

Effects of Landscape Change on Fish Assemblage Structure in a Rapidly Growing Metropolitan Area in North Carolina, USA

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Abstract.—We evaluated a comprehensive set of natural and land-use attributes that represent the major facets of urban development at fish monitoring sites in the rapidly growing Raleigh-Durham, North Carolina metropolitan area. We used principal component and correlation analysis to obtain a nonredundant subset of variables that extracted most variation in the complete set. With this subset of variables, we assessed the effect of urban growth on fish assemblage structure. We evaluated variation in fish assemblage structure with nonmetric multidimensional scaling (NMDS). We used correlation analysis to identify the most important environmental and landscape variables associated with significant NMDS axes. The second NMDS axis is related to many indices of land-use/land-cover change and habitat. Significant correlations with proportion of largest forest patch to total patch size ($r = -0.460$, $P < 0.01$), diversity of patch types ($r = 0.554$, $P < 0.001$), and population density ($r = 0.385$, $P < 0.05$) helped identify NMDS axis 2 as a disturbance gradient. Positive and negative correlations between the abundance of redbreast sunfish *Lepomis auritus* and bluehead chub *Nocomis leptocephalus*, respectively, and NMDS axis 2 also were evident. The North Carolina index of biotic integrity and many of its component metrics were highly correlated with urbanization. These results indicate that aquatic ecosystem integrity would be optimized by a comprehensive integrated management strategy that includes the preservation of landscape function by maximizing the conservation of contiguous tracts of forested lands and vegetative cover in watersheds.

Introduction

Conversions of rural and forest lands to urban land degrade streams (Booth and Jackson 1997; Kennen 1999; Wang et al. 2000, 2001) by altering the composition, structure, and function of aquatic ecosystems (Frissell et al. 1986; Jones and Clark 1987; Richards and Host 1994; Richards et al. 1997; Lammert and Allan 1999; Kennen and Ayers 2002; Roy et al. 2003). Landscape changes associated with urbanization include terrestrial habitat loss, landscape

fragmentation, increased impervious surface area, increased storm runoff, reduced groundwater recharge, and riparian habitat loss (Wang and Lyons 2003). Urbanization is linked consistently to stream degradation, which results from increased peak flows, stream power and stream sedimentation, reduced base flows, and modified instream habitat and substrate complexity (Klein 1979; Schueler 1994; Booth and Jackson 1997; Wang et al. 2000, 2001; Kennen and Ayers 2002; Walters et al. 2003). These changes are accentuated when connected forests and undeveloped lands are replaced with a patchwork of smaller and smaller interspersed parcels of lands fragmented

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by urban land uses. The changes not only modify the landscape in measurable ways, but also increase the extent of stream ecosystem degradation. Riparian areas are particularly susceptible to urbanization impacts and habitat fragmentation. Loss of riparian vegetation can destabilize stream banks, increase summer water temperatures and diel fluctuations, alter the recharge of shallow aquifers, and reduce the effectiveness of these natural filters (Karr and Gorman 1975; Kleiss et al. 1989; Jensen and Platts 1990). Loss of riparian vegetation results in increased surface runoff, increased erosion and sedimentation, and reduced woody debris and leaf litter that are used by many aquatic organisms for food and shelter (Finkenbine et al. 2000). Declines of native fish, amphibian, and aquatic invertebrate assemblages have been linked to deterioration of riparian habitats (Dodd and Smith 2003).

Terrestrial habitat fragmentation results in smaller habitats suitable for survival and fewer corridors suitable for dispersion and migration (Noss 1987). It is one of the most commonly cited threats to loss of biological diversity (D'Eon et al. 2002), and its effects on terrestrial systems have been extensively studied (Saunders et al. 1991; Brooker and Cale 1999; McCoy and Mushinsky 2000). How forest fragmentation affects water quality and stream processes is less well known; however, such understanding is important for evaluating the ways in which humans can minimize their impacts on aquatic ecosystems. Thus, the goal of this study was to evaluate the effects of urban development on fish assemblage structure. Our specific objectives were to (1) identify a subset of urbanization indicators (e.g., land use/land cover, fragmentation indices, riparian habitat) that extract most of the variation along a disturbance gradient, and (2) determine correlations between those urbanization indicators and variations in fish assemblage structure.

Study Area

The Raleigh-Durham, North Carolina study area (RDU; Figure 1) covers approximately 8,579 km², has a population of nearly 1.5 million and is the second fastest growing metropolitan area in the United States (<http://www.census.gov/prod/www/statistical-abstract-O2.html>). In addition, this area was the third most sprawling metropolitan area of 83 measured in the conterminous United States and Hawaii (Ewing et al. 2002). Population in the study area doubled from 1970 to 2000, and the amount

of urbanized land increased by 150% over the same time period.

The study area is primarily within the Northern Outer Piedmont level IV ecoregion with small parts in the Triassic Basins, the Carolina Slate Belt, and the Rolling Coastal Plain (Griffith et al. 2002). All of the streams have moderate gradients. Streams in the Northern Outer Piedmont have mostly cobble, gravel, and sandy substrates. Streams in the Triassic Basins tend to have mostly sand and clay substrates; however, substrates in the Carolina Slate Belt are composed primarily of boulders and cobbles, and those in the Rolling Coastal Plain have sandy substrates. Natural forest vegetation in the study area typically includes mixed stands of hardwoods and some pines. Land use/land cover (LU/LC) is deciduous forest, pine plantations, pasture, row crops, and hay; cattle and poultry production is common. Landscapes in the RDU have changed from lightly harvested forests interspersed with light residential and agricultural lands to heavily urbanized landscapes with ever smaller parcels of intensively harvested forests. Annual precipitation and runoff in the study area are about 103 and 38 cm per year, respectively.

Methods

Study Design

The RDU comprises the contiguous metropolitan area plus a surrounding 32-km buffer inclusive of the major drainage basins. The conservative 32-km buffer was chosen to incorporate projected urban growth of these metropolitan areas beyond the year 2000 and to evaluate the effects of expanding urban development on aquatic communities. Thirty-nine sites were selected on the basis of a stratified approach designed to control for natural environmental differences (Figure 1). Site selection was intended to exhibit a range of urban LU/LC from low to high and to minimize nested catchments (i.e., spatial autocorrelation). Catchments in rapidly growing regions of the study area were targeted, and natural variability associated with elevation, slope, stream size, substrate, and physiographic region was minimized.

ATtILA

The Analytical Tools Interface for Landscape Assessments (ATtILA) program was used to generate a comprehensive list of landscape metrics. Analytical Tools Interface for Landscape Assessments is an ArcView

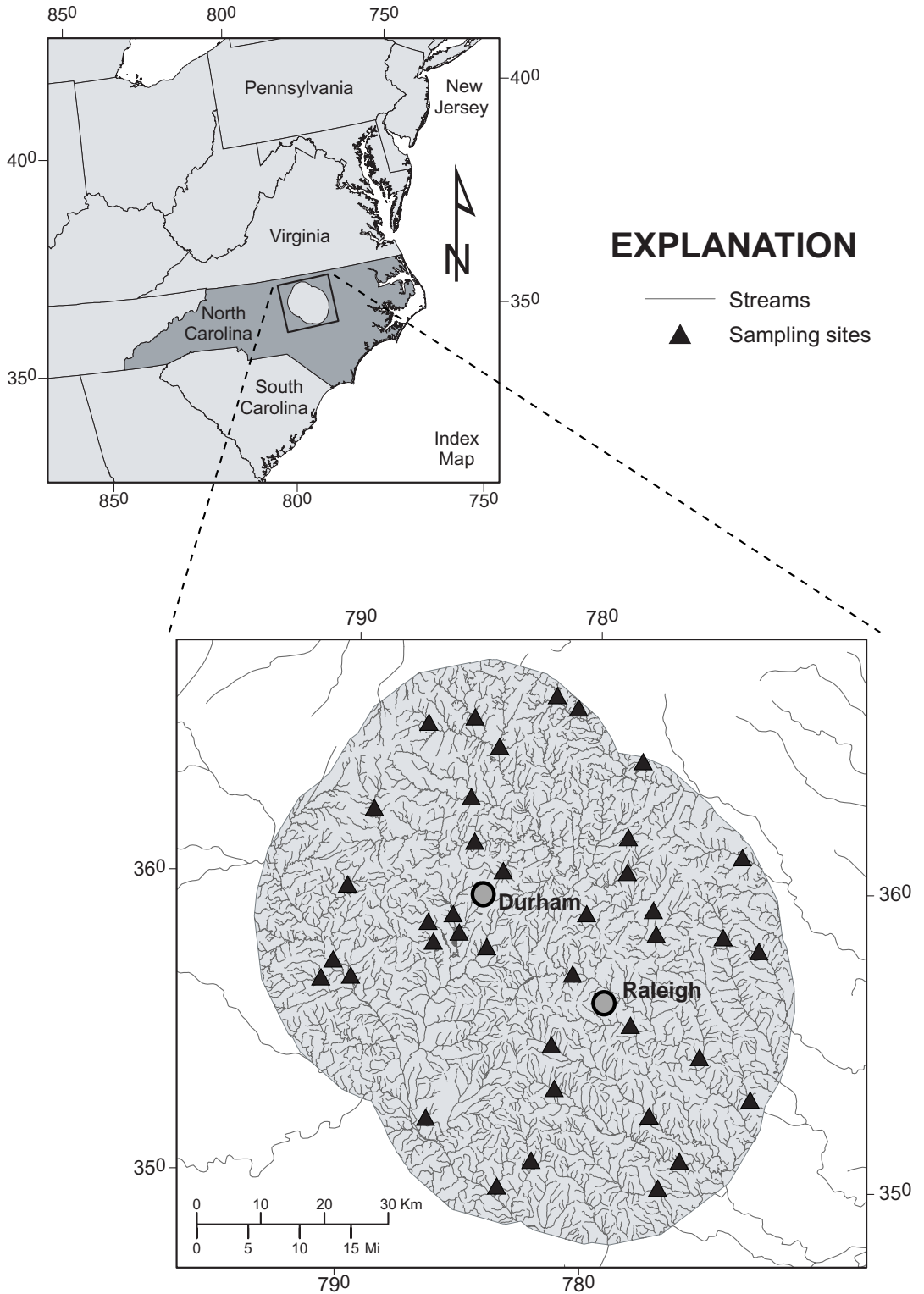


FIGURE 1. Map of the Raleigh-Durham, North Carolina study area.

extension that allows users to calculate and evaluate many types of landscape attributes (Ebert and Wade 2004). Four different metric groups can be calculated by ATtILA, including landscape characteristics, riparian characteristics, human stress, and physical characteristics. Data on LU/LC, elevation, slope, and precipitation are in raster format; stream and road data are lines; and population and census data are in polygon format for ATtILA. The LU/LC is the core input and is represented by specific land-use codes. Coding also can be customized to aggregate similar land-use components for metric calculation. Numerous land-use, diversity, and forest patch metrics were calculated for this study. Forest fragmentation was classified into five categories as defined in Riitters et al. (2000). Many small patches are representative of a fragmented forest, whereas larger patches represent a clumped or contiguous distribution of land uses that provide essential habitat and connected corridors for many interior species. Proportions of land-cover types in the riparian zone also were evaluated (Ebert and Wade 2004). Human-stress metrics, such as amount of impervious surfaces, road density, and population density, were derived from census and roads data. Many physical characteristics, such as elevation, stream slope, density, and length also were included in the assessment. Although not indicators of disturbance, these metrics were used to evaluate whether natural factors accounted for a part of the variation along significant ordination axes.

Digital Data Used in Watershed Assessment

Digital LU/LC, census, and transportation data were aggregated for this study. The 1990s land cover for the RDU was derived from the 1992 National Land Cover Data for North Carolina at 30-m resolution. The 2000 RDU northeastern area land coverage was derived from the 1999 Neuse River basin (NRB) land cover and was used for all areas it covers within the RDU study area. Landsat Enhanced Thematic Mapper Plus was used for the remaining area to perform an unsupervised classification to create 150 clusters. Each cluster was labeled for the land cover it contained and was then combined according to similar land-cover types to create a recoded thematic image. The NRB land-cover classification was recoded to match the desired scheme and resampled at 15–30-m resolution. The study area imagery then was processed into the common Albers Conical Equal Area projection with mosaic processing. The finished product was clipped to the RDU study-area boundary

and used to derive many of the LU/LC and fragmentation metrics.

Fish Assemblage Sampling

Wadeable streams (4–12 m wide) were sampled from 1991 to 2000 using a two-pass procedure with dual backpack electrofishing units and two netters. All collections were made at base flow, and reach length was scaled to a distance 20 times the mean channel width (average length = 180 m; Leopold et al. 1992; Meador et al. 1993; NCDENR 2001a). At each representative reach, all available macro- and microhabitats were sampled (e.g., riffles, pools, runs, snags, undercutts, deadfalls, and quiescent leaf-covered substrates). Block nets were not used. Juvenile and adult fish were collected, and readily identifiable fish were examined for external anomalies (sores, lesions, fin damage, skeletal abnormalities), measured (total length to the nearest 1 mm), and then released. Those fish that were not readily identifiable (e.g., *Notropis* spp.) were preserved in 10% formalin and returned to the laboratory for identification, examination, and total length measurement. Year-0 fish were excluded from all analyses because they pose several challenges when applying index of biotic integrity (IBI) metrics (Angermeier and Karr 1986; Angermeier and Schlosser 1987).

North Carolina Index of Biotic Integrity

The North Carolina index of biotic integrity (NCIBI) is a modification of the IBI initially proposed by Karr (1981) and Karr et al. (1986). The IBI is a quantitative measure that can be used to distinguish among a range of conditions (poor through excellent). It preserves the integrity of the data and incorporates professional judgment (Miller et al. 1988). Although Karr's (1981) original IBI was designed for use in warmwater systems in the Midwest, many regional applications have been presented (e.g., Fausch et al. 1984; Miller et al. 1988; Halliwell et al. 1999; Daniels et al. 2002). Like most biomonitoring tools, the IBI is based on the premise that pristine systems have biological characteristics that can be accurately measured and that departure from these characteristics is directly related to the severity of degradation (Fausch et al. 1990; Bramblett and Fausch 1991). The IBI used to assess North Carolina streams consists of 12 metrics that retain many of Karr's standard components, including species richness and composition, trophic composition, and fish abundance and

condition. Additional information on the development of the NCIBI and species classifications can be found in NCDENR (2001a).

Physical Habitat Assessment

We employed a habitat assessment procedure (NCDENR 2001b), which is a modification of the U.S. Environmental Protection Agency rapid bioassessment protocols (Barbour et al. 1999) to evaluate channel modification, proportion of instream habitat types, type of bottom substrate, pool variety, bank stability, light penetration, and riparian zone conditions at stream reaches. Piedmont streams of moderate to high quality have sticks, leafpacks, snags, undercut banks, root mats, gravels and cobbles with low embeddedness, frequent pools and riffles of varying depths and widths, stable banks with a moderate to full tree canopy, and an intact riparian zone with no or rare breaks in the forest cover. In contrast, Piedmont streams of low to poor quality have a sand substrate, embedded riffles, if present, and incised, sparsely vegetated banks.

Data Analysis

A combination of correlation, regression, and multivariate analyses was used to quantify the variation in the landscape, environmental quality, and fish assemblage structure and to identify possible linkages among these attributes.

Fish assemblages were analyzed on the basis of species composition. Species composition was calculated as relative abundance (i.e., proportion of total catch) of various species, which provides detailed information on species tolerance of environmental conditions and is useful in identifying environmental determinants of assemblage structure (Poff and Allan 1995; Walters et al. 2003). Rare species that accounted for less than 0.1% of overall abundance and that were present in less than 5% of the samples were excluded from multivariate analyses. This approach reduced the number of fish taxa from 76 to 57. However, all fish species were used in the calculation of the NCIBI.

Patterns in fish assemblage structure among the sites were examined using nonmetric multidimensional scaling (NMDS; McCune and Mefford 1999; McCune et al. 2002). The distance measure used was Sorensen, and all NMDS procedures (Kruskal 1964a, 1964b; Mather 1976) were performed using PC-ORD software (McCune and Mefford 1999). Forty runs and 400 iterations were made using real data

with a final instability of 0.00005. Stress was evaluated using a Monte Carlo test ($P < 0.05$) that was based on 50 randomized runs and indicated that the 3-dimensional solution was the best solution and could not have occurred by chance alone. Higher dimensions did little to improve the model. The NMDS analysis allowed us to determine which environmental variables accounted for the majority of the variability in the distribution of fish species in ordination space (e.g., Roy et al. 2003; Walters et al. 2003). Because NMDS axes are acquired independently of gradient length and amount of variance explained, it was important to establish which of the NMDS axes accounted for the primary gradient.

A total of 126 environmental variables were evaluated for this study. We used principal components analysis (PCA; SAS Institute Inc. 1989) in combination with collinearity assessment to isolate a subset of variables that accounted for the greatest proportion of variance while minimizing redundancy. Distributions of all response and explanatory variables used in the PCA analysis were evaluated for normality and were appropriately transformed when necessary. Variables based on amount of land use in a basin were standardized by basin area (percentage data) and arcsine square-root transformed. We conducted PCA on the correlation matrix and evaluated the significance of principal components using the broken stick method (Jackson 1993). The broken stick method is used to determine statistically significant principal component axes by comparing the observed eigenvalues to the eigenvalues from random data. In addition, by using the correlation matrix, we ensured that all the environmental variables contributed equally to the PCA and that the contributions were scale-independent (Legendre and Legendre 1998; Olden and Poff 2003). Loadings of the environmental variables on each significant principle component were used to identify variables that extracted dominant patterns of variation. A Spearman rank correlation matrix (SAS Institute Inc. 1989) of the environmental variables was examined to eliminate redundant variables with an r greater than 0.80. We used a combination of correlation and linear regression analysis to link environmental variables with changes in fish assemblage structure. Axis 2 from the NMDS analysis was correlated with environmental variables, NCIBI metric scores, and fish species. Linear regression analysis was used to directly link changes in sensitive species abundance with the disturbance gradient and to evaluate the relation between urban land use and the NCIBI. The later analyses were used to

exemplify the strength of relations between key land use stressors and fish assemblage structure.

Results

Environmental Disturbance Gradient

The NMDS identified three primary gradients that together accounted for 88% of the variance in the analytical data set. The first and third axes accounted for a significant but small proportion of variance (6% and 15%, respectively) and were not considered for further analysis. The second axis accounted for 67% of the fish assemblage variation.

Data reduction using PCA and correlation analysis reduced the environmental data to nonredundant subsets of 35 and 28 variables, respectively. Of these, only 12 habitat and landscape variables (Table 1) were significantly ($P < 0.05$) related to the extracted NMDS axis 2 scores for the fish assemblage. Nonmetric multidimensional scaling axis 2 correlated

most strongly with specific environmental disturbance variables. In particular, population density, percent urban land cover, and diversity of patch types were positively related, and variables such as proportion of largest forest patch to total forest area, and the percent of forest land were negatively related to NMDS axis 2 (Table 1). Habitat variables most significantly related to NMDS axis 2 include light penetration, riffle habitats, and riparian zone vegetation width (Table 1). Natural environmental factors such as stream slope, length, density, and elevation accounted for an insignificant amount of the variation in NMDS axis 2 scores (Table 1).

Linking Fish Assemblage and Landscape Change

More than 30,700 fish representing 76 species in 12 families were collected (Table 2); however, only 57 species met the censoring criteria and were retained for ordination analysis. The most commonly collected

TABLE 1. Spearman's correlation coefficients of selected environmental variables and NMDS ordination axis 2 (***, $P < 0.001$; **, $P < 0.01$; *, $P < 0.05$).

Variable	NMDS axis 2	Mean	Standard deviation	Minimum	Maximum
Population density in watershed (people/km ²)	0.385*	153.0	3,150.7	8.5	5,509.2
Urban land cover (%)	0.357*	23.36	21.17	2.70	81.13
Area of watershed classified as edge forest (%)	-0.404**	23.27	6.51	6.47	32.70
Amount of largest forest patch to total forest area (%)	-0.460**	76.39	23.13	29.50	99.55
Number of forest patches in watershed	0.371*	264.4	204.5	24.0	1,120.0
Patch Diversity (Shannon-Wiener)	0.554***	0.95	0.17	0.49	1.18
Area of watershed classified as patch forest (%)	0.395*	2.93	1.24	0.57	5.76
Latitude (decimal degrees)	-0.380*	356,758	3,175	352,800	362,000
Forest land cover (%)	-0.408**	58.94	16.30	18.57	87.41
Riffle habitat ^a	-0.302*	5.34	2.74	0.00	10.00
Light penetration ^a	-0.530***	8.11	2.51	2.00	10.00
Density of riparian vegetation ^a	0.347*	4.36	0.91	2.00	6.00
Mean slope (%)	-0.073	4.48	1.22	2.41	7.50
Stream length (km)	0.100	81.85	42.69	9.53	211.39
Stream density (kilometers of stream/km ²)	0.093	0.13	0.01	0.00	0.32
Mean elevation (m)	-0.023	121.90	39.40	56.21	201.69
Bottom substrate ^a	0.071	6.88	1.92	3.00	13.00

^a Nominal variables derived from visual characterization of habitat condition (NCDENR 2001b).

TABLE 2. Occurrence frequency, total abundance, tolerance, and trophic guild of fish species collected in the Raleigh-Durham study area. Only those species included in the ordination analysis are shown.

Family name <i>Scientific name</i>	Common name	Occurrence frequency	Total abundance	Tolerance ^a	Trophic guild ^a
Anguillidae					
<i>Anguilla rostrata</i>	American eel	22	409	Intermediate	Piscivore
Clupeidae					
<i>Dorosoma cepedianum</i>	gizzard shad	4	70	Intermediate	Omnivore
Cyprinidae					
<i>Clinostomus funduloides</i>	rosyside dace	7	48	Intermediate	Insectivore
<i>Cyprinella analostana</i>	satinfish shiner	23	942	Tolerant	Insectivore
<i>C. nivea</i>	whitefin shiner	3	318	Intermediate	Insectivore
<i>Cyprinus carpio</i>	common carp	3	37	Tolerant	Omnivore
<i>Luxilus albeolus</i>	white shiner	29	3,841	Intermediate	Insectivore
<i>Lythrurus matutinus</i>	pinewoods shiner	19	565	Intolerant	Insectivore
<i>Nocomis leptocephalus</i>	bluehead chub	29	4,708	Intermediate	Omnivore
<i>N. raneyi</i>	bull chub	8	386	Intermediate	Omnivore
<i>Notemigonus crysoleucas</i>	golden shiner	12	87	Tolerant	Omnivore
<i>Notropis alborus</i>	whitemouth shiner	2	37	Intermediate	Insectivore
<i>N. altipinnis</i>	highfin shiner	8	284	Intermediate	Insectivore
<i>N. amoenus</i>	comely shiner	8	139	Intermediate	Insectivore
<i>N. cummingsae</i>	dusky shiner	3	77	Intermediate	Insectivore
<i>N. hudsonius</i>	spottail shiner	4	44	Intermediate	Omnivore
<i>N. procne</i>	swallowtail shiner	26	3,324	Intermediate	Insectivore
<i>N. scepcticus</i>	sandbar shiner	2	37	Intermediate	Insectivore
<i>N. volucellus</i>	mimic shiner	5	109	Intolerant	Insectivore
<i>Semotilus atromaculatus</i>	creek chub	15	149	Tolerant	Insectivore
Catostomidae					
<i>Catostomus commersonii</i>	white sucker	9	27	Tolerant	Omnivore
<i>Erimyzon oblongus</i>	creek chubsucker	25	249	Intermediate	Omnivore
<i>Hypentelium nigricans</i>	northern hog sucker	15	183	Intermediate	Insectivore
<i>Moxostoma collapsum</i>	notchlip redhorse	16	182	Intermediate	Insectivore
<i>M. pappillosum</i>	V-lip redhorse	7	39	Intermediate	Insectivore
<i>M. cervinum</i> (sometimes called <i>Scartomyzon cervinus</i>)	blacktip jumprock (formerly black jumprock)	11	203	Intermediate	Insectivore
Ictaluridae					
<i>Ameiurus catus</i>	white catfish	2	7	Tolerant	Omnivore
<i>A. natalis</i>	yellow bullhead	25	104	Tolerant	Omnivore
<i>A. nebulosus</i>	brown bullhead	10	16	Tolerant	Omnivore
<i>A. platycephalus</i>	flat bullhead	11	39	Tolerant	Insectivore
<i>Ictalurus punctatus</i>	channel catfish	8	20	Intermediate	Omnivore
<i>Noturus insignis</i>	margined madtom	30	639	Intermediate	Insectivore
Esocidae					
<i>Esox americanus</i>	redfin pickerel	19	95	Intermediate	Piscivore
Umbridae					
<i>Umbra pygmaea</i>	eastern mudminnow	2	11	Intermediate	Insectivore
Aphredoderidae					
<i>Aphredoderus sayanus</i>	pirate perch	24	137	Intermediate	Insectivore
Fundulidae					
<i>Fundulus rathbuni</i>	speckled killifish	7	195	Intermediate	Insectivore
Poeciliidae					
<i>Gambusia holbrooki</i>	eastern mosquitofish	22	384	Tolerant	Insectivore

TABLE 2. Continued.

Family name <i>Scientific name</i>	Common name	Occurrence frequency	Total abundance	Tolerance ^a	Trophic guild ^a
Centrarchidae					
<i>Ambloplites cavifrons</i>	Roanoke bass	5	89	Intermediate	Piscivore
<i>Centrarchus macropterus</i>	flier	7	21	Intermediate	Insectivore
<i>Enneacanthus gloriosus</i>	bluespotted sunfish	9	28	Intermediate	Insectivore
<i>Lepomis auritus</i>	redbreast sunfish	38	4,727	Tolerant	Insectivore
<i>L. cyanellus</i>	green sunfish	31	688	Tolerant	Insectivore
<i>L. gibbosus</i>	pumpkinseed	25	347	Intermediate	Insectivore
<i>L. gulosus</i>	warmouth	22	67	Intermediate	Insectivore
<i>L. macrochirus</i>	bluegill	36	3,117	Intermediate	Insectivore
<i>L. microlophus</i>	redear sunfish	16	191	Intermediate	Insectivore
<i>Lepomis</i> sp.	Hybrid sunfish	8	30	Tolerant	Insectivore
<i>Micropterus salmoides</i>	largemouth bass	28	172	Intermediate	Piscivore
<i>Pomoxis nigromaculatus</i>	black crappie	9	30	Intermediate	Piscivore
Percidae					
<i>Etkeostoma collis</i>	Carolina darter	2	11	Intermediate	Insectivore
<i>E. flabellare</i>	fantail darter	6	232	Intermediate	Insectivore
<i>E. nigrum</i>	johnny darter	19	1,136	Intermediate	Insectivore
<i>E. olmstedii</i>	tessellated darter	20	581	Intermediate	Insectivore
<i>E. vitreum</i>	glassy darter	12	84	Intermediate	Insectivore
<i>Perca flavescens</i>	yellow perch	6	80	Intermediate	Piscivore
<i>Percina nevisense</i>	chainback darter	18	89	Intolerant	Insectivore
<i>P. roanoka</i>	Roanoke darter	22	856	Intolerant	Insectivore

^a North Carolina Department of Environment and Natural Resources 2001a.

family was the Centrarchidae. Redbreast sunfish was the most abundant fish ($N = 4,727$) and was found at all but one site. Bluegill and green sunfish were collected at 36 and 31 sites, respectively (Table 2). In addition, two common cyprinids (white shiner and bluehead chub) occurred at 29 of the sites sampled. Highly significant relations were found between NMDS axis 2 and the abundance of redbreast sunfish ($r^2 = 0.70$) and bluehead chub ($r^2 = 0.65$; Figure 2). In total, the abundance of 11 fish species was significantly correlated with NMDS axis 2 (Table 3).

The NCIBI was significantly related to the amount of watershed urbanization ($N = 39$, $r^2 = 0.44$, $P < 0.0001$; Figure 3). Sites with the highest NCIBI scores typically fell in watersheds with a high percentage of forest and a low percentage of urban land. Four of the NCIBI component metrics were significantly related to NMDS axis 2, including percentage of tolerant individuals, percentage of omnivorous and herbivorous individuals, percentage of insectivorous individuals, and percentage of piscivorous individuals (Table 4).

Discussion

Our analyses identified ecologically relevant landscape and habitat factors that were directly related to changes in fish assemblage structure across a disturbance gradient in North Carolina streams (Table 1). Many of the fragmentation, patch, and riparian metrics accounted for a significant amount of the variability in fish assemblage structure and were important in differentiating fish assemblages along a disturbance gradient. For example, the percentage of the watershed classified as patch forest, the diversity of patch types, and the number of forest patches in the watershed were significant indicators of watershed change and were directly related to NMDS axis 2. In contrast, factors such as the proportion of largest forest patch to total forest area, the percent of the watershed classified as forest, and the amount of riffle habitat were inversely related to NMDS axis 2. The direct link from forest fragmentation, habitat loss, and changes in patch dynamics to ecological consequence for many terrestrial species is well established (e.g., Saunders et al. 1991; Andren 1994; Debinski and Holt 2000; Trombulak

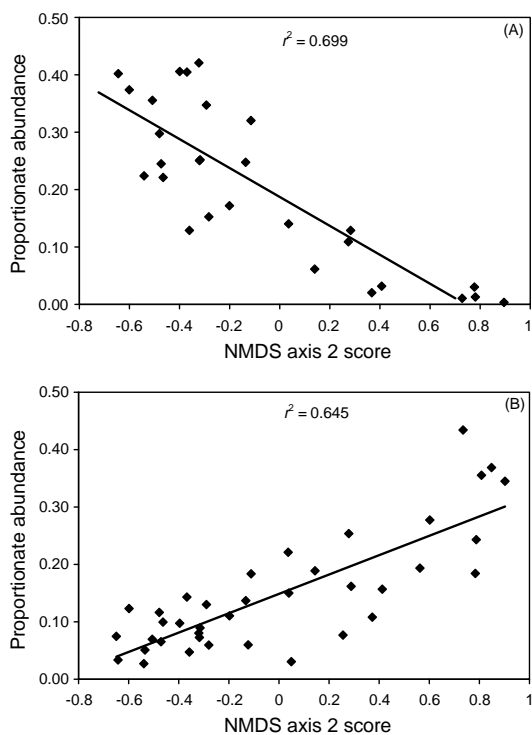


FIGURE 2. Regression relation between NMDS axis 2 scores and proportionate abundance of bluehead chub (A) and redbreast sunfish (B).

and Frissell 2000). However, lotic systems also are linked directly and indirectly to terrestrial ecosystem fragmentation (Conroy et al. 2003).

The same anthropogenic processes that affect terrestrial species by reducing landscape connectivity,

TABLE 3. Significant correlations (Spearman's rho) between relative abundance of fish species collected from the Raleigh-Durham study area and NMDS ordination axis 2. Tolerance classes are defined in Table 2.

Species	<i>r</i>	<i>P</i>
Bluehead chub	-0.616	<0.001
Redbreast sunfish	0.697	<0.001
American eel	0.492	0.001
White shiner	-0.484	0.002
Bull chub	0.466	0.003
Tessellated darter	0.419	0.008
Satinfish shiner	0.379	0.017
Dusky shiner	0.376	0.018
Bluespotted sunfish	0.344	0.032
Speckled killifish	-0.336	0.036
Largemouth bass	0.326	0.043

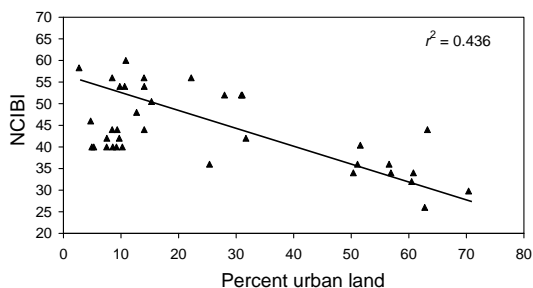


FIGURE 3. Regression relation between percent urban land and the North Carolina index of biotic integrity (NCIBI).

habitat availability, and increasing patch diversity can have a quantifiable effect on aquatic assemblage structure. One possible scenario is that as fragmentation increases, forested watershed and riparian areas become more and more discontinuous and patchy. This process results in the loss of protective forested areas that can alter riparian conditions by reducing buffering capacity and ultimately modifying water quality and aquatic assemblage structure.

Results of previous investigations indicate that aquatic-assemblage health is affected negatively by the alteration of forested and agricultural lands (Benke et al. 1981; Garie and McIntosh 1986; Jones and Clark 1987; Kennen 1999). Roth et al. (1996) indicate that the amount of forested land in a basin was posi-

TABLE 4. Correlation coefficients (Spearman's rho) between NCIBI metrics and NMDS ordination axis 2. Metrics with a significant correlation ($P < 0.05$) with NMDS axis 2 are in bold.

Metric	<i>r</i>	<i>P</i>
Number of species	0.338	0.838
Number of fish	-0.184	0.262
Number of darter species	-0.027	0.872
Number of sunfish species	-0.049	0.766
Number of sucker species	-0.135	0.411
Number of intolerant species	0.021	0.899
Percentage of tolerant individuals	0.609	<0.001
Percentage of omnivorous and herbivorous individuals	-0.500	0.001
Percentage of insectivorous individuals	0.463	0.003
Percentage of piscivorous individuals	-0.467	0.003
Percentage of diseased fish	-0.053	0.341
Percentage of species with multiple age-groups	-0.231	0.156

tively related to fish-assemblage IBI scores. May et al. (1997) report that stream buffer width, vegetative condition, and longitudinal connectedness of buffer vegetation were altered as the level of development in a watershed increased. Urbanization fragments landscapes and riparian corridors and greatly reduces the amount of mature vegetation in watersheds. In addition, soil erosion during and after development activities affects vegetative cover by promoting the movement of invasive species, increasing pathogens, inhibiting vegetative regeneration, and increasing foliar damage (Schlosser and Karr 1981; Reid 1993). Consequently, fragmentation of forested areas and a concomitant reduction in canopy cover can result in changes in stream temperature and habitat structure, which can alter the structure and function of the fish assemblage.

In the RDU, sensitive species such as the bluehead chub, which typically occur in clear creeks and small rivers with medium to fast current and a substrate composed of rock, gravel, or sand (Rohde et al. 1994), decreased in abundance across the disturbance gradient (Figure 2). In contrast, the opportunistic and tolerant redbreast sunfish, which occurs throughout the mid-Atlantic region in lotic systems (Rohde et al. 1994), increased in abundance with increasing watershed disturbance. Abundance of other tolerant and intermediate tolerant species, such as the tessellated darter *Etheostoma olmstedii*, satinfish shiner *Cyprinella analostana*, and the bull chub *Nocomis biguttatus*, increased along the disturbance gradient (Table 3). In contrast, intermediate species such as the white shiner and speckled killifish *Fundulus rathbuni* decreased. In addition, the NCIBI metric percentage of tolerant individuals was significantly related to the disturbance gradient (Table 4), further supporting the linkage between landscape change and watershed disturbance.

In this study, fragmentation of the landscape associated with the loss of forested lands, changes in patch dynamics, and an increase in developed lands appeared to promote highly tolerant aquatic species. In contrast, species sensitive to environmental disturbance became much less prevalent (Figure 2; Table 3). This is only one of many processes associated with landscape disturbance that are known to degrade stream condition. Other pressures that can affect native species include impoundments, streamflow regulation, and water-resource development. These practices can have profound effects on the structure, function, and resilience of lotic systems (Conroy et al. 2003), yet often are essential to meet the water needs of a growing population. Consequently, changes in

aquatic species composition throughout this and other regions are the legacy of historical management practices.

Linking the effects of landscape fragmentation in a spatial context to ecological consequences is often a difficult task for aquatic ecologists. Moreover, translating the effects into procedures that can be used in management is even more complicated because ecological complexity tends to blur results with patterns and processes that can be difficult to distinguish because of the high level of variability inherent in aquatic ecological data. This tendency is apparent in these data because the strongest relations with metrics and landscape and habitat variables, although significant, account for only a part of the overall variability in the fish assemblage data. However, management of forest and riparian fragmentation may become necessary if ecosystem function is to be maintained, especially for aquatic systems that exhibit a response to landscape change. Additionally, an increase in patch diversity indicates that greater attention be given to the prevention of perforated landscape conditions, especially those along the stream corridor that may result in modification in aquatic-assemblage structure. Some development practices promote habitat fragmentation and this process ultimately may jeopardize the conservation goals of many state agencies (Keitt et al. 1997) by creating smaller and smaller noncontiguous forested areas in watersheds. This approach is in direct contrast to "smart growth" efforts that support "green infrastructure" (that is, large conservation areas, connected forest and riparian corridors, and green belts) that promote the maintenance and restoration of intact forests and riparian areas that have consistently been shown to be protective of stream integrity (e.g., Karr and Gorman 1975; Kleiss et al. 1989; Leavitt 1998).

The results of this study indicate that smart growth policies complemented by low-impact development and open-space preservation appear to promote the retention of sensitive species. Watershed-level low-impact development patterns likely result in improved sediment mitigation, hydrologic stability, and biotic integrity because these approaches protect larger tracts of land and preserve critical ecological functions (i.e., they maximize open space and minimize impervious cover). Thus, natural landscapes that maintain contiguous areas of forest in a watershed and along the stream corridor are important for the maintenance of ecosystem health. Nonproactive growth policies that promote or allow scattered development eventually replace naturally vegetated areas with impervious surfaces and increase the volume and frequency of

stormwater runoff following a rain event. The resulting hydrologic alteration is but one of the many consequences of urban growth that have been linked to changes in the composition, structure, and function of aquatic systems (see Ward and Stanford 1989; Richter et al. 1996; Bunn and Arthington 2002; Wang and Lyons 2003). Structurally engineered stormwater-management measures (e.g., swales, retention basins) often have been used to control the environmental impact of new development; however, few have demonstrated the long-term ability to control the movement of contaminants or maintain ecosystem integrity (Coffman and Smith 1996; Maxted and Shaver 1998). These results indicate that for preservation of open land to be most effective as a management tool, the preserved forest and riparian lands not only need to be large enough to serve an environmental function, but also should be connected. To manage aquatic ecosystem integrity in a comprehensive, integrated manner, resource managers will need to consider restoration of the hydrologic regime, protection of ground-and surface water quality by minimizing disturbance to the riparian corridor, and preservation of landscape function by maximizing the conservation of contiguous tracts of forested lands and vegetative cover in watersheds.

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References

- Andren, H. 1994. Effects of habitat fragmentation on birds and mammals in landscapes with different proportions of suitable habitat. *Oikos* 71:344–366.
- Angermeier, P. L., and J. R. Karr. 1986. Applying an index of biotic integrity based on stream-fish communities: considerations in sampling and interpretation. *North American Journal of Fisheries Management* 6:418–429.
- Angermeier, P. L., and I. J. Schlosser. 1987. Assessing biotic integrity of the fish community in a small Illinois stream. *North American Journal of Fisheries Management* 7:331–338.
- Barbour, M. T., J. Gerritsen, B. D. Snyder, and J. B. Stribling. 1999. Rapid bioassessment protocols for use in streams and Wadeable rivers: periphyton, benthic macroinvertebrates, and fish, 2nd edition. U.S. Environmental Protection Agency, EPA 841-B-99-002, Washington, D.C.
- Benke, A. C., G. E. Willke, F. K. Parrish, and D. L. Stites. 1981. Effects of urbanization on stream ecosystems. Georgia Institute of Technology, Report ERC07-81, Atlanta.
- Booth, D. B., and C. R. Jackson. 1997. Urbanization of aquatic systems: degradation thresholds, stormwater detection, and the limits of migration. *Journal of the American Water Resources Association* 33:1077–1090.
- Bramblett, R. G., and K. D. Fausch. 1991. Variable fish communities and the index of biotic integrity in a western Great Plains river. *Transactions of the American Fisheries Society* 120:752–769.
- Brooker, L. B., and P. Cale. 1999. Animal dispersal in fragmented habitat: measuring habitat connectivity, corridor use, and dispersal mortality. *Conservation Ecology* 3(1):4.
- Bunn, S. E., and A. H. Arthington. 2002. Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity. *Environmental Management* 30:492–507.
- Coffman, L. S., and J. Smith. 1996. Environmentally sensitive low-impact development. Proceedings of Watershed 96, Water Environment Federation, June 8–12, Baltimore, Maryland.
- Conroy, M. J., C. R. Allen, J. T. Peterson, L. Pritchard Jr., and C. T. Moore. 2003. Landscape change in the southern Piedmont: challenges, solutions, and uncertainty across scales. *Conservation Ecology* 8(2):3.
- Daniels, R. A., K. Riva-Murray, D. B. Halliwell, D. L. Vana-Miller, and M. D. Bilger. 2002. An index of biological integrity for northern mid-Atlantic slope drainages. *Transactions of the American Fisheries Society* 131:1044–1060.
- Debinski, D. M., and R. D. Holt. 2000. A survey and overview of habitat fragmentation experiments. *Conservation Biology* 14:342–353.
- D'Eon, R. G., S. M. Glenn, I. Parfitt, and M-J Fortin. 2002. Landscape connectivity as a function of scale and organism vagility in a real forested landscape. *Conservation Ecology* 6(2):10.
- Dodd, C. K., Jr., and L. L. Smith. 2003. Habitat destruc-

- tion and alteration: historical trends and future prospects for amphibians. Pages 94–112 in R. D. Semlitsch, editor. *Amphibian Conservation*, Smithsonian Institution Press, Washington, D.C.
- Ebert, D. W., and T. G. Wade. 2004. Analytical Tools Interface for Landscape Assessment (ATtILA), user manual, Version 3.0. U.S. Environmental Protection Agency, Office of Research and Development, National Exposure Research Laboratory, Environmental Sciences Division, Landscape Ecology Branch, Las Vegas, Nevada.
- Ewing, R., R. Pendall, and D. Chen. 2002. Measuring sprawl and its impact, Smart Growth America, Washington, D.C. Available: <http://www.smartgrowthamerica.org/sprawlindex/MeasuringSprawl.PDF> (February 2004)
- Fausch, K. D., J. Lyons, J. R. Karr, and P. L. Angermeier. 1990. Fish communities as indicators of environmental degradation. Pages 123–144 in S. M. Adams, editor. *Biological indicators of stress in fish*. American Fisheries Society, Symposium 8, Bethesda, Maryland.
- Fausch, K. D., J. R. Karr, and P. R. Yant. 1984. Regional application of an index of biotic integrity based on stream fish communities. *Transactions of the American Fisheries Society* 113:39–55.
- Finkenbine, J. K., J. W. Atwater, and D. S. Mavinic. 2000. Stream health after urbanization. *Journal of the American Water Resources Association* 36:1149–1160.
- Frissell, C. A., W. J. Liss, C. E. Warren, C. E., and M. D. Hurley. 1986. A hierarchical framework for stream habitat classification: viewing streams in a watershed context. *Environmental Management* 10:199–214.
- Garie, H. L., and A. McIntosh. 1986. Distribution of benthic macroinvertebrates in a stream exposed to urban runoff. *Water Resources Bulletin* 22:447–455.
- Griffith, G., J. Omernik, and J. Comstock. 2002. Ecoregions of North Carolina. U.S. Environmental Protection Agency, Office of Research and Development, NHEERL, Western Ecology Division, Corvallis, Oregon.
- Halliwel, D. B., R. W. Langdon, R. A. Daniels, J. P. Kurtenbach, and R. A. Jacobson. 1999. Classification of freshwater fish species of the northeastern United States for use in the development of indices of biological integrity, with regional applications. Pages 301–333 in T. P. Simon, editor. *Assessing the sustainability and biological integrity of water resources using fish communities*. CRC Press, Boca Raton, Florida.
- Jackson, D. A. 1993. Stopping rules in principal components analysis: a comparison of heuristical and statistical approaches. *Ecology* 74:2204–2214.
- Jensen, S. E., and W. S. Platts. 1990. Restoration of degraded riverine/riparian habitat in the Great Basin and Snake River region. Pages 367–404 in J. A. Kusler and M. E. Kentula, editors. *Wetland creation and restoration: the status and the science*, Island Press, Washington, D.C.
- Jones, R. C., and C. C. Clark. 1987. Impact of watershed urbanization on stream insect communities. *Water Resources Bulletin* 23:1047–1055.
- Karr, J. R. 1981. Assessment of biotic integrity using fish communities. *Fisheries* 6(6):21–27.
- Karr, J. R., K. D. Fausch, P. L. Angermeier, P. R. Yant, and I. J. Schlosser. 1986. *Assessing biological integrity in running water: a method and its rationale*. Illinois Natural History Survey Special Publication 5, Champaign.
- Karr, J. R., and O. T. Gorman. 1975. Effects of land treatment on the aquatic environment. Pages 4–1 to 4–18 in U.S. Environmental Protection Agency non-point source pollution seminar, Report 905/9–75-007, Washington, D.C.
- Keitt, T. H., L. U. Dean, and B. T. Milne. 1997. Detecting critical scales in fragmented landscapes. *Conservation Ecology* 1(1):4.
- Kennen, J. G. 1999. Relation of macroinvertebrate community impairment to catchment characteristics in New Jersey streams. *Journal of the American Water Resources Association* 35:939–955.
- Kennen, J. G., and M. A. Ayers. 2002. Relation of environmental characteristics to the composition of aquatic assemblages along a gradient of urban land use in New Jersey, 1996–98. U.S. Geological Survey Water-Resources Investigations Report 02–4069, Trenton, New Jersey.
- Klein, R. D. 1979. Urbanization and stream quality impairment. *Water Resources Bulletin* 15:948–963.
- Kleiss, B. A., E. E. Morris, J. F. Nix, and J. W. Barko. 1989. Modification of riverine water quality by an adjacent bottomland hardwood wetland. Pages 429–438 in D. W. Fisk, editor. *Proceedings of wetlands: concerns and successes*. American Water Resources Association, TPS 89–3, Bethesda, Maryland.
- Kruskal, J. B. 1964a. Multidimensional scaling by optimizing goodness of fit to nonmetric hypothesis. *Psychometrika* 29:1–27.
- Kruskal, J. B. 1964b. Nonmetric multidimensional scaling: a numerical method. *Psychometrika* 29:115–129.
- Lammert, M., and J. D. Allan. 1999. Assessing biotic integrity of streams: effects of scale in measuring the influence of land use/cover and habitat structure on fish and macroinvertebrates. *Environmental Management* 23:257–270.

- Leavitt, J. 1998. The functions of riparian buffers in urban watersheds. Master's thesis. University of Washington, Seattle.
- Legendre, P., and L. Legendre. 1998. Numerical ecology. Elsevier, Amsterdam.
- Leopold, L. B., M. G. Wolman, and J. P. Miller. 1992. Fluvial process in geomorphology. Dover Publications, Inc., New York.
- Mather, P. M. 1976. Computational methods of multivariate analysis in physical geography. Wiley, London.
- Maxted, J. R., and E. Shaver. 1998. The use of retention basins to mitigate stormwater impacts on aquatic life. Pages 494–512, *in*, L. A. Roesner, editor. Effects of watershed development and management on aquatic ecosystems. American Society of Civil Engineers, New York.
- May, C. W., R. R. Horner, J. B. Karr, B. W. Mar, and E. G. Welch. 1997. Effects of urbanization on small streams in the Puget Sound lowland ecoregion. *Watershed Protection Techniques* 2:483–493.
- McCoy, E. D., and H. R. Mushinsky. 2000. Effects of fragmentation on the richness of vertebrates in the Florida scrub habitat. *Ecology* 75:446–457.
- McCune, B., J. B. Grace, and D. L. Urban. 2002. Analysis of ecological communities. MjM Software Design, Glenden Beach, Oregon.
- McCune, B., and M. J. Mefford. 1999. PC-ORD – multivariate analysis of ecological data Version 4.25. MjM Software, Glenden Beach, Oregon.
- Meador, M. R., T. R. Cuffney, and M. E. Gurtz. 1993. Methods for characterizing stream habitat as part of the National Water-Quality Assessment Program. U.S. Geological Survey Open-File Report 93-408, Raleigh, North Carolina.
- Miller, D. L., P. M. Leonard, R. M. Hughes, J. R. Karr, P. B. Moyle, L.H. Schrader, B. A. Thompson, R. A. Daniels, K. D. Fausch, G. A. Fitzhugh, J. R. Gammon, D. B. Halliwell, P. L. Angermier, and D. O. Orth. 1988. Regional application of an index of biotic integrity for use in water resource management. *Fisheries* 13(5):12–20.
- NCDENR (North Carolina Department of Environment and Natural Resources). 2001a. Standard operating procedure: biological monitoring, stream fish community assessment and fish tissue. North Carolina Department of Environment and Natural Resources, Raleigh. Available: <http://www.esb.enr.state.nc.us/BAU.html> (December 2003)
- NCDENR (North Carolina Department of Environment and Natural Resources). 2001b. Standard operating procedures for benthic macroinvertebrates. North Carolina Department of Environment and Natural Resources, Raleigh. Available: <http://www.esb.enr.state.nc.us/BAU.html> (December 2003)
- Noss, R. F. 1987. Corridors in real landscapes: a reply to Simerloff and Cox. *Conservation Biology* 1:159–164.
- Olden, J. D., and N. L. Poff. 2003. Redundancy and the choice of hydrologic indices for characterizing streamflow regimes. *River Research and Applications* 19:101–121.
- Poff, N. L., and J. D. Allan. 1995. Functional organization of stream fish assemblages in relation to hydrological variability. *Ecology* 76:606–627.
- Reid, L. 1993. Research and cumulative watershed effects. U.S. Forest Service, Pacific Southwest Research Station, General Technical Report PSW-GTR-141, Albany, California.
- Richards, C., R. J. Haro, L. B. Johnson, and G. E. Host. 1997. Catchment and reach-scale properties as indicators of macroinvertebrate species traits. *Freshwater Biology* 37:219–230.
- Richards, C., and G. E. Host. 1994. Examining land influences on stream habitats and macroinvertebrates: a GIS approach. *Water Resources Bulletin* 30:729–738.
- Richter, B. D., J. V. Baumgartner, J. Powell, and D. B. Braun. 1996. A method for assessing hydrologic alteration within ecosystems. *Conservation Biology* 10:1163–1174.
- Riitters, K., J. Wickham, R. O'Neill, B. Jones, and E. Smith. 2000. Global-scale patterns of forest fragmentation. *Conservation Ecology* 4(2):3.
- Rohde, F. C., R. G. Arndt, D. G. Lindquist, and J. F. Parnell. 1994. Freshwater fishes of the Carolinas, Virginia, Maryland, and Delaware. The University of North Carolina Press, Chapel Hill.
- Roth, N. E., J. D. Allan, and D. E. Erickson. 1996. Landscape influences on stream biotic integrity assessed at multiple spatial scales. *Landscape Ecology* 11:141–156.
- Roy, A. H., A. D. Rosemond, M. J. Paul, D. S. Leigh, and J. B. Wallace. 2003. Stream macroinvertebrate response to catchment urbanisation (Georgia, U.S.A.). *Freshwater Biology* 48:1–18.
- SAS Institute Inc. 1989. SAS/STAT® users guide, version 6, 4th edition, volume 2. SAS Institute Inc., Cary, North Carolina.
- Saunders, D. A., R. J. Hobbs, and C. R Margules. 1991. Biological consequences of ecosystem fragmentation: a review. *Conservation Biology* 5:18–32.
- Schlosser, I. J., and J. R. Karr. 1981. Riparian vegetation and channel morphology impact on spatial patterns of water quality in agricultural watersheds. *Environmental Management* 5:233–243.
- Schueler, T. 1994. The importance of imperviousness. *Watershed Protection Techniques* 1:100–111.

- Trombulak, S. C., and C. A. Frissell. 2000. Review of ecological effects of roads on terrestrial and aquatic communities. *Conservation Biology* 14:18–30.
- Walters, D. M., D. S. Leigh, M. C. Freeman, B. J. Freeman, B. J., and C. M. Pringle. 2003. Geomorphology and fish assemblages in a Piedmont river basin, U.S.A. *Freshwater Biology* 48:1950–1970.
- Wang, L., J. Lyons, P. Kanehl, R. Bannerman, and E. Emmons. 2000. Watershed urbanization and changes in fish communities in southeastern Wisconsin streams. *Journal of the American Water Resources Association* 36:1173–1189.
- Wang, L., J. Lyons, P. Kanehl, and R. Bannerman. 2001. Impacts of urbanization on stream habitat and fish across multiple spatial scales. *Environmental Management* 28:255–266.
- Wang, L., and J. Lyons. 2003. Fish and benthic macroinvertebrate assemblages as indicators of stream degradation in urbanizing watersheds. Pages 227–249 *in* T. P. Simon, editor. *Biological response signatures: indicator patterns using aquatic communities*. CRC Press, Boca Raton, Florida.
- Ward, J. V., and J. A. Stanford. 1989. *Riverine ecosystems: the influence of man on catchment dynamics and fish ecology*. Canadian Special Publications in Fisheries and Aquatic Sciences 106:56–64.