

Importance of Riparian Forests in Urban Catchments Contingent on Sediment and Hydrologic Regimes

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ABSTRACT / Forested riparian corridors are thought to minimize impacts of landscape disturbance on stream ecosystems; yet, the effectiveness of streamside forests in

mitigating disturbance in urbanizing catchments is unknown. We expected that riparian forests would provide minimal benefits for fish assemblages in streams that are highly impaired by sediment or hydrologic alteration. We tested this hypothesis in 30 small streams along a gradient of urban disturbance (1–65% urban land cover). 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 percent forest cover in the riparian corridor and a principal components axis describing sediment disturbance. Only sites with coarse bed sediment and low bed mobility (vs. sites with high amounts of fine sediment) had increased richness and abundances of sensitive species with higher percent riparian forests, supporting our hypothesis that response to riparian forests is contingent on the sediment regime. Abundances of *Etheostoma scotti*, the federally threatened Cherokee darter, were best predicted by models with single variables representing stormflow ($r^2 = 0.34$) and sediment ($r^2 = 0.23$) conditions. Lentic-tolerant species richness and abundance responded only to a variable representing prolonged duration of low-flow conditions. For these species, hydrologic alteration overwhelmed any influence of riparian forests on stream biota. These results suggest that, at a minimum, catchment management strategies must simultaneously address hydrologic, sediment, and riparian disturbance in order to protect all aspects of fish assemblage integrity.

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, see Schueler (1994) and 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 and others 1997; Wang and others 1997, 2001; Snyder and others 2003). In the southeastern United States, homogenization of fish assemblages, as indicated by changes in the ratio of endemic (i.e., native, specialist) to cosmopolitan (i.e., widespread, generalist) species, has been documented with increased urbanization (Scott and Helfman 2001; Walters and others 2003a). Landscape development has also been associated with diet shifts of generalist fishes with foraging flexibility (Weaver and Garman 1994; M. J. Paul, TetraTech, Inc., personal communication). 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 and others 2001; Miltner and others 2004).

KEY WORDS: Fish assemblage; Urbanization; Sedimentation; Hydrologic alteration; Riparian buffers; Watershed management

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Minimizing the effects of urbanization on aquatic biota is a particular concern where systems support exceptional biodiversity. We are investigating shifts in fish assemblages as a consequence of urban development in the Etowah River basin, Georgia, an area that harbors rich biological diversity and is threatened by rapid suburbanization from metropolitan Atlanta (Burkhead and others 1997). Of the 76 native fish species in the basin, 8 are protected by the federal Endangered Species Act or listed as imperiled 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 and others (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 km²). For the same sites used in the present study, Roy and others (2005b) found that increases in magnitude and frequency of storm events and prolonged duration of low-flow conditions (i.e., longer duration of stage <25%, 10%, and 5% of the median stage) resulted in reduced numbers of sensitive fishes and increased numbers of tolerant fishes. Thus, both bed texture and hydrologic regime are known stressors to fish assemblages in the Etowah River basin.

Managers have proposed numerous policies and ordinances to minimize the effects of development on stream ecosystems; however, many of these tools have been employed inconsistently and with variable effectiveness. In the United States, sedimentation remains the second most cited cause of impairment to streams and rivers, despite erosion and sedimentation ordinances that 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 catchments does not allow for complete catchment protection (Hassett and others 2005).

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 extent to which these results can be translated to

urban systems is not well known, and applying riparian buffers as management tools based on false assumptions of performance could lead to ineffective protection. Researchers have predicted that effectiveness of streamside forests will depend on landscape context and upland stressors (Roth and others 1996; Naiman and Decamps 1997; Lowrance 1998); yet there have been few empirical tests of this idea, particularly in urban settings [but see Hession and others (2003)].

Because 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 (1) sediment or (2) hydrologic conditions in the stream [as analyzed by Roy and others (2005b)]. 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 (e.g., reduced baseflows or increased stormflows). For example, if stormwater conveyances bypass riparian corridors, the resulting increases in stormflow magnitude, frequency, duration, and flashiness might provide an overriding filter on fish assemblages. The response of fishes to these reach-scale conditions is also expected to vary based on species life history requirements, habitat preferences, and tolerances to disturbance. This information will be used to guide policy-makers interested in determining what components of catchment management are necessary for maintaining healthy fish assemblages.

Methods

Study Sites

Streams in 30 small catchments (8–20 km²) 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) percent impervious surface cover (<10%, 10–20%, >20%) and (2) baseflow yield (defined as discharge divided by catchment area) determined in the field at one sample date. For sites with 10–20% and >20% impervious surface cover, we randomly selected five streams above the median baseflow yield and five streams below the median baseflow yield within each impervious surface category. For sites with <10% im-

pervious surface cover, there was a 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 five sites located in the northeast (most of which were above the median baseflow yield) and five sites located in the southwest (mostly below the median baseflow yield) portions of the basin. Sites with known point sources of pollution (e.g., quarries) or 100% impoundment of 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 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. 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 and others 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 estimated rather than sampled species richness to account for species with low capture probabilities and differences in detection across sites and to more accurately reflect total species richness. We used model M(h), which assumes heterogeneity of capture probabilities among species, to estimate species richness (Williams and others 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. See Roy and others (2003b) for species list and designated composition categories. Sensitive species are 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, unpublished data). Fluvial specialists are species that require lotic environments for at least part of their life cycle; we classified these species using the work of Etnier and Starnes (1993) and Mettee and others (1996). Lentic tolerants are 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 land cover, hydrology, and/or sediment 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[®] 3.3 geographic information systems (GIS) software. Classified *Landsat* Thematic Mapper (TM) satellite imagery (30-m resolution) was used to calculate percentages of land-cover categories within subcatchments 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 subcatchment 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 and others (2005b) 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 but included more headwater streams, 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. As is true in many catchments (Lammert and Allan 1999; Wang and others 2001; Allan 2004), riparian land cover was correlated with land cover in the entire catchment. Specifically, percent forest cover in riparian corridors was tightly correlated with catchment percent forest ($r = 0.95$), percent urban ($r = 0.90$), and percent impervious ($r = 0.88$). Thus, analyses using riparian forest cover can be assumed to have similar results if

Table 1. Description of habitat variables used in principal components analysis (PCA)

Habitat variable	Eigenvector 1	Description
Velocity variability (m/s)	-0.07	Standard deviation of velocity (m/s) at 70 points along thalweg for reach
Bed sediment variability (Φ)	-0.34	Standard deviation of modal bed sediment size (Φ) ^a measured at 70 0.5-m ² plots along thalweg for reach
Depth variability (m)	0.21	Standard deviation of depth (m) at 70 points along thalweg for reach
Rugosity	0.21	Length of wetted surface divided by water width, average of 10 transects
Percent riffle habitat	-0.39	Percentage of total reach length designated as riffle habitat
Mean percent fines in riffles	0.49	Mean percent dry sieved riffle material < 2 mm, average of three 3-L replicates
Variability percent fines in riffles	0.36	Standard deviation of percent dry sieved riffle material < 2 mm, average of three 3-L replicates
Froude number	-0.14	Channel froude number for 0.5-year recurrence interval flood ^b
Bed mobility	-0.44	Mean velocity for 0.5-year recurrence interval flood divided by critical velocity to move mean thalweg Φ ^b
Turbidity (NTU)	0.25	Mean turbidity (NTU) at baseflow conditions, average of three or four dates

Note: The first principal component (sediment PC₁) explained 32.2% of the variation in habitat variables. Numbers in bold indicate large weightings on the first eigenvector.

^a $\Phi > 0$ (i.e., particle size < 1 mm) converted to zero for consistency among observers.

^bCalculated using HEC-RAS[®].

catchment land cover (forest or urban) was used. Because this correlation could not be avoided, management implications associated with this caveat are further addressed in the discussion.

Geomorphology

Ten variables intended to describe the habitat and sediment conditions for fishes were measured in each stream reach (see Table 1). Velocity, depth, and modal bed sediment size (ϕ , for a 0.5-m² 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 extended along the bottom of the stream by the wetted width of the stream at that location, and averaged for 10 random transects along the stream reach (Garcia-Charton and Perez-Ruzafa 2001). We measured the length of stream designated as riffle habitat to determine the percent riffle habitat in each stream reach. We collected 3 L of bed sediment from the top 10 cm of three riffles to determine percent fines in riffle habitats. Samples were dried, sieved, and weighed in the lab to determine mean percent fines (< 2 mm) for each stream.

We used steady-flow analysis in HEC-RAS[®] (version 2.2; Hydrologic Engineering Center, US Army Corps of Engineers) to determine the mean hydraulic depth, mean velocity, and Froude number (a descriptor of main channel flow) for the 0.5-year recurrence interval (RI) flood at each site. Discharges for the 0.5-year 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 AquaRod[®] water-level sensor were determined from field measurements and incorporated into the HEC-RAS[®] model for each stream. We used a Topcon[®] AT-F6 level and stadia rod 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. Although some variables incorporated in the HEC-RAS[®] model are estimated and therefore might be inaccurate (e.g., discharges based on published regional flood-frequency curves and Manning's n), we feel that the model provides useful data for comparison across sites. Bed mobility was calculated as a ratio of the mean velocity (V) for the 0.5-year RI flood near the streambed (V_b ; $V_b = 0.7V$) divided by critical velocity (V_c) to move the modal thalweg bed sediment size (d ; $V_c = 0.155\sqrt{d}$) (Gordon and others 2004).

Hydrology

We used hydrologic data previously computed by Roy and others (2005b) to evaluate the interactive effects of flow alteration, sediment condition, and riparian forest cover on fish assemblages. Briefly, streams were gauged at the base of each catchment during 2003 using 2-m AquaRod[®] water-level sensors (Advanced Measurements & Controls, Inc., Woodinville, WA, USA) that use electrical capacitance to measure stage height. Because many of the AquaRods[®] (24 of the 30) failed at various times throughout the year, we divided data into seasons based on downloading dates. 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 for both time periods. 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. We divided baseflow magnitude variables 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 \times 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-year RI flood divided by the mean daily stage (to adjust for differences in gauge location across sites) were used to calculate stormflow variables.

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 $\arcsin(\sqrt{\%/100})$ to improve linearity.

Principal components analysis (PCA) of the correlation matrix was used to reduce geomorphic variables into a single metric describing bed texture and mobility (sediment PC₁). We also used PCA to reduce the variables into metrics describing stream hydrologic conditions for the most appropriate season, calculating separate PC axes for baseflow and stormflow variables. We evaluated stormflow conditions during summer (May–August), when spawning and small young-of-year fishes were most vulnerable to elevated flows. Summer stormflow PC₁ included a combination of storm frequency, magnitude, duration, volume, and rates of the rising and falling limbs of the hydrograph and explained 54.8% of the variation across sites (Roy and others 2005b). Baseflow conditions were calculated for autumn, the low-flow time of year, when we expected the strongest influence of reduced baseflow conditions on fish assemblages. Autumn baseflow PC₂ reflected increased duration of low-flow conditions and was previously shown to explain variation in some fish assemblage metrics (Roy and others 2005b).

We used multiple linear regression analysis to predict fish assemblages using PCA axis scores for sedi-

Table 2. Variables in each model set for Akaike's Information Criterion analysis

Sediment model set ($n = 30$ sites)	
Riparian	
Sediment PC ₁	
Riparian + sediment PC ₁	
Riparian + riparian \times sediment PC ₁	
Sediment PC ₁ + riparian \times sediment PC ₁	
Hydrologic model set ($n = 16$ sites)	
Riparian	
Base PC ₂	
Storm PC ₁	
Riparian + base PC ₂	
Riparian + storm PC ₁	
Riparian + riparian \times base PC ₂	
Riparian + riparian \times storm PC ₁	
Base PC ₂ + riparian \times base PC ₂	
Storm PC ₁ + riparian \times storm PC ₁	

Note: Riparian = percent forest in a 30-m riparian buffer. Sediment PC₁, base PC₂, and storm PC₁ indicate principal component axes representing sediment, baseflow, and stormflow conditions, respectively.

ment, summer stormflow, and autumn baseflow. We compared five least-squared models, including single variables of percent riparian forest cover or sediment PC₁, addition of the two variables, and percent riparian forests or sediment PC₁, plus an interaction term between percent riparian forests and sediment PC₁ to predict various fish assemblage metrics based on 30 sites (total of 5 models; Table 2). 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 percent riparian forest cover, stormflow PC₁, baseflow PC₂, riparian \times stormflow PC₁ interaction, and riparian \times baseflow PC₂ interaction (total of 9 models; Table 2). 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-support-

Table 3. Summary data for 30 study sites and the subset of 16 sites used in hydrologic analyses

	Land cover ^a and geomorphology						Fish assemblage ^b						
	30 Sites			16 Sites			30 Sites			16 Sites			
	Min	Mean	Max	Min	Mean	Max	Min	Mean	Max	Min	Mean	Max	
Basin area (km ²)	8.5	13.6	19.9	8.5	12.9	17.0	Total						
% Riparian forest cover ^c	39.1	76.3	100	55.1	80.5	100	Richness	8	17.4	25	12	17.3	25
Catchment land cover (2001)							Abundance	37	184.9	512	37	171.8	394
% Impervious ^c	1.7	12.0	31.0	3.3	11.5	31.0	Sensitive						
% Urban ^c	1.4	25.2	65.1	1.7	23.4	65.1	Richness	0	2.7	7	0	2.5	6
% Forest ^c	29.0	56.6	96.4	29.0	57.7	86.2	Abundance ^d	0	16.7	103	0	15.0	49
% Agriculture ^c	0.1	9.2	27.3	0.1	9.4	24.4	Fluvial specialist						
% Open water ^c	0	0.8	2.0	0.1	0.8	2.0	Richness	5	12.3	21	5	12.1	21
Geomorphology							Abundance	10	128.5	469	34	135.1	362
Slope (%)	0.10	0.39	0.88	0.10	0.40	0.87	Lentic tolerant						
% Riffles ^c	0	22.8	62.3	1.5	25.3	62.3	Richness	3	5.4	11	3	5.4	11
Bed sediment variability (ϕ)	0.6	2.3	4.6	0.6	2.1	4.6	Abundance ^d	2	56.9	412	2	37.4	89
Mean % fines in riffles ^c	1.5	25.3	89.0	1.5	22.5	62.5	<i>Etheostoma scotti</i>						
Variability % fines in riffles ^c	0.9	8.2	26.1	0.9	9.4	26.1	Abundance ^d	0	6.2	40	0	6.9	40
Bed mobility ^f	0.7	4.9	28.0	0.9	6.2	28.0							

^aLand cover was calculated from *Landsat* TM imagery for each subcatchment.

^bGeomorphic 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 three consecutive 50-m reaches and to calculate capture probabilities from three-pass removal in one 50-m reach.

^cTransformed using $\arcsin(\sqrt{\%/100})$ for analysis.

^dTransformed using $\log(x + 1)$ for analysis.

^eTransformed using $\log(\arcsin(\sqrt{\%/100}) + 1)$ for analysis.

^fTransformed using y^{-1} for analysis.

ing model by incorporating a penalty for inclusion of additional variables into the model. Akaike weights (w_i) were computed as $w_i = \exp(-\frac{1}{2}\Delta_i) / \sum \exp(-\frac{1}{2}\Delta_i)$, 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 $\exp(-\frac{1}{2}\Delta_i)$ 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 the highest percent

fines in riffles and bed mobility (within top 40% of sites) and had the lowest sediment PCA values were labeled as "fine bed texture" ($n = 9$ sites). Sites that had the lowest percent fines in riffles and bed mobility (within bottom 40% of sites) and had the highest sediment PCA values were labeled as "coarse bed texture" ($n = 9$ 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 condition categories. All analyses were performed using JMP[®] version 4.0 statistical software (SAS Institute Inc., Cary, NC, USA).

Results

Sediment Condition and Riparian Forests

On average, land cover in the subcatchments was dominated by percent urban and percent forest land cover; however, there was a wide range in land-cover types across sites (Table 3). Percent riparian forest cover reflected similar differences in land cover, ranging from 39% to 100% across the 30 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 3).

Table 4. Linear regression results for models predicting fish assemblage characteristics with riparian and sediment variables

	Adj. R^2	Δ_i	w_i
Sensitive richness			
Riparian + riparian \times sediment PC₁	0.40	0.00	0.41
Riparian + sediment PC₁	0.37	0.72	0.28
Riparian	0.22	1.24	0.22
Sediment PC ₁	0.05	3.78	0.06
Sediment PC ₁ + riparian \times sediment PC ₁	0.11	5.19	0.03
Sensitive abundance			
Riparian + riparian \times sediment PC₁	0.46	0.00	0.49
Riparian + sediment PC₁	0.41	1.25	0.26
Sediment PC ₁	0.20	2.92	0.11
Sediment PC ₁ + riparian \times sediment PC ₁	0.30	3.47	0.09
Riparian	0.10	4.44	0.05
Fluvial specialist richness			
Riparian	0.17	0.00	0.41
Riparian + riparian \times sediment PC₁	0.24	1.15	0.23
Riparian + sediment PC₁	0.23	1.38	0.21
Sediment PC ₁	0.00	2.46	0.12
Sediment PC ₁ + riparian \times sediment PC ₁	-0.01	4.86	0.04
Fluvial specialist abundance			
Riparian	0.17	0.00	0.31
Riparian + riparian \times sediment PC₁	0.26	0.28	0.27
Riparian + sediment PC₁	0.25	0.51	0.24
Sediment PC ₁	0.03	1.57	0.14
Sediment PC ₁ + riparian \times sediment PC ₁	0.01	4.12	0.04
<i>E. Scotti</i> abundance			
Sediment PC₁	0.20	0.00	0.34
Riparian + riparian \times sediment PC₁	0.29	0.63	0.25
Riparian + sediment PC₁	0.28	0.83	0.23
Sediment PC ₁ + riparian \times sediment PC ₁	0.19	2.33	0.11
Riparian	0.00	2.94	0.08

Note: Models including riparian (percent forest in 30-m riparian buffer) and sediment principal component axis 1 (sediment PC₁) 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 AIC_c from minimum (Δ_i), and Akaike weights (w_i) of each model are reported. Bold type indicates best-supported models.

Sites exhibited a range in percent fines (1.5–89.0%) and variability in percent fines (0.9–26.1%; Table 3) 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-year RI flood (i.e., bed mobility >1; Table 3). When combining the 10 geomorphic variables using PCA, PC₁ explained 32.2% of the variance in geomorphic variables across sites (Table 1). PC₁ increased with increased sediment disturbance, as reflected by increased percent 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 percent riparian forest plus an interaction between percent riparian forest and sediment PC₁ (Table 4). The model of percent riparian forest plus sediment PC₁ (i.e., linear addition rather than an interaction term) was also a strong model. Fluvial specialist species richness and abundance was best supported by a model including

percent riparian forest; however, an interaction between percent riparian forest and sediment PC₁ explained the highest amount of variation in fluvial specialist species (i.e., highest adjusted R^2). The best-supported model for abundance of *E. scotti* was with sediment PC₁, whereas the model with percent riparian forest was least supported (Table 4). Richness and abundance of lentic-tolerant species were not predicted well by any of the models with percent riparian forest and sediment PC₁ (i.e., low adjusted R^2), so we did not compare fit among models.

We added the weights of all models with percent riparian forest (four) and all models with sediment PCA (four) 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 percent riparian forest and sediment PC₁ variables are equally important in predicting richness and

Table 5. Total combined Akaike weights (w_i)

	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	
<i>E. scotti</i>					
Abundance					
Sediment	0.93	1.4	Stormflow	0.70	4.1
Riparian	0.67		Baseflow	0.17	

Note: Combined weights are 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 w_i .

abundance of sensitive species and fluvial specialist species, and abundance of *E. scotti* (Table 5).

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 percent riparian forest cover to test whether relationships between percent riparian forest and fishes were contingent on levels of hydrologic alteration. Lentic-tolerant species richness was best predicted by autumn baseflow PC₂ (Table 6). When comparing the summed weights of four models with stormflow versus four models with baseflow, there was substantial evidence that baseflow was more important than stormflow in predicting richness of lentic-tolerant fishes (i.e., combined weights of all baseflow PC₂ models were 7.8 times higher than weights of stormflow PC₁ models; Table 5). Abundance of lentic-tolerant species was equally well predicted by percent riparian forest and baseflow PC₂; however, there was little support for including an interaction term in the

models (Table 6). Sensitive species richness and abundance were best explained by the model with stormflow PC₁ (Table 6). Combined weights for stormflow and baseflow models revealed that stormflow was 3.3 and 2.0 times more influential than baseflow on sensitive species richness and abundance, respectively (Table 5). Abundance of *E. scotti* was also best explained by stormflow PC₁ (Table 6). Fluvial specialist species richness and abundance were not well predicted by any of the models including stormflow, baseflow, and/or riparian condition with this dataset (i.e., low adjusted R^2), so we did not compare models using 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 PC₁, stormflow PC₁, and baseflow PC₂ using AIC_c. Thus, we compared adjusted R^2 values for the best-supported models determined by AIC_c (while accounting for differences

Table 6. Linear regression results for models predicting fish assemblage characteristics with riparian and hydrologic variables

	Adj. R^2	Δ_i	w_i
Lentic tolerant richness			
Base PC₂	0.63	0.00	0.51
Riparian	0.44	2.82	0.12
Riparian + base PC ₂	0.62	3.20	0.10
Base PC ₂ + riparian × base PC ₂	0.61	3.35	0.10
Riparian + riparian × base PC ₂	0.57	4.06	0.07
Storm PC ₁	0.25	4.83	0.05
Riparian + storm PC ₁	0.43	6.09	0.02
Riparian + riparian × storm PC ₁	0.43	6.09	0.02
Storm PC ₁ + riparian × storm PC ₁	0.20	8.36	0.01
Lentic tolerant abundance			
Riparian	0.24	0.00	0.28
Base PC₂	0.24	0.01	0.28
Storm PC ₁	0.04	1.57	0.13
Base PC ₂ + riparian × base PC ₂	0.25	2.76	0.07
Riparian + riparian × base PC ₂	0.26	2.91	0.07
Riparian + base PC ₂	0.23	3.19	0.06
Riparian + riparian × storm PC ₁	0.18	3.63	0.05
Riparian + storm PC ₂	0.18	3.64	0.05
Storm PC ₁ + riparian × storm PC ₁	-0.03	5.19	0.02
Sensitive richness			
Storm PC₁	0.30	0.00	0.45
Riparian	0.02	2.29	0.14
Base PC ₂	-0.07	2.89	0.10
Storm PC ₁ + riparian × storm PC ₁	0.25	3.57	0.07
Riparian + storm PC ₁	0.25	3.58	0.07
Riparian + riparian × storm PC ₁	0.22	3.83	0.07
Riparian + riparian × base PC ₂	0.08	4.98	0.04
Riparian + base PC ₂	0.01	5.48	0.03
Base PC ₂ + riparian × base PC ₂	-0.01	5.60	0.03
Sensitive abundance			
Storm PC₁	0.16	0.00	0.36
Riparian	-0.04	1.47	0.17
Base PC ₂	-0.06	1.60	0.16
Riparian + storm PC ₁	0.12	3.46	0.06
Storm PC ₁ + riparian × storm PC ₁	0.10	3.59	0.06
Riparian + riparian × storm PC ₁	0.09	3.66	0.06
Base PC ₂ + riparian × base PC ₂	0.07	3.78	0.05
Riparian + riparian × base PC ₂	-0.10	4.96	0.03
Riparian + base PC ₂	-0.12	5.11	0.03
<i>E. scotti</i> abundance			
Storm PC₁	0.30	0.00	0.45
Riparian	-0.02	2.54	0.12
Base PC ₂	-0.05	2.76	0.11
Riparian + riparian × storm PC ₁	0.29	3.20	0.09
Storm PC ₁ + riparian × storm PC ₁	0.27	3.35	0.08
Riparian + storm PC ₁	0.27	3.37	0.08
Riparian + riparian × base PC ₂	-0.09	6.13	0.02
Riparian + base PC ₂	-0.09	6.17	0.02
Base PC ₂ + riparian × base PC ₂	-0.10	6.22	0.02

Note: Models including riparian (with percent forest in 30-m riparian buffer), summer stormflow principal component axis 1 (storm PC₁), and autumn baseflow principal components axis 2 (base PC₂) are 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 AIC_c from minimum (Δ_i), and Akaike weights (w_i) of each model are reported. Bold type indicates best-supported model(s).

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. Sensitive species richness and abundance were best explained by a combination of riparian forest and

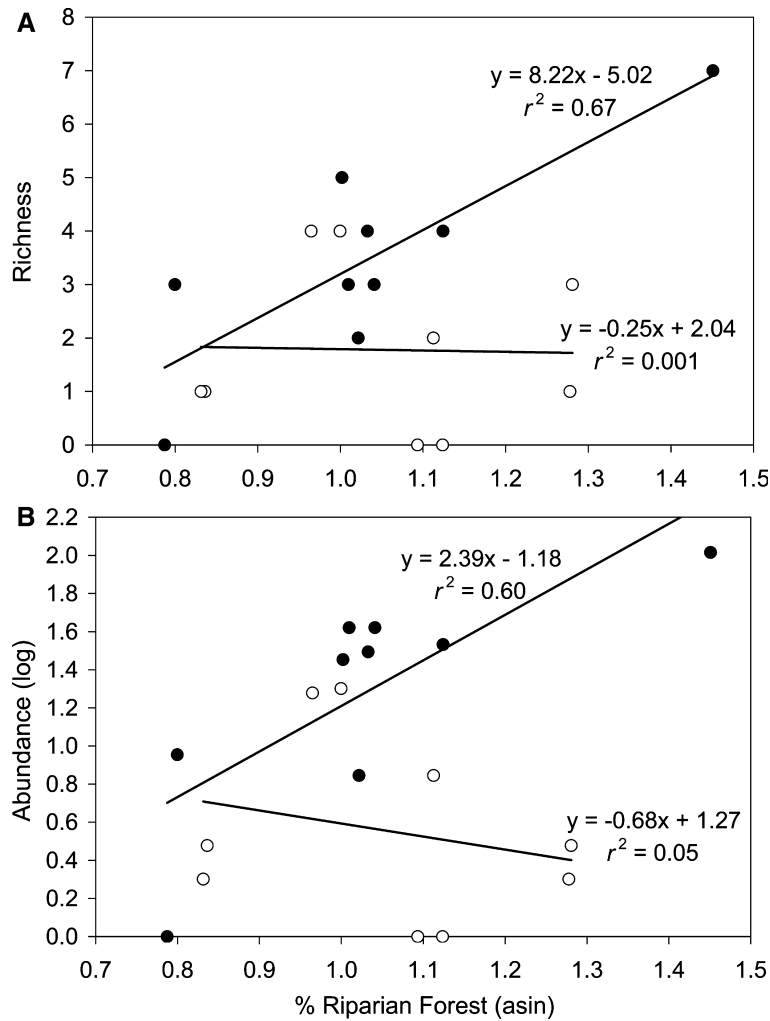


Figure 1. Relations between sensitive species richness (**A**) and abundance (**B**) and percent forest cover in 30-m riparian buffer for upstream network. Lines represent best-fit linear regressions for coarse (●) and fine (○) bed texture ($n = 9$ sites for each category). Sites designated as having coarse bed texture were among the top 40% of sites with low percent fines in riffles and low bed mobility, and sites designated as having fine bed texture were among the top 40% of sites with high percent fines in riffles and high bed mobility.

sediment variables, accounting for 40% and 46% of the variation in fishes, respectively (Table 4). When sites were divided into classes based on bed sediment conditions, regressions indicated a significant positive relationship between percent riparian forest and richness and abundance of sensitive fishes at sites with coarse bed texture, but no relationship at sites with fine bed texture (Figure 1).

In contrast to sensitive fish species, fluvial specialists, lentic tolerants, and *E. scotti* were best predicted by single variables (Figure 2). Fluvial specialist species were best predicted exclusively by percent riparian forests; however, this variable explained only 17% of the variation in richness and abundance in the 30 sites (Table 4 and

Figures 2A and 2B). Richness and abundance of lentic-tolerant species were best predicted by increases in the duration of low-flow conditions (baseflow PC_2 ; Table 4 and Figures 2C and 2D). Abundance of *E. scotti* was related to reduced stormflow alteration (storm flow PC_2) and coarse bed texture (Tables 4 and 5 and Figures 2E and 2F). The relationship with summer stormflow revealed a threshold-type response, where no individuals were found at sites with high levels of stormflow disturbance (Figure 2E). The relationship with sediment revealed a wedge-shaped pattern, indicating that increases in fine sediment result in an upper limit in the abundance of this species; however, there are other variables explaining low abundances in sites with coarse bed tex-

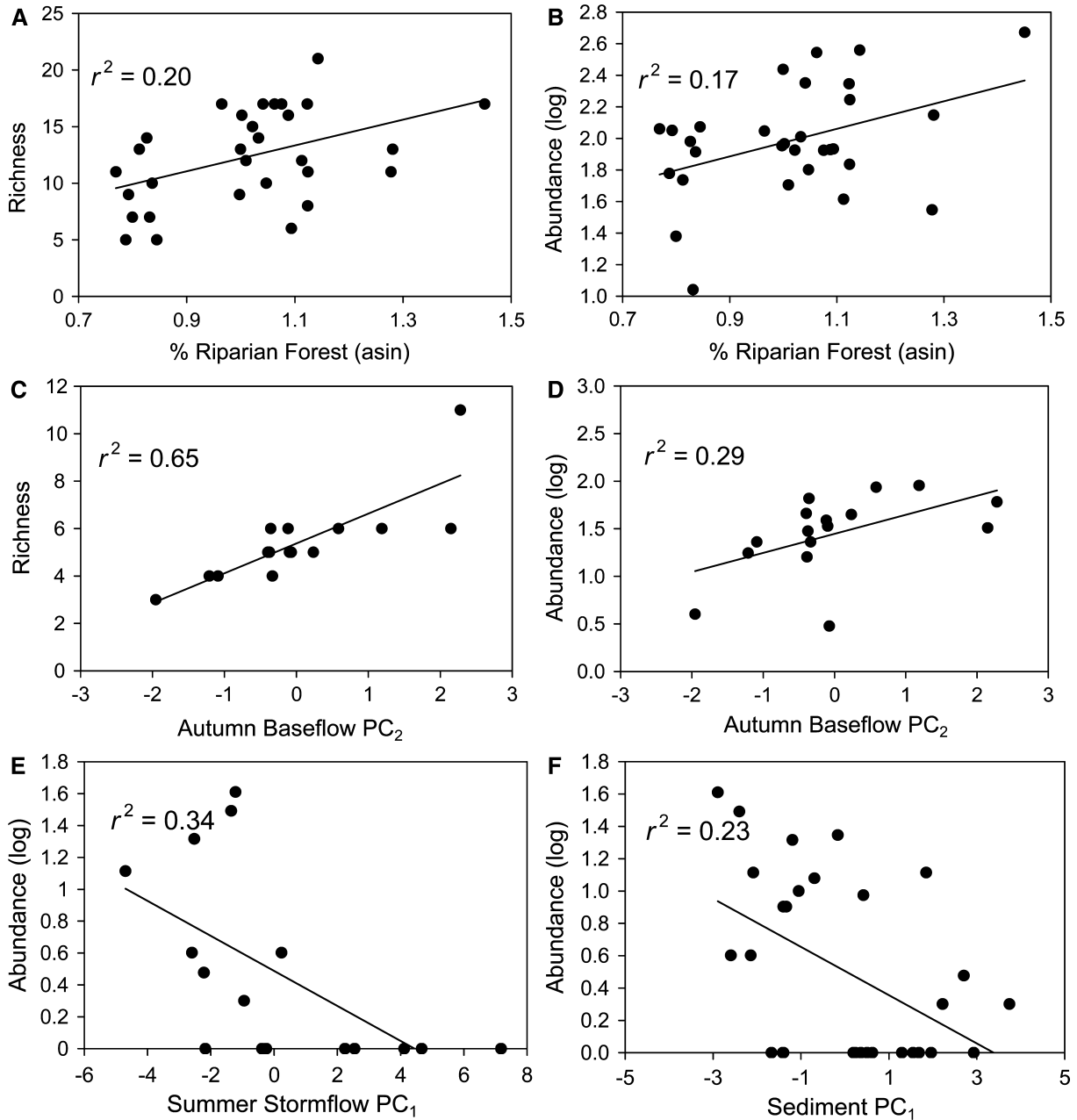


Figure 2. Linear regressions models (r^2) for fluvial species richness (A) and abundance (B), lentic-tolerant richness (C) and abundance (D), and *E. scotti* abundance (E, F) versus independent variables for the best-supported models (based on Akaike's weight comparisons). Relations with percent riparian forest and sediment PC₁ include 30 sites (A, B, F), and relations with hydrologic variables include 16 sites (C, D, E).

ture (Figure 2F). If sites without *E. scotti* are excluded (e.g., abundance = 0), sediment explains 35% of the variation in abundances of *E. scotti* across sites.

We also asked whether stream slope (a natural factor) or landscape disturbance (anthropogenic factors) were related to riparian forest, sediment, and hydro-

logic conditions at these sites. Increased sediment disturbance (sediment PC₁) was related to decreased slope and greater reductions in percent forest cover from 1992 to 2001, but not to other land-cover variables (Table 7). Percent riparian forest was strongly correlated with catchment land cover (percent urban,

Table 7. Pearson's correlations (r) between land-cover characteristics and predictor variables

	Sediment PC ₁ (30)	% Riparian Forest (30)	% Riparian Forest (16)	Stormflow PC ₁ (16)	Baseflow PC ₂ (16)
Slope	-0.54	0.03	-0.12	-0.56	0.20
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

Note: Higher principal components (PC) scores indicate increased disturbance. Numbers in parentheses indicate number of sites in analysis. Bold numbers indicate $r \geq 0.40$.

percent forest) and changes in land cover (percent urban, percent agriculture) in the last decade. Stormflow and baseflow alterations were also negatively related to percent forest cover and positively related to percent urban cover (Table 7).

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 and others 1997). In our study, local-scale sediment condition (increased percent fines, increased bed mobility, decreased variability in bed sediment size) affected the ability to predict sensitive fish and fluvial specialist species based on riparian (or catchment) forest cover. Interestingly, these fishes responded primarily to percent riparian forest (vs. sediment regime), although inclusion of an interaction between these variables was necessary to best explain richness and abundance of sensitive species. The interaction effect implies differential responses of fishes to riparian cover depending on the level of sedimentation. Increased fines (and other habitat/sediment 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 with elevated fine sediment, subsequently influencing sensitive and fluvial specialist species. Under these conditions, forest cover will not be sufficient to protect these fishes in developing catchments. Conversely, streams that have coarse bed

texture might not harbor high abundances of sensitive species where streams have low ($\sim < 70\%$) riparian forest cover along the upstream network. Thus, maintaining streams with both coarse bed texture and adequate forest cover (in riparian or catchment) might be an effective strategy for protecting sensitive fish species.

Lentic-tolerant species appeared facilitated by reduced baseflow conditions, suggesting that maintenance of adequate low flows throughout the year is important for preventing the shift of assemblages to dominance by tolerant species. In small streams, prolonged low flows might create conditions of depleted oxygen, thus favoring species that are tolerant to oxygen depletion typical of lentic environments (Mulholland and others 1997). It is also possible that prolonged low flows create conditions unsuitable for fluvial specialists, and the release of competitive pressure from the loss of these species results in more favorable conditions for lentic-tolerant species. However, we did not find strong relationships between fluvial specialist richness or abundance and baseflow conditions. Abundances of lentic-tolerant species were also explained by percent riparian forest in the 16 sites, but the trend did not hold for the 30 sites. The 16 sites had a narrower range in lentic-tolerant abundance and percent riparian forest cover (Table 3), so we expect that the significant relationship with the 16 sites is an artifact of the data and does not reflect a strong pattern.

This study was designed, in part, to guide planning and management in north Georgia that will ensure protection of imperiled fish species as part of the Etowah Regional Aquatic 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 US Endangered Species Act. The Etowah HCP will cover as many as nine fish species, of which the Cherokee darter, *E. scotti*, is the most common in small streams in the Etowah River basin. Our results demonstrate that streams with the highest stormflow alteration (the top five) did not have *E. scotti*. In streams that harbored *E. scotti*, abundances were reduced with increased fine sediments. Although we did not test an interaction between stormflow and sediment conditions (because of differences in sample sizes for each variable), these results confirm that management must account for both factors in order to protect populations of *E. scotti*.

Despite the relations observed between fish assemblage measures and riparian, hydrologic, and sediment variables, these variables 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 percent 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 and others 1996; Wang and others 2001; Walters and others 2003a; Miltner and others 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 and others 2003). These generally weak relations suggest that natural and/or anthropogenic stressors other than those measured in this study (e.g., biogeography, dispersal barriers, water quality, biotic interactions, etc.) influence these fish assemblages. Even where we observed a strong relationship with riparian, sediment, and/or hydrology variables, such relationships do not imply a mechanism of fish assemblage shifts (e.g., stormflow alteration could reflect effects of pollutant delivery). Any of these responses could be a result of other variables that are also correlated with the predictor variables. Although this study does not provide a comprehensive picture of factors affecting fish assemblages, it demonstrates that multiple factors at various spatial scales are likely to be influencing fishes in these urban headwater streams.

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 and others 2004). Stream restoration efforts have also focused on riparian buffers; approximately one-third of the restoration projects in the United States (Bernhardt and others 2005) and 63% of the projects in the Chesapeake Bay watershed (Hassett and others 2005) list riparian management as the major restoration goal. Although riparian buffers have been promoted as an essential tool for stream mitigation, there is insufficient scientific research, especially in urban areas, to support this bias in stream management toward riparian buffers. Naiman and Decamps (1997) suggest that the role and extent of influence of riparian areas on aquatic systems depend on landscape context and associated upland stressors. In other words, riparian forests might only be important for maintaining a high-quality habitat if upstream processes do not have an overwhelming impact on biotic assemblages (Roth and others 1996; Hession and others 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.

Due to the tight correlation between riparian forest cover and catchment forest and urban cover, we were unable 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 landcover types within each basin, the location of forests within a catchment (i.e., adjacent to streambanks or upland) might not be critical for stream protection, as long as a certain percent forest cover is maintained throughout the catchment. Likewise, it appears equally important to address impervious and urban land cover within the catchment. The inherent correlation among these scales and land cover suggest that it is inappropriate to assume that riparian forests will provide adequate stream protection regardless of upstream uses. Nonetheless, maintenance of riparian forests will necessarily be beneficial because this area is part of the overall catchment land cover. However, the function of these forests remains dependent on other catchment factors.

Although sediment, hydrology, and riparian forests were considered separately in these analyses, they are necessarily dependent on each other. Altered hydro-

ogy can increase bank erosion, acting as a source of sediment disturbance, whereas large, scouring flows associated with hydrologic alteration can also transport fine sediments, acting to increase streambed coarseness (Booth and Jackson 1997; Finkenbine and others 2000). Further, riparian forests have been used to trap sediment and slow down overland flows (Lowrance 1998). 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 hydrologic 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 and others 2003). Hession and others (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).

Although the functions of riparian forests in urban areas might be limited, trees on streambanks still provide essential functions of shading, stabilizing banks, and providing organic matter inputs to streams (Lowrance 1998). Two recent studies comparing reach-scale differences in open and forested reaches in urbanizing catchments found that riparian forest cover can be an important regulator of stream width, temperature, and the food/energy base in local stream reaches; however, they do not control habitat quality or biotic integrity

(Hession and others 2003; Roy and others 2005a). In heavily urbanized catchments where forest cover is limited, riparian trees might provide the only source of organic inputs and shading for stream ecosystems.

Holistic Approach for Catchment Management

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

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
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 and others 2002) and disconnecting drainage systems to allow source infiltration of stormwater (Walsh 2004) should permit development with less loss of assemblage integrity than conventional development.

It is important to acknowledge that even small transformations of the natural landscape might result in aquatic impacts, and maintaining healthy biotic assemblages will necessarily require limits to impervious surface cover (Booth and others 2002). Stakeholder appreciation of essential services provided by natural stream ecosystems is necessary to connect these scientific findings to management solutions (Palmer and others 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). Because many of these species reside in the main stem of the Etowah River, protection of the entire catchment, especially the headwater regions, will be critical. The HCP might also include prioritizing catchments for protection: maximizing protection in

some subcatchments 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 allow natural processes to occur (Lloyd and others 2002). Many have suggested a degradation threshold of ~10% impervious surface cover, but poorly designed urban land use can impact aquatic systems at levels much less than 10% impervious cover (Walsh 2004). Conversely, greater development might be possible if we can disconnect stormwater pathways (Walsh 2004) and maintain functional riparian buffers (May and others 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 and others 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 in maintaining healthy fish assemblages will continue to be tested via adaptive management (Wilhere 2002). For this basin and likely other catchments threatened by urbanization, this research has highlighted the importance of combining multiple management tools to protect fish assemblage integrity.

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