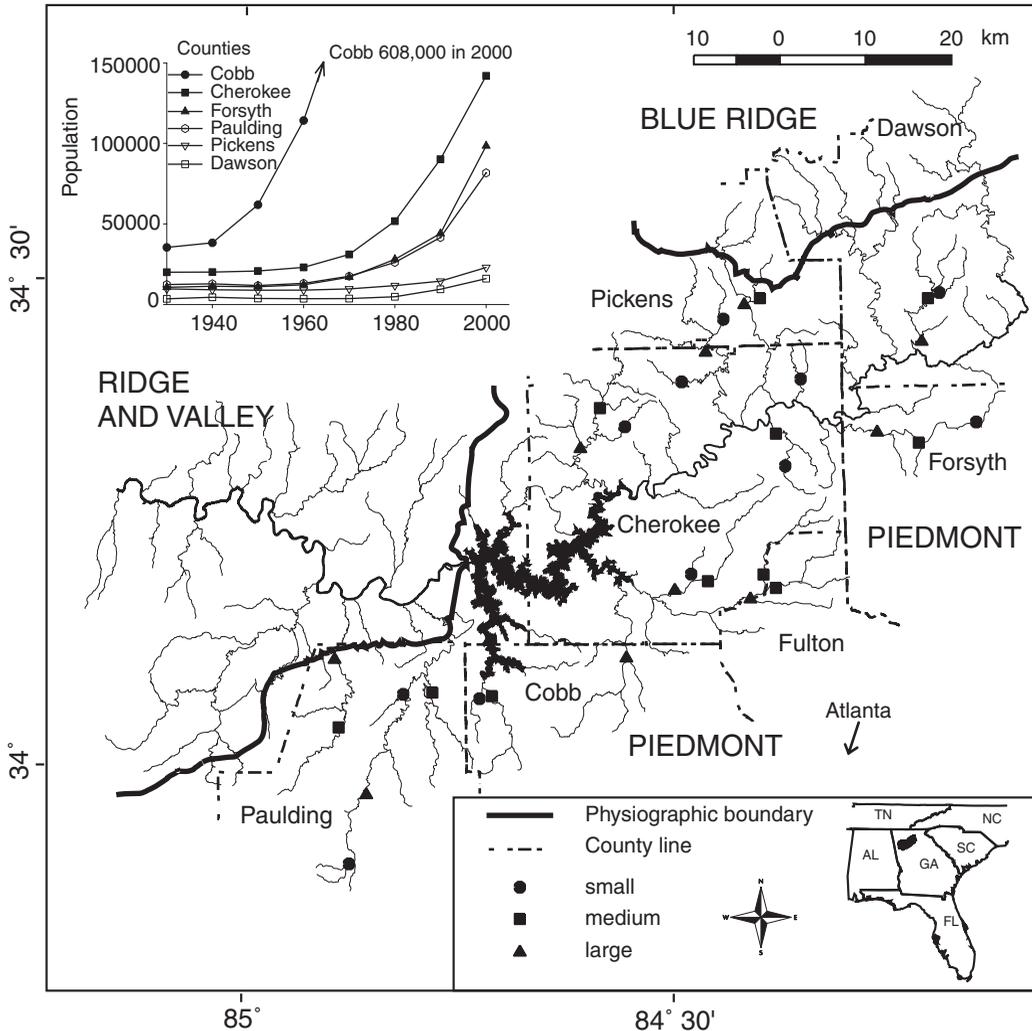


FIGURE 1. Map of sample sites in the Etowah River basin. The inset graph shows population growth 1930–2000 (U.S. Census Office 1930–2000) for counties in the study area except for Fulton County, which was strongly influenced by the population of Atlanta. Population growth in Fulton County basins in this study is similar to the trend observed for neighboring Cherokee County.

al. 1992; Warren and Burr 1994; Warren et al. 2000), and Burkhead et al. (1997) estimated that 91 species from 18 families are native to the Etowah system. The most diverse families are Cyprinidae (31 species), Percidae (19 species), and Centrarchidae (13 species). Warren et al. (1997) classified 16 species of the Tallapoosa and Coosa River systems as endemic, a level of endemic species richness unsurpassed in southeastern river systems.

Land cover in the Etowah basin is a mixture of forest, agriculture, and urban development (Leigh et al. 2001). Agricultural land use is primarily pasture

for hay production and grazing, and row crops are a minor component. Recent urban development in the basin is driven by population increases fueled by the spread of metropolitan Atlanta. Several counties have experienced exponential population growth that began in the 1970s and 1980s. The southern most counties near Atlanta show the greatest population increases (Figure 1, inset graph), and two of these, Forsyth and Paulding, were among the 10 fastest growing counties in the United States in 1997 (USCB 1998).

We randomly selected 30 Etowah basin streams ranging from 0% to 37% urban land cover. Streams

were stratified into small, medium, and large basin size groups of approximately 15, 50, and 100 km² ($\pm 26\%$) (Figure 1, Appendix 1). All sample reaches are in the Piedmont, but a few of the catchments have headwaters in the Blue Ridge.

Methods

Fish Sampling and Assemblage Variables

Fishes were sampled during summer base flow conditions in 1999 and 2000. Reaches of 200, 300, and 400 m were sampled in small, medium, and large streams, respectively. These reaches were approximately 40 times the average base flow wetted channel width within each stream size-group, a reach slightly longer than the 35 times stream width recommended by Simonson and Lyons (1995) to assess fish assemblages in wadeable streams. Reaches were sampled in a single pass (Simon and Lyons 1995) using a crew of four to six persons equipped with a backpack electric shocker, seine, and dip nets. Reaches were divided into two sections of equal length. The first half of the reach (i.e., 20 times mean width) was sampled to obtain quantitative catch data for calculating relative abundances (Angermeier

and Smogor 1995). Large individuals (>20 cm) and fishes with protected status were counted and released. All other fishes were euthanized and preserved for laboratory identification. The second half of the reach was sampled to improve our estimate of site species richness, and we retained voucher specimens of species not observed in the first half of the reach.

Assemblage variables used for analyses included richness and density as well as the relative contribution of selected taxonomic groups (Table 1). Centrarchids, darters, and cyprinids were selected as the primary taxonomic groups because they account for more than 70% of the species we collected, and these account for much of the among-site variance in species composition in these streams (Walters et al. 2003a). Sculpin were included with darters because they utilize similar benthic habitats and prey (Etnier and Starnes 1993). We also included insectivorous cyprinids, a subgroup of cyprinids commonly used as a positive indicator in biotic indices (Miller et al. 1988), based on diet information in Etnier and Starnes (1993) (Appendix 2). In addition to these major species groups, we included a category for endemic species, those fishes distributed primarily in the upper Alabama River basin above the Fall Line (Appendix 2).

TABLE 1. Fish assemblage variables used in correlation and regression analysis. Scatterplots and trial and error were used to identify the best transformations for richness and density (Ott 1988).

Fish assemblage variable	Description	Transformation
Richness ^a	total species	$\log_{10}(x)$
Density ^b	fishes/m ²	$x^{0.25}$
Number of species ^a		
darters and sculpin		none
centrarchids		none
cyprinids		none
insectivorous cyprinids		none
endemic species		none
Proportion of species ^a		
darters and sculpin	# (darter and sculpin spp.)/richness	none
centrarchids	# centrarchid spp./richness	none
cyprinids	# cyprinid spp./richness	none
insectivorous cyprinids	# insectivorous cyprinid spp./richness	none
endemic species	# endemic spp./richness	none
Relative abundance ^b		
darters and sculpin	abundance/total catch	none
centrarchids	abundance/total catch	arc-sine(square-root (x))
cyprinids	abundance/total catch	none
insectivorous cyprinids	abundance/total catch	none
endemic species	abundance/total catch	none (unable to normalize)

^a Calculated from total reach (40× stream width).

^b Calculated from first half of reach (20× stream width). See Methods for details.

We designated endemics based on distributional information provided by Mettee et al. (1996). All endemics were either darters or cyprinids except for *Cottus carolinae zopherus*, a taxonomically distinct form of banded sculpin endemic to the upper Coosa system (Etnier and Starnes 1993).

The contribution of each species group to the population was calculated as the number of species, the proportion of total richness, and the proportion of total catch (Table 1). Measures of proportional richness are not commonly used in assemblage studies. We used these variables because they can identify groups that persist relative to other species and they can identify patterns in species composition even if overall richness is similar among sites. Numbers of species and proportional richness were derived from species lists compiled from the entire sampled reach (i.e., 40 times stream width). Relative abundance was calculated using catch data from the first half of each reach. Relative abundance of darters was calculated without blackbanded darter *Percina nigrofasciata*, a species that is widespread and locally common in the Etowah system (Walters et al. 2003a).

Stream and Catchment Characteristics

Stream slope and basin area were the only geomorphic variables compiled for this study. We included these variables in this study because a prior multivariate analysis of these streams identified slope and basin area (a surrogate for stream size) as the primary predictors of fish assemblages (Walters et al. 2003a). Stream slope was surveyed with an electronic total station over reaches scaled to approximately 20 times the average base flow width for streams in each size-class (i.e., 100-, 150-, and 200-m reaches). Slope was measured as the average gradient through the tops of riffles. ArcView software was used to calculate area and percentage of urban land cover for catchments. Catchment boundaries upstream of sample sites were delineated on digital raster graphics of the latest 7.5 minute USGS quadrangles.

We used the USEPA Rapid Habitat Assessment Protocol (RHAP) to assess stream habitat quality (Barbour et al. 1999). The RHAP uses 10 metrics to assess the condition of various stream habitat parameters, including benthic habitat quality, channel and bank condition, instream habitat heterogeneity, and riparian vegetation. Each metric is scored on a continuous scale ranging from 0 to 20 based on a visual assessment of the habitat feature. Metric scores were averaged for an overall RHAP score. We assessed habi-

tat metrics over the entire reach sampled for fishes. The RHAP was usually calculated the same day as fish sampling and was always evaluated at base flow to allow comparisons among all sites. Two to four investigators completed RHAP at each site, and final scores were based on the mean of multiple assessments.

Land cover data were derived from Landsat TM images from June 1987 and July 1997 (Lo and Yang 2000). The images have 25-m resolution and overall land cover classification accuracy of about 90%, based on visual interpretation of color infrared aerial photographs (Lo and Yang 2000). Two urban land cover types were classified. High-density urban (HDU) has approximately 80–100% ground coverage with construction material and includes commercial buildings, parking lots, residential development within city cores, and multi-lane highways. Low-density urban (LDU) is roughly 50–80% construction material and is characterized by single or multiple family housing developments and two-lane roads. The LDU and HDU were summed to calculate the total basin percent coverage of urban land cover (U). Totals from 1987 were subtracted from 1997 totals to calculate the percentage of basin area converted during the decade. Three rural basins had slightly less (e.g., <2%) urban land cover in 1997 than 1987. We attributed this decline to small error in the accuracy of the data rather than actual loss of urban land. For these catchments, we assigned a value of 0 for 1987–1997 change. We limited our spatial scale to land cover of the entire catchment to assess the total impact of urbanization within a basin (Wang et al. 1997).

Data Analysis

Normality of all variables was checked using the Kolmogorov-Smirnov test using SigmaStat 4.0 and transformations were applied if needed (Table 1). Land cover percentages were converted to decimal proportions and transformed using arc-sine (square-root (x)) for correlation analysis (Sokal and Rohlf 1995). Untransformed land cover data were used in some scatter plots to illustrate trends and identify possible thresholds in the response of dependent variables. Land cover variables covary to some extent because as one element increases, others decline (Cain et al. 1997). The urban variables used in this study suffer from a lack of independence. For instance, all 1987 urban cover is contained in the 1997 urban cover because urbanized land seldom reverts to agricultural or forested cover. We used correlation analysis to quantify colinearity among land cover variables.

Relationships among fishes, land cover, stream slope, basin area, and habitat quality were quantified using correlation analysis, linear regression, and nonlinear regression. We report actual probability values for tests, unadjusted for potential inflation of type I error rate resulting from multiple analyses derived from a single dataset. If scatter plots indicated nonlinear relationships between variables, curves were fit to the data using nonlinear equations derived from Sigma Plot 4.0. Scatterplots were evaluated to identify categories of nonlinear equations (e.g., polynomial and sigmoidal) that provided the most realistic fit to the data. Equations were applied to the data and were evaluated based on minimizing sum of squares error, maximizing r^2 , and homoscedasticity of residuals. If multiple equations provided similar results for these criteria, the simplest model was selected.

We used forward stepwise multiple regression analysis to build predictive models for proportional richness, relative abundance, and RHAP. This procedure is recommended for exploratory analyses of large data sets (Neter et al. 1996) and allowed us to partition the variance in dependent variables relative to geomorphology and urban land cover. We set $P = 0.1$ to enter or remove variables. Variables were not included in the models unless they were significant at $P = 0.05$ and explained more than 4% of the variance in dependent variables. The analyses were conducted using JMP 5.0 (Freund et al. 2003). We transformed slope using $\log_{10}(x)$, a robust and widely used transformation for nonlinear data (Ott 1988), in multiple linear regression analysis. Other nonlinear equations provided a better fit for some dependent variables, but these transformations (e.g.,

sigmoidal) were variable specific and complicated comparisons among models (Walters 2002). Additionally, geomorphic analysis at these sites showed that \log_{10} was an appropriate transformation for linking stream slope with other physical variables such as particle size distribution (Walters et al. 2003a).

Results

Land Cover Characteristics

The LDU accounted for about 87% of urban land cover (Figure 2). Mean U across basins nearly doubled from around 8% to 15% from 1987 to 1997. Mean HDU only increased from 1.1% to 1.7% over the decade, so most of the increase in U resulted from low-density housing development. Land cover variables demonstrated a high degree of colinearity ($r = 0.70$ – 0.99 , Table 2), and correlation was highest among categories within a single year ($r = 0.85$ – 0.99). Correlations of 1987 and 1997 land cover with the 1987–1997% urban change were weaker, although all correlations with 1997 land cover were significant (Table 2). Basin area and slope were generally uncorrelated with urban cover except for a weak negative correlation between slope and 1987–1997 LDU.

Correlations of Fishes and RHAP with Urban Land Cover, Basin Area, and Slope

Assemblage variables were significantly correlated with urban land cover, RHAP, and geomorphic variables (Figure 3). Urban land cover (1997 U) was unrelated

TABLE 2. Correlation matrix (Pearson's r) of independent variables. Land cover abbreviations: U, urban; HDU, high-density urban; LDU, low-density urban. All land-cover variables are arc-sine (square-root (x)) transformed percentages. Correlations significant at $P < 0.05$ (uncorrected for Type I error rate) are in bold; $P < 0.01$ are in bold and italics; $P < 0.001$ are in bold, italics, and underlined ($n = 30$ sites).

	Basin area (km ²)	\log_{10} slope	1987%			1997%			1987–1997%			
			U	HDU	LDU	U	HDU	LDU	U	HDU	LDU	
\log_{10} slope	-0.37											
1987 U	-0.10	0										
1987 HDU	-0.18	0.10	<i><u>0.90</u></i>									
1987 LDU	-0.07	-0.02	<i><u>0.99</u></i>	<i><u>0.86</u></i>								
1997 U	-0.11	-0.17	<i><u>0.85</u></i>	<i><u>0.75</u></i>	<i><u>0.85</u></i>							
1997 HDU	-0.22	0.04	<i><u>0.82</u></i>	<i><u>0.86</u></i>	<i><u>0.78</u></i>	<i><u>0.84</u></i>						
1997 LDU	-0.08	-0.20	<i><u>0.83</u></i>	<i><u>0.70</u></i>	<i><u>0.83</u></i>	<i><u>0.99</u></i>	<i><u>0.78</u></i>					
1987–1997 U	-0.04	-0.35	<i><u>0.36</u></i>	0.28	<i><u>0.38</u></i>	<i><u>0.79</u></i>	<i><u>0.50</u></i>	<i><u>0.82</u></i>				
1987–1997 HDU	-0.11	-0.03	0.35	0.31	0.34	<i><u>0.58</u></i>	<i><u>0.74</u></i>	<i><u>0.53</u></i>	<i><u>0.56</u></i>			
1987–1997 LDU	-0.04	-0.37	0.35	0.26	0.36	<i><u>0.77</u></i>	<i><u>0.43</u></i>	<i><u>0.81</u></i>	<i><u>0.99</u></i>	<i><u>0.46</u></i>		
RHAP	-0.08	<i><u>0.66</u></i>	-0.47	-0.42	-0.47	-0.56	-0.48	-0.56	-0.47	-0.31	-0.46	

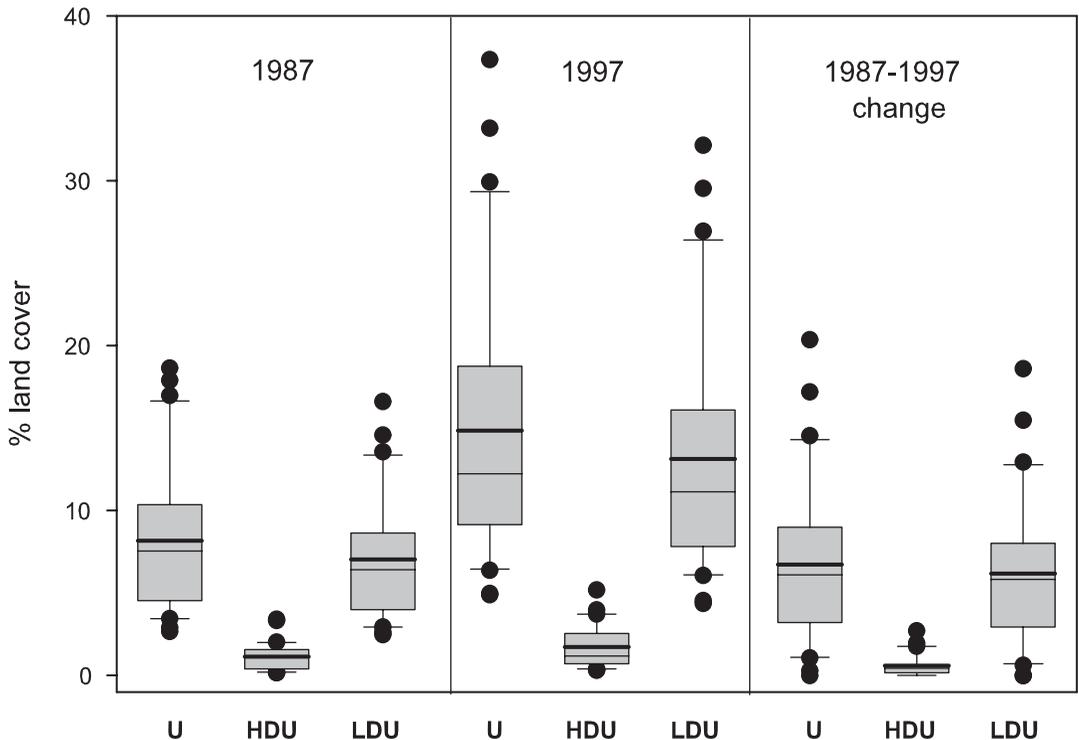


FIGURE 2. Box and whisker plots of total urban (U), high-density urban (HDU), and low-density urban (LDU) for basins upstream of 30 sample reaches in Etowah River tributaries. Top and bottom boundaries of the box indicate 75th and 25th percentiles, respectively. Within the boxes, thin lines indicate the median and thick lines indicate the mean. Whiskers above and below the box indicate the 90th and 10th percentiles.

to richness and density but was significantly related to changes in species composition. Centrarchids increased with 1997 U, whereas measures (e.g., relative abundance) of all other species groups showed significant declines. Correlations between fishes and RHAP showed the opposite trend (Figure 3). All groups except centrarchids were positively correlated with habitat quality. Assemblage structure also varied with stream geomorphology. Changes in species composition with slope showed a similar pattern to those for RHAP scores. All groups increased with stream slope except for centrarchids. Basin area, a surrogate for stream size, correlated positively with richness and negatively with density but with few of the species composition variables.

The patterns in species composition identified by correlation analysis suggested that the effects of urbanization, habitat quality, and stream slope were interrelated. To better illustrate these relationships we selected percentage endemic species, the fish variable best predicted by independent variables, for further analysis. Scatterplots of untransformed data show significant, nonlinear declines of percentage endemics

and RHAP with urban land cover (Figure 4A, B). Because percentage endemics were strongly associated with RHAP ($r = 0.84$, Figure 3), this suggested that RHAP would be a suitable indicator of urban effects on fishes. However, stream slope was an even better predictor of percentage endemics and RHAP (Figure 4C, D). Both variables demonstrated a strong positive sigmoidal response with slope and a predictable coarsening of the stream bed.

Multiple Linear Regression Models of Fishes and Habitat Quality

We used multiple linear regression analysis to test the hypothesis that urban land cover explained variance remaining after accounting for effects of slope and/or basin area on fish assemblage variables and RHAP scores. Drainage area and slope were the primary predictors of RHAP and most fish assemblage variables (Table 3). After accounting for variation attributable to stream slope and basin area, urban land cover was significantly related to all of the dependent variables

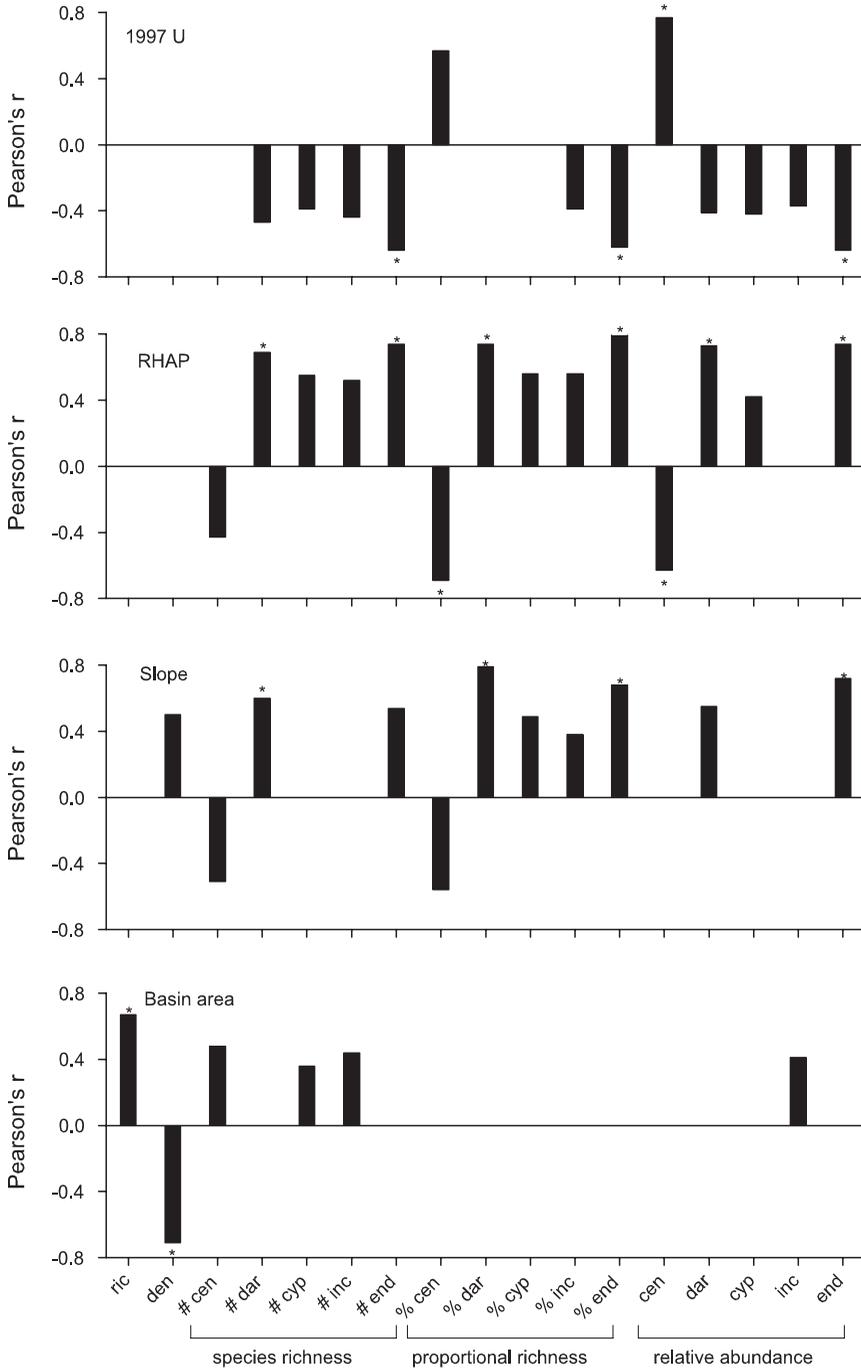


FIGURE 3. Bar charts depicting significant correlations (Pearson's r) of fish assemblage variables and 1997 urban land cover, RHAP, stream slope, and basin area ($n = 30$ sites). Values for relative abundance of endemics are Spearman's r because this variable was nonnormal. Bars are not shown for nonsignificant correlations ($|r| < 0.36$). Due to spatial and temporal colinearity among urban variables, only results for 1997 U are shown. Other urban variables showed an identical trend with slight variation in r values (Walters 2002). Asterisks indicate $P < 0.0001$. P -values are uncorrected for multiple comparisons and are shown as a rough guide to correlation strength. Variable abbreviations: ric, richness; den, density; cen, centrarchids; dar, darters and sculpin; cyp, cyprinids; inc, insectivorous cyprinids; end, endemics.

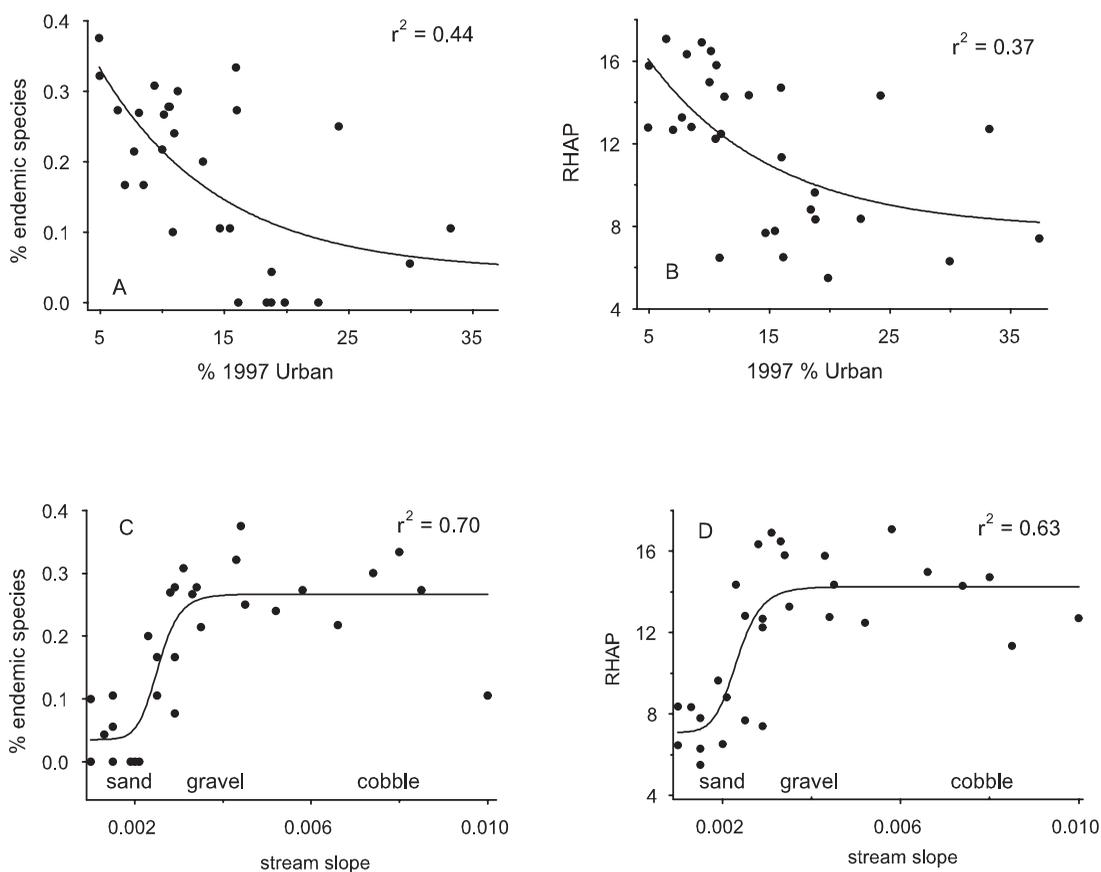


FIGURE 4. Scatterplots showing the relationships of percentage endemic species and RHAP with 1997 U (A and B) and stream slope (C and D) ($n = 30$ sites). Slope explains about 85% of the variance in mean particle size in these streams (Walters et al. 2003a), so the increase in percentage endemics and RHAP also corresponded with a predictable coarsening of the streambed. Particle size-classes corresponding with mean particle size for streams of a given slope are labeled on the x -axis for C and D. Nonlinear curves were fit to the data using Sigma Plot 4.0. Three parameter exponential decay models were fit to 1997 urban land cover data and four parameter sigmoidal models were fit to slope data. (A) $y = 0.044 + 0.481e^{(-0.103 \cdot \%1997 \text{ urban})}$;

(B) $y = 7.832 + 13.249e^{(-0.0962 \cdot \%1997 \text{ urban})}$; (C) $y = 0.0356 + 0.231 \cdot \text{slope}^{\frac{10.492}{0.0025^{10.492} + \text{slope}^{10.492}}}$; (D) $y = 7.097 + 7.158 \cdot \text{slope}^{\frac{8.591}{0.0023^{8.591} + \text{slope}^{8.591}}}$.

except relative abundance of insectivorous cyprinids. Centrarchids increased with urban land cover whereas RHAP, richness, density, darters and sculpin, endemics, cyprinids, and insectivorous cyprinids declined. Models for cyprinids and insectivorous cyprinids were the weakest among species groups, and urban land was unrelated to relative abundance of insectivorous cyprinids. The high-density urban variable was selected in 7 of the 11 models that included land cover.

Urbanization was most strongly associated with increasing centrarchid relative abundance and declining proportional richness of endemics (Figure 3). We plotted the residuals from the linear regression of these variables against \log_{10} slope (independent variable) and

1997 U to illustrate spatial and temporal trends in the response of these species groups (Figure 5A, B). Catchments were categorized as having more than 10% U in 1987, less than 10% U in 1987, but more than 10% in 1997 and less than 10% U in 1997. Catchments with more than 10% U in 1987 tended to have the highest urban cover, indicating that the most heavily urbanized catchments have also been urbanized for the longest time. The largest residuals in both models were from the most developed basins. At levels greater than 15% U, the majority of residuals from the centrarchid model were positive and all but three of the residuals from the endemic model were negative. Thus, for basins with more than 15% U,

Table 3. Multiple linear regression models of RHAP and fish variables. Transformations for dependent variables are given in Table 1. Land cover variables were transformed by arc-sine (square-root(x)). Only 1997 urban and 1987–1997 land cover change variables were used due to strong colinearity between the 1987 and 1997 datasets. The relative abundance of endemics could not be normalized and is not included in this analysis. F -values are given for the whole model; P -values are uncorrected for multiple comparisons and shown as a rough guide to correlation strength ($n = 30$ sites and eight predictor variables).

Independent variable	Variables in model	Trend	Cumulative r^2	P	F
RHAP	\log_{10} slope	+	0.43	<0.001	28.55
	97 HDU	–	0.68	<0.001	
richness	basin area	+	0.45	<0.001	14.96
	\log_{10} slope	+	0.59	0.02	
	97 HDU	–	0.63	0.03	
density	basin area	–	0.50	<0.001	25.29
	97 U	–	0.65	0.002	
Proportion of species					
darters and sculpin	\log_{10} slope	+	0.63	0.001	28.80
	97 HDU	–	0.68	0.02	
centrarchids	\log_{10} slope ^a	–	0.31	0.001	16.48
	97 U	+	0.54	<0.001	
cyprinids	\log_{10} slope	+	0.24	0.006	7.73
	87–97 HDU	–	0.35	0.04	
insectivorous cyprinids	\log_{10} slope ^a	+	0.15	0.04	6.29
	97 HDU	–	0.32	0.009	
	basin area	+	0.44	0.03	
endemics	\log_{10} slope	+	0.46	<0.001	36.09
	97 HDU	–	0.73	<0.001	
Relative abundance					
darters and sculpin	\log_{10} slope	+	0.30	0.04	9.35
	97 LDU	–	0.41	0.002	
centrarchids	\log_{10} slope ^a	–	0.08	0.12	20.00
	97 U	+	0.63	<0.001	
	87–97 HDU	+	0.70	0.02	
cyprinids	87–97 HDU	–	0.2	0.01	7.05
insectivorous cyprinids	basin area	+	0.017	0.26	5.56

^a \log_{10} slope manually entered into because scatterplots of \log_{10} slope versus these variables showed clear relationships (D.M.W., unpublished data).

observed centrarchid relative abundance is usually higher and the proportion of endemics is consistently lower than predicted by slope.

Urbanization led to higher relative abundance of centrarchids even in steep streams with relatively high habitat quality. Pie charts (Figure 5C) compared the relative abundance of fishes from two sites of similar size, slope, and RHAP scores but different levels of urban cover. Site 111 had the steepest slope we observed (0.01) and plots as a positive residual in Figure 5B. Slope accurately predicted relative abundance of centrarchids we observed at site 20 (slope = 0.007). Centrarchids were 59.2% of the catch at site 111 in contrast to the general trend for lower centrarchids abundance at high slope streams.

Discussion

Fish Assemblage Response to Basin Urbanization

Our results showed that urbanization effects on fishes are detectable even in systems with strong geomorphic controls on fish assemblages. Stream size predicted richness and density, whereas species composition changed along a slope gradient from darter, sculpin, cyprinid complexes characterized by a high degree of endemism to assemblages dominated by centrarchids. As urban land cover increased, richness and density declined, and centrarchids became the dominant group as other species declined or were locally extirpated. By

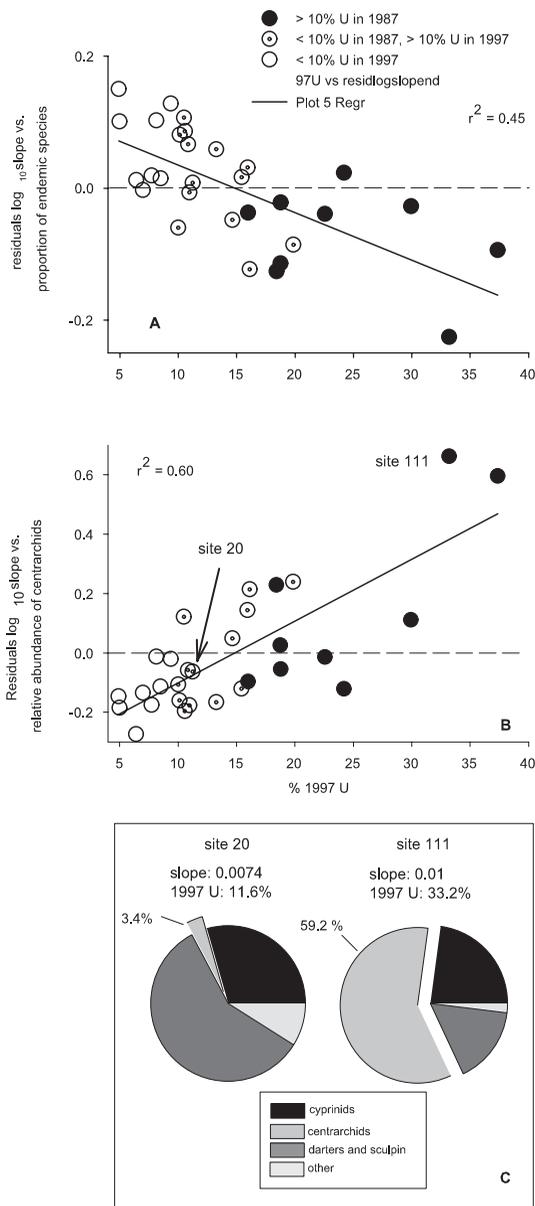


Figure 5. Linear regression of percentage 1997 U versus residuals from slope models of the proportional richness of endemics (A) and the relative abundance of centrarchids (B) ($n = 30$ sites). Basins are coded based on 1987 and 1997 urban land cover data. Panel C compares the relative abundance of major taxonomic groups and characteristics of two sites identified in (B).

disrupting geomorphic control of among-site differences in fishes, urban development homogenizes (sensu McKinney and Lockwood 2001) fish faunas at the broader scale of the Etowah basin (Walters et al.

2003b). Observed assemblages are characteristic of what we would expect in smaller, low slope streams (i.e., low richness, low endemism, and centrarchid dominance).

Darters and sculpin and endemics as a group declined with increasing urban land cover. Similar results were reported for darters in urban streams in Alabama (Onorato et al. 2000) and for the response of endemics to catchment disturbance in southern Appalachia (Scott and Helfman 2001). Darters are considered to be sensitive to disturbance (Kuehne and Barbour 1984; Jenkins and Burkhead 1994), and endemics suffer a higher degree of imperilment and extinction risk than other taxa (Meffe and Carrol 1994; Angermeier 1995). All metrics related to darters and sculpin increased with stream slope. Six of these species were included in the endemic group (Appendix 2), and the endemics showed a similar trend. These species likely increase with slope due to their dependence on coarse stream beds commonly associated with high quality benthic habitat. All of these species demonstrate a high degree of benthic specialization (Etnier and Starnes 1993; Jenkins and Burkhead 1994; Burkhead et al. 1997). They spawn on or in coarse particles, feed benthically or primarily on benthic macroinvertebrates, and are morphologically adapted (e.g., reduced air bladder, large pectoral fins, or dorso-ventral compression, Helfman et al. 1997) for high-velocity, riffle-run habitats prevalent in steep streams.

Centrarchid relative abundance and proportional richness increased with urban land cover. We analyzed data published by Weaver and Garman (1994, Table 2) for six reaches in a Piedmont stream in Virginia and found the same trend. According to these data, centrarchid proportional richness increased from 0.24 to 0.41 and relative abundance almost doubled from 0.25 to 0.48 following urban development. These increases in relative abundance occurred even though absolute abundance was lower. As in the Etowah streams, these increases occurred because centrarchids persisted in urban streams whereas other fishes declined more precipitously or were extirpated altogether. Even though centrarchid species richness was not correlated with urban land cover, all 11 species except for longear sunfish *Lepomis megalotis* were collected in streams with more than 20% urban land cover.

Our findings contribute to a growing body of evidence that centrarchids are more resilient to disturbance than other stream fishes (Detenbeck et al. 1992). Centrarchids increase with turbidity, bed sedimentation, riparian deforestation, and agricultural land cover in other southeastern river systems (Jones et al. 1999;

Meyer et al. 1999; Walser and Bart 1999). Waite and Carpenter (2000) also found that introduced populations of centrarchids increased along disturbance gradients in the Willamette River, Oregon. Centrarchid species richness was used as a positive indicator of stream integrity in the original index of biotic integrity and in subsequent regional variations (Karr 1981; Miller et al. 1988; Shaner 1999; Schleiger 2000). Our results, together with the studies reviewed above, suggest that the use of centrarchids as a positive indicator may be inappropriate for measures of stream health.

Cyprinids and insectivorous cyprinids were the least correlated with geomorphic and urban land cover gradients (i.e., multiple linear regression models $r^2 = 0.20\text{--}0.34$). Models for these groups may be weak because these fishes respond to variables we did not consider in this analysis (e.g., water chemistry and predation pressure) or because different cyprinid species exhibit too much ecological variability to model effectively as a group. Based on life history information provided in Jenkins and Burkhead (1994) and Etnier and Starnes (1993), the cyprinids we collected belong to six spawning guilds, three feeding guilds, and prefer habitats ranging from pools to riffles (Walters et al. 2003a). We developed much stronger models for centrarchids, darters and sculpin, and endemics, and these species have greater within-group ecological similarity than cyprinids (Walters et al. 2003a).

Numerous studies document declines in fish richness, abundance, and biotic integrity with urbanization (e.g., Lenat and Crawford 1994; Onorato et al. 2000; Wang et al. 2000). After accounting for changes in richness associated with geomorphic controls, urban land cover only explained about 5% of the total variance in richness among sites. These results support the findings of Scott and Helfman (2001) that human disturbance of southern Appalachian streams may lead to minor changes in species richness but major changes in species composition. Measures of fish assemblage health such as the index of biotic integrity (IBI, Karr 1981) are designed to account for variability associated with stream size. Our results indicate that stream slope may also warrant consideration for species composition metrics, at least in wadeable Piedmont streams.

Habitat Quality as an Indicator of Urban Effects

The RHAP scores were not a suitable indicator of urban impacts because significant changes in fish assemblages precede gross changes in habitat quality. This finding supports the hypothesis of Wang et al. (1997)

that communities are more sensitive to urban impacts related to altered water quality and hydrology than those related to habitat alteration. The disconnection between urbanization, fishes, and RHAP was most apparent in high-slope streams that had relatively high habitat quality but drastically altered species composition. Wang et al. (1997) also showed that fish-urban-habitat quality relationships were confounded in high gradient, rocky streams in Wisconsin. In their case, high slope streams had higher habitat and biotic integrity than predicted by urban land cover. The inability of habitat quality indices to serve as an indicator of urban impacts in our Piedmont streams is similar to results published for streams in the upper Midwest (Roth et al. 1996; Wang et al. 1997, 2001).

Our regression analyses indicated that RHAP scores did decline with urban land cover, but urban effects were subtle compared with the overriding influence of stream slope. The strong correspondence between slope and RHAP likely exists because slope is a primary determinant of benthic habitat variation at these sites (Walters et al. 2003a) and RHAP includes several metrics to assess benthic habitat quality (e.g., epifaunal substrate, sediment deposition, and embeddedness; Barbour et al. 1999). Because steep streams naturally score higher, RHAP will likely underestimate human disturbance in these systems. We documented this in a high-slope, urbanized stream (site 111). Rapid Habitat Assessment Protocol scores remained high even though the fish assemblage showed a clear shift to centrarchid dominance. Alternatively, steep streams may be more resilient to some types of disturbance (e.g., excessive sediment inputs) and maintain higher quality habitats relative to low slope streams (Wang et al. 1997). The range of urban land cover in our study was 5–37%, and most of this land was converted within the last 20 years. We predict that more obvious changes in stream habitat (e.g., severe bank erosion and channel incision) will be prevalent as urban land cover increases or as catchments are urbanized for longer time periods.

Temporal and Spatial Aspects of Fish Assemblage Response

We were unable to statistically isolate temporal changes in the fish assemblage due to colinearity among urban land cover variables. However, our results did show that fishes responded to low levels of urbanization, and we can infer that the response was rapid (i.e., <10 years) given the rate of development in Etowah River system. For example, the residual analysis showed that

effects were most pronounced for basins that (1) had more than 10% U for at least 10 years, or (2) had more than 15% U. In 1987, only three of the study basins had more than 15% U. By 1997, 13 basins exceeded this level, and the mean across basins was 15%. Our results are consistent with other studies that found that stream ecosystems respond strongly to low levels of urban land cover or impervious surface coverage (Klein 1979; Booth and Jackson 1997; Wang et al. 2001; and studies reviewed by Paul and Meyer 2001). The pace of urban development has not slowed since 1997, and most of these basins will surely surpass 15% U within the next decade. Given this scenario, we predict that the decline of endemics and other species groups will become more acute and that centrarchid-dominated streams will become the norm within the Etowah basin.

Collinearity among urban variables also hindered our ability to isolate the relative impacts of HDU and LDU on the Etowah streams. For example, LDU was the dominant form of urban land cover in these catchments but was statistically indistinguishable from the total U. However, HDU variables were selected most frequently by the forward stepwise procedure even though HDU accounted for a small fraction of total U. This suggests that HDU may have a disproportionate impact on stream systems compared with LDU. High-density urban has a higher proportion of impervious surface coverage, a key element of the urban landscape that contributes to stream degradation (Klein 1979; Arnold and Gibbons 1996; Booth and Jackson 1997). High- and low-density urban land cover may affect stream systems differently, but we could not isolate these effects at the basin scale.

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Appendix 1. Characteristics of study streams and catchments in the Etowah River basin. Attributes shown include scores for Rapid Habitat Assessment Protocol (RHAP) and percent urban land cover in total urban (U), high-density urban (HDU), and low-density urban (LDU) in 1987 and 1997.

Site	Name	Basin area (km ²)	Reach slope	RHAP	1987%			1997%		
					U	HDU	LDU	U	HDU	LDU
1	Pumpkinvine Cr.	16.6	0.0029	12.2	2.9	0.3	2.6	10.5	1.0	9.5
3	Avery Cr.	22.3	0.0015	5.5	7.5	1.3	6.2	19.8	4.0	15.9
4	Smithwick Cr.	15.6	0.0023	14.4	8.2	1.3	6.9	13.3	1.1	12.1
5	McCanless Cr.	13.1	0.0031	16.9	5.8	0.2	5.6	9.4	0.9	8.5
6	Bluff Cr.	14.5	0.0035	13.3	6.6	0.6	6.0	7.7	1.1	6.6
7	Settingdown Cr.	17.1	0.0020	6.5	9.9	1.5	8.4	16.1	2.0	14.1
8	Conns Cr.	14.9	0.0058	17.1	3.7	0.5	3.2	6.4	0.3	6.1
9	Polecat Branch	11.3	0.0085	11.3	17.9	3.3	14.6	16.0	3.7	12.3
10	Burt Cr.	12.2	0.0080	14.7	5.9	1.2	4.7	15.9	3.2	12.7
11	Raccoon Cr.	50.7	0.0034	15.8	4.1	0.2	3.9	10.6	0.6	9.9
12	Little Pumpkinvine	52.0	0.0045	14.3	13.6	2.0	11.5	24.2	2.4	21.8
13	Chicken Cr.	59.1	0.0015	6.3	12.7	1.3	11.5	29.9	3.0	26.9
14	Little River	52.8	0.0025	7.7	7.6	0.8	6.8	14.6	0.7	13.9
15	Mill Cr.	50.7	0.0015	7.8	9.4	1.3	8.1	15.4	1.3	14.2
16	Smithwick Cr.	38.6	0.0052	12.5	7.7	1.1	6.6	11.0	0.8	10.2
17	Shoal Cr. ^a	53.2	0.0029	12.7	3.7	0.3	3.4	7.0	0.5	6.5
18	Settingdown Cr.	53.6	0.0019	9.6	10.1	2.0	8.1	18.7	2.6	16.2
19	Darnell Cr.	60.3	0.0044	12.8	2.7	0.2	2.5	4.9	0.3	4.6
20	Shoal Cr. ^b	53.8	0.0074	14.3	4.7	0.7	4.0	11.2	1.2	10.0
21	Pumpkinvine Cr.	125.7	0.0010	6.5	6.4	0.4	5.9	10.8	1.0	9.9
22	Raccoon Cr.	108.5	0.0028	16.3	4.1	0.2	3.9	8.1	0.4	7.7
24	Settingdown Cr.	96.1	0.0021	8.8	10.4	1.9	8.6	18.4	2.3	16.1
25	Little River	122.1	0.0010	8.4	10.5	1.2	9.4	22.6	1.8	20.7
26	Mill Cr.	84.6	0.0013	8.3	10.3	1.4	8.9	18.8	2.0	16.8
27	Shoal Cr. ^a	101.9	0.0025	12.8	5.4	0.4	5.0	8.5	0.6	7.9
28	Sharp Mountain Cr.	103.9	0.0066	15.0	9.7	2.0	7.7	10.0	2.6	7.4
29	Long Swamp Cr.	77.4	0.0043	15.8	3.5	0.5	2.9	5.0	0.6	4.4
30	Shoal Cr. ^b	90.7	0.0033	16.5	4.7	0.7	4.1	10.1	1.1	9.0
101	Little Allatoona Cr.	14.8	0.0029	7.4	17.0	3.4	13.6	37.3	5.2	32.2
111	Allatoona Cr.	48.4	0.0100	12.7	18.6	2.0	16.6	33.2	3.6	29.5

^a Cherokee County

^b Dawson County

Appendix 2. Fishes collected in the Etowah River basin.

Family name		Family name	
<i>Scientific name</i>	Common name	<i>Scientific name</i>	Common name
Petromyzontidae		Salmonidae	
<i>Ichthyomyzon</i> sp.		<i>Onchorhynchus mykiss</i>	rainbow trout
Cyprinidae		Fundulidae	
<i>Campostoma oligolepis</i>	largescale stoneroller	<i>Fundulus stellifer</i>	southern studfish
<i>Cyprinella callistia</i>	Alabama shiner ^a	Poeciliidae	
<i>C. trichroistia</i>	tricolor shiner ^{a, b}	<i>Gambusia affinis</i>	western mosquitofish
<i>C. venusta</i>	blacktail shiner ^a	<i>G. holbrooki</i>	eastern mosquitofish
<i>Hypopsis lineapunctata</i>	lined chub ^{a, b}	Cottidae	
<i>Luxilus zonistiis</i>	bandfin shiner ^a	<i>Cottus carolinae zopherus</i>	Coosa banded sculpin ^b
<i>Nocomis leptocephalus</i>	bluehead chub	Centrarchidae	
<i>Notemigonus crysoleucas</i>	golden shiner	<i>Ambloplites ariommus</i>	shadow bass
<i>Notropis chrosomus</i>	rainbow shiner ^{a, b}	<i>Lepomis auritus</i>	redbreast sunfish
<i>N. longirostris</i>	longnose shiner ^a	<i>L. cyanellus</i>	green sunfish
<i>N. lutipinnis</i>	yellowfin shiner ^a	<i>L. gulosus</i>	warmouth
<i>N. stilbius</i>	silverstripe shiner ^a	<i>L. macrochirus</i>	bluegill (also known as bluegill sunfish)
<i>N. xaenocephalus</i>	Coosa shiner ^{a, b}	<i>L. megalotis</i>	longear sunfish
<i>Phenacobius catostomus</i>	riffle minnow ^{a, b}	<i>L. microlophus</i>	redeer sunfish
<i>Pimephales vigilax</i>	bullhead minnow	<i>Micropterus coosae</i>	redeye bass (also known as Coosa bass)
<i>Semotilus atromaculatus</i>	creek chub	<i>M. punctulatus</i>	spotted bass
Catostomidae		<i>M. salmoides</i>	largemouth bass
<i>Hypentelium etowanum</i>	Alabama hog sucker	<i>Pomoxis nigromaculatus</i>	black crappie
<i>Minytrema melanops</i>	spotted sucker	Percidae	
<i>Moxostoma duquesnei</i>	black redhorse	<i>Etbeostoma etowahae</i>	Etowah darter ^b
<i>M. erythrurum</i>	golden redhorse	<i>E. jordani</i>	greenbreast darter ^b
<i>M. poecilurum</i>	blacktail redhorse	<i>E. scotti</i>	Cherokee darter ^b
Ictaluridae		<i>E. stigmaeum</i>	speckled darter
<i>Ameiurus brunneus</i>	snail bullhead	<i>Percina kathae</i>	Mobile logperch
<i>A. natalis</i>	yellow bullhead	<i>P. nigrofasciata</i>	blackbanded darter
<i>A. nebulosus</i>	brown bullhead	<i>P. palmaris</i>	bronze darter ^b
<i>Ictalurus punctatus</i>	channel catfish	<i>P. sp. cf. P. macrocephala</i>	“bridled darter” ^b
<i>Noturus leptacanthus</i>	speckled madtom		

^a Insectivorous cyprinid^b Endemic species

