DOI: 10.1007/s00267-002-2805-5

Assessing Water Quality at Large Geographic Scales: Relations Among Land Use, Water Physicochemistry, Riparian Condition, and Fish Community Structure

MICHAEL R. MEADOR
US Geological Survey
12201 Sunrise Valley Drive, MS 413
Reston, Virginia 20192, USA

ROBERT M. GOLDSTEIN
US Geological Survey
2280 Woodale Drive
Mounds View, Minnesota 55112, USA

ABSTRACT / Data collected from 172 sites in 20 major river basins between 1993 and 1995 as part of the US Geological Survey's National Water-Quality Assessment Program were analyzed to assess relations among basinwide land use (agriculture, forest, urban, range), water physicochemistry, riparian condition, and fish community structure. A multimetric approach was used to develop regionally referenced indices of fish community and riparian condition. Across large geographic areas, decreased riparian condition was associated with water-quality constituents indicative of nonpoint source inputs—total nitrogen and suspended sediment and basinwide urban land use. Decreased fish community condition was associated with increases in total dissolved solids and rangeland use and decreases in riparian condition and agricultural land use. Fish community condition was relatively high even in areas where agricultural land use was relatively high (>50% of the basin). Although agricultural land use can have deleterious effects on fish communities, the results of this study suggest that other factors also may be important, including practices that regulate the delivery of nutrients, suspended sediments, and total dissolved solids into streams. Across large geographic scales, measures of water physicochemistry may be better indicators of fish community condition than basinwide land use. Whereas numerous studies have indicated that riparian restorations are successful in specific cases, this analysis suggests the universal importance of riparian zones to the maintenance and restoration of diverse fish communities in streams.

In order to maintain and improve water quality, managers have an increasing need to understand the relations among basin land use, riparian zones, and stream ecosystems (Lammert and Allan 1999, Wang and others 2001). Studies have shown that stream ecosystems can be influenced by land use at regional or broad geographic scales (Richards and others 1996). However, land use within a basin and local riparian zone condition can interact to affect the severity of water-quality degradation. Riparian buffers of undisturbed vegetation have been shown to be effective in reducing nutrient and sediment loading to streams and are suggested for consideration in water-quality restoration programs (Karr and Schlosser 1978, Osborne and Kovacic 1993, Barling and Moore 1994). Several studies have indicated that basinwide landscape factors are better indicators of in-stream biotic integrity than local stream conditions (Roth and others 1996, Allan and others 1997, Wang and others 1997, Harding and others 1998), and landscape factors have been correlated with water quality (Johnson and others 1997). Richards and others (1996) proposed that landscape characteristics of geology and land use may be more important than stream buffers for stream restoration. Conversely, Lammert and Allen (1999) found local, site-specific stream conditions more important than basinwide landscape factors for explaining biological conditions. Marsh-Matthews and Matthews (2000), in a study of streams from Iowa to south Texas, reported that both broad geographic factors (particularly latitude) and riparian characteristics explained significant variations in fish community composition. Stauffer and others (2000) determined that basinwide soils and riparian vegetative cover were factors that accounted for a significant portion of the variance in index of biotic integrity (IBI) scores and fish species richness in agricultural streams in Minnesota. What, then, can be concluded regarding relations among land use, water

KEY WORDS: Water quality; Fish community; Riparian zone; Land use; Ecological indicators

*Author to whom correspondence should be addressed; e-mail: mrmeador@usgs.gov

Table 1. List of 20 NAWQA study units sampled between 1993 and 1995

NAWQA study unit	States in study unit	Study unit abbreviation	Number of sites
Apalachicola-Chattahoochee-Flint River Basin	AL, FL, GA	ACFB	6
Albemarle-Pamlico Drainage	NC, VA	ALBE	10
Central Columbia Plateau	ID, WA	CCPT	6
Central Nebraska Basins	NE	CNBR	8
Connecticut, Housatonic, and Thames River Basins	CT, MA, NH, NY, RI, VT	CONN	10
Georgia-Florida Coastal Plain	GA, FL	GAFL	6
Hudson River Basin	NY, CT, MA, NJ, VT	HDSN	7
Lower Susquehanna River Basin	PA, MD	LSUS	7
Nevada Basin and Range	NV, CA	NVBR	7
Ozark Plateaus	AR, KS, MO, OK	OZRK	13
Potomac River Basin	DC, MD, PA, VA, WV	POTO	9
Red River of the North	MN, ND, SD	REDN	9
Rio Grande Valley	CO, NM, TX	RIOG	8
San Joaquin-Tulare	CA	SANJ	8
South Platte River Basin	CO, NE, WY	SPLT	10
Trinity River Basin	TX	TRIN	10
Upper Snake River Basin	ID, MT, NV, UT, WY	USNK	12
White River Basin	IN	WHIT	11
Willamette Basin	OR	WILL	7
Western Lake Michigan Drainage	MI, WI	WMIC	8

physicochemistry, riparian condition, and fish community structure at large geographic scales? Are patterns evident that can lead to generalizations about the relative influences of basinwide land use and site-specific water physicochemistry, and riparian conditions on fish communities?

Data from the US Geological Survey's (USGS) National Water-Quality Assessment (NAWQA) program provided the opportunity to examine relations among basin land use, water physicochemistry, riparian conditions, and fish community structure across the United States. Studies that have examined relations between land use and fish community structure generally have been limited in geographic scale to one or a few adjacent watersheds or a single river system. The advantages of small-geographic-scale studies include consistency of environmental and climatic conditions, and relatively limited variability in ichthyofauna. Examination of fish community relations with land use or other anthropogenic variables across multiple regions of the United States is lacking. In addition, analytical interpretation is challenged by broad-scale variability in environmental conditions and zoogeographic distribution of fishes.

We hypothesized that even at a broad geographic scale, fish community condition, riparian conditions, and water physicochemistry are affected by agricultural and urban land uses. Specific objectives of this investigation include: (1) assessment of relations among land use, water physicochemistry, and measures of riparian and fish community condition; (2) examination of relations between the amount of basinwide land use (agricultural, forested, urban, and range) and riparian and fish community condition; and (3) examination of relations between riparian and fish community condition across land uses.

National Water-Quality Assessment Program

The design of the NAWQA program focuses on 59 major river basins (study units) across the United States (Gilliom and others 1995). Twenty study units were sampled from 1993 to 1995 (Table 1) and data collected from these study units were analyzed for this study. The NAWQA study units were selected based on several factors, including human population and water use, importance of water-quality issues, and geographic distribution (Gilliom and others 1998).

Sampling Sites

A total of 226 stream sites were sampled in the study units from 1993 to 1995. Two types of sites were sampled—one type represented streams that drain basins with a single dominant land use, and the second type represented streams that drain two or more integrated types of land use. The sites were not selected to be statistically representative of the Nation's streams. Thus, characterizations of water-quality conditions are relative to the distribution of water-quality conditions at the sites included in this analysis.

Sites typically were located near USGS stream-gauging stations, and a sampling reach was identified at each site. Sampling reach lengths were determined based on criteria including the number of riffle–pool sequences, meander wavelength, and a minimum and a maximum length. Such criteria have been suggested as important considerations to optimal sampling efforts for characterizing stream fish species richness and species relative abundances (Lyons 1992a, Meador and others 1993a).

Attempts were made to include at least two riffle-pool sequences within each sampling reach. When this was not possible, reach length was determined based on a distance of 20 times mean channel width, roughly equivalent to one meander wavelength (Fitzpatrick and others 1998). For wadeable sites, a minimum reach length of 150 m and a maximum reach length of 300 m were established prior to sampling. For nonwadeable sites, minimum and maximum reach lengths were 300 and 1000 m, respectively.

Land-Use Identification

Land use in the basin upstream from a reach was determined from a classification of land cover derived from spectral information collected in 1990 by advanced very high resolution radiometer (AVHRR) aboard National Oceanic and Atmospheric Administration earth-orbiting satellites (US Geological Survey 1993). The AVHRR data were used to identify land-cover classes that were grouped into categories, including agricultural, forested, urban, and rangeland uses following definitions of Anderson and others (1976). Amounts of each land-use category in each basin were determined by overlaying basin boundaries on a 1-km grid of land use. Drainage area (square kilometers) was determined from basin boundaries delineated from a 1-km resolution digital-elevation model of the conterminous United States. Site elevation (meters) also was determined from this model.

Water Physicochemical Data

Surface-water samples were collected according to NAWQA protocols (Shelton 1994). Water-column physicochemical data collection at each site included (in milligrams per liter) total nitrogen, total phosphorus, suspended sediment, and total dissolved solids. Sites were sampled approximately monthly for 1 or 2 years. To facilitate comparisons of total nitrogen and total phosphorus among sites that might be biased by varying streamflow and sampling frequencies, flow-weighted concentrations were determined (Clark and others 2000). The rating curve method was used to estimate a concentration value for each day of a common period of streamflow record, and based on these daily estimates, a mean annual flow-weighted concentration was computed for each site.

Riparian Data

Riparian data were collected at each sampling reach by using standard NAWQA sampling protocols (Meador and others 1993b, Fitzpatrick and others 1998). Data used in this study were restricted to variables describing riparian condition, which included data on stream-channel modification, bank erosion, bank vegetative stability, and woody riparian vegetation density.

Stream-channel modification was categorized for each sampling reach as modified, moderately modified, or unmodified. Modified stream channels included reaches that had been channelized and/or had artificial banks or beds. Moderately modified streams included channels that had modifications upstream or downstream from the sampling reach, such as dam construction ranging from large reservoirs to small low-head dams.

Frequency of occurrence of bank erosion was determined from 12 observations of erosion along each reach (6 observations along each bank). Combined observations were then grouped, based on the 1st–25th percentile, 26th–75th percentile, or 76th–100th percentile of the data for each study unit. Vegetative bank stability was assessed by using a rating based on four classes (1–4) representing the percentage vegetative cover on the bank surface (Fitzpatrick and others 1998). A mean vegetative bank stability rating was calculated from the 12 sampling points along the reach. Relative density of woody riparian vegetation was calculated for the sampling reach from point-quarter vegetation sampling (Meador and others 1993b, Fitzpatrick and others 1998). Three groups of mean relative density of riparian vegetation were determined based on the 1st–25th percentile, 26th–75th percentile, or 76th–100th percentile of the data for each study unit.

Fish Community Data

Fish were collected in the 20 study units during summer low-flow periods during 1993–1995, using a NAWQA standard sampling protocol consisting of two-pass electrofishing followed by seining (Meador and others 1993a). Fish were identified to species, counted, and examined for external anomalies. Fish that could not be identified in the field were retained for identification and processing in a laboratory (Walsh and Meador 1998).

In each study unit, fish species were classified as native or introduced. Classification of the majority of fish species was established on the basis of two national databases—the Texas Natural History Collections

Table 2. Riparian condition metric scoring criteria

Metric	Degradation scoring criteria		
	5 (low)	3 (moderate)	1 (high)
Stream channel modification	Unaffected by modification	Partly affected by modification	Largely affected by modification
Bank erosion (within study unit percentile of occurrence)	1–25	26–75	76–100
Vegetative bank stability	4.0–3.5	3.4–2.5	<2.5
Relative density of woody riparian vegetation (within study unit percentile)	76–100	26–75	1–25

North American freshwater fishes index (Texas Memorial Museum 1998), and the Nonindigenous Aquatic Species database (US Geological Survey 2000). In addition, Lee and others (1980), and various state and regional fish books were used (Laerm and Freeman 1986, Robison and Buchanan 1988, Etnier and Starnes 1993, Jenkins and Burkhead 1993, Rhode and others 1994, Cross and Collins 1995, Mettee and others 1996). In a few cases, fish species status was determined by consulting with regional experts.

A tolerance category (intolerant, intermediate, tolerant) was assigned to each species on the basis of sensitivity of the species to anthropogenic changes in the environment, including pollution, water temperature, or habitat alteration. Trophic ecology of adults was classified as omnivore, detritivore, piscivore, invertivore, or other. For many species, tolerance and trophic categories were compiled from previous classifications, usually conducted as part of the development of local or regional bioassessment procedures (Karr and others 1986, Leonard and Orth 1986, Angermeier and Schlosser 1987, Ohio Environmental Protection Agency 1987, Plafkin and others 1989, Bramblett and Fausch 1991, Simon 1991, Lyons 1992b, Gatz and Harig 1993, Hall and others 1994, Shields and others 1995, Lyons and others 1996, Halliwell and others 1999, Mundahl and Simon 1999, Zaroban and others 1999). Species not previously classified were assigned tolerance and trophic categories on the basis of accounts in state or regional fish references (Laerm and Freeman 1986, Robison and Buchanan 1988, Etnier and Starnes 1993, Jenkins and Burkhead 1993, Rhode and others 1994, Cross and Collins 1995, Mettee and others 1996), taxon-specific references (Kuehne and Barbour 1983), or the primary literature (Yerger and Relyea 1968, Heins and Clemmer 1975, Hurst and others 1975, Olmsted and Cloutman 1979, Brown 2000). In cases where different authors assigned the same species to different classifications, the classification assigned most often was used. A few newly described endemic species could not be assigned tolerance or trophic classifications.

Riparian and Fish Community Condition Scores

The riparian condition (RIPCON) score approach developed for this investigation was calculated based on rankings of the four diagnostic metrics—stream modification, bank erosion, bank vegetative stability, and riparian vegetation. Each of the four metrics was scored as 5 (low), 3 (moderate), or 1 (high) to represent levels of degradation (Table 2). The scores were summed to provide a site score ranging from 4, representing a site where riparian conditions were relatively degraded, to 20, representing a site where riparian conditions were relatively undegraded. Scores were averaged for each site studied during the 1993–1995 sampling period.

A measure of fish community condition (FISHCON) was developed using four attributes of fish communities that were comparable within study units and was based on a subset of metrics commonly used in the IBI. The conceptual framework of the IBI is based on underlying hypotheses of how fish communities respond to increasing environmental degradation (Karr 1981, Fausch and others 1990, Yoder and Rankin 1995). Among these hypotheses, the following attributes are expected to increase with increasing environmental degradation: (1) the proportion of individuals that are members of tolerant species; (2) the proportion of omnivores; (3) the proportion of individuals that are members of introduced species; and (4) the incidence of externally evident disease, parasites, and morphological anomalies. Trophic generalists were substituted for omnivores. Trophic generalists also include detritivores (Goldstein and Simon 1998). Whereas a local or regional IBI contains more detail on the composition and structure of the fish community by necessity, the fish condition index included only metrics commonly used that respond to a broad signal of degradation across the entire gradient of conditions. At low levels of degradation, the number of introduced species is sensitive to degradation; at moderate levels of degradation, the proportions of tolerant species and trophic generalists increase; and at high levels of degradation, the propor-

Table 3. Example scoring criteria for calculating fish condition scores in western Great Plains streams[a]

	Degradation scoring criteria		
Metric	5 (low)	3 (moderate)	1 (high)
Tolerant individuals (%)	0–25	26–50	51–100
Trophic generalist individuals (%)	0–35	36–60	61–100
Introduced individuals (%)	0–2	3–8	>8
Individuals with external anomalies (%)	0–2	3–5	>5

[a]This example was derived from criteria developed by Bramblett and Fausch (1991).

Table 4. References used in developing fish condition scores for 20 NAWQA study units

Study unit	Reference
ACFB	Devio and others (1997), Schleiger (2000)
ALBE	North Carolina Department of Environment, Health, and Natural Resources (1997)
CCPT	Zaroban and others (1999)
CNBR	Bramblett and Fausch (1991), Frenzel and Swanson (1996)
CONN	Langdon (1989), Jacobson (1994), Halliwell and others (1999),
GAFL	Devio and others (1997), Schleiger (2000)
HDSN	Langdon (1989), Jacobson (1994), Halliwell and others (1999),
LSUS	Versar, Inc. (1992), Scott and Hall (1997), McCormick and others (2001)
NVBR	Bramblett and Fausch (1991)
OZRK	Hlass and others (1998)
POTO	Versar, Inc. (1992), Scott and Hall (1997), McCormick and others (2001)
REDN	Goldstein and others (1994), Niemela and others (1999)
RIOG	Bramblett and Fausch (1991)
SANJ	Moyle and Marchetti (1999)
SPLT	Schrader (1986), Bramblett and Fausch (1991)
TRIN	Bramblett and Fausch (1991)
USNK	Chandler and others (1993), Zaroban and others (1999)
WHIT	Simon (1992), Frey and others (1996)
WILL	Hughes and Gammon (1987)
WMIC	Lyons (1992b), Lyons and others (1996)

tion of individuals with external anomalies increases (Karr and others 1986). Thus, the multimetric approach used for this analysis consisted of a composite of the percentage of tolerant, trophic generalist, and introduced individuals and the percentage of individuals with external anomalies.

Each metric received a score following a concept similar to that proposed by Karr and others (1986) for the IBI. Thus, a metric received a score of 5 points if it had a value within the range expected for a fish community with little human influence; 1 point if the metric had a value within the range expected for a fish community that departs significantly from a reference condition; and a score of 3 points if the metric had an intermediate value. Metric scores were derived using scoring criteria for reference conditions, degraded conditions, and intermediate conditions, from locally or regionally developed IBIs applicable to that study unit. For example, Bramblett and Fausch (1991) suggested that individuals of tolerant species ranging from 0 to 25% in a sample represented a fish community in a western Great Plains stream experiencing relatively little human influence (Table 3). Therefore, the metric for the individuals of tolerant species in fish community samples collected, for example, from streams in the Rio Grande Study Unit, was given a score of 5 points if the individuals ranged from 0 to 25%. In some cases, expectations were derived from multiple published references to score metrics for a particular study unit (Table 4).

Scores for each of the four metrics were summed to provide a site score ranging from 4, indicating a fish community in a degraded condition, to 20, indicating a fish community not in a degraded condition. Scores for sites sampled more than once during the 1993–1995 sampling period were averaged based on the results of Niemela and others (1999), who found that temporal variability in IBI scores sampled in successive years and repeatedly during a single year was not significant.

Data Analysis

Of the 226 sites sampled, complete physical, chemical, and biological data were available for 172 sites, with the number of sites per study unit ranging from 6 to 13 (Table 1). The data set consisted of 12 variables, including FISHCON, RIPCON, drainage area, elevation, percentage of land use (agricultural, forested, urban, and range) within the basin, total nitrogen, total phosphorus, total dissolved solids, and suspended sediment. Variables were examined for normality using normal probability plots and transformed to improve normality when necessary. Percentage of land use was arcsine square-root transformed. Drainage area, elevation, total nitrogen, total phosphorus, total dissolved solids, and suspended sediment were transformed using $\log_{10}(x + 1)$. Statistical analyses were considered significant at $\alpha < 0.05$.

Stepwise least-squares multiple regression with FISHCON or RIPCON scores as dependent variables was used to assess relations among basin land use, water physicochemistry, and measures of riparian and fish community condition. Final models were selected when none of the variables outside the model had significant *F* statistics and every variable in the model was significant.

Principal components analysis (PCA) of correlation matrices of basin land use and water physicochemical variables was conducted to assess patterns in environmental variations among sites. Variables were standardized to mean = 0 and standard deviation = 1 before analysis. Site scores on PCA axes were then used to summarize environmental conditions and to relate site characteristics to riparian and fish community condition using Pearson correlation. The number of PCA axes examined was determined by Kaiser's rule, which states that the minimum eigenvalue should be 1 when correlation matrices are used (Legendre and Legendre 1983).

Cluster analysis was used to determine whether physicochemical conditions indicative of specific basin land uses could be discerned at the large scale of the analysis. If streams in the land-use categories had such physicochemical conditions, then FISHCON and RIPCON scores could be compared among the different land uses and physicochemical conditions to determine which land uses were associated with high or low scores of FISHCON and RIPCON. Variables were standardized to mean = 0 and standard deviation = 1 before clustering. Clustering was conducted using Euclidean distances and Ward's method. To verify that the clusters actually represented groups of sites with distinct environmental characteristics and to test for differences in mean RIPCON and FISHCON scores among clusters, analysis of variance (ANOVA) using Tukey's studentized range test was conducted.

Relations between riparian and fish community condition across basin land uses were also examined by comparing RIPCON scores to FISHCON scores within specific land uses. Scores were classified by quartile, and the frequency of scores within each quartile was examined to determine whether riparian condition consistently paralleled fish community condition across sites where a single land use represented greater than 50% of the drainage area. Some sites were not used in this analysis because a single land use was not distinctly dominant. The numbers of sites by land-use category were agriculture, 70; forest, 50; urban, 14; and range, 9. Because of the relatively low number of sites with range as the dominant land use, this category was deleted from the analysis.

Results

Across all 172 sites, the mean (± SD) RIPCON score was 12.8 ± 3.63, whereas the mean FISHCON score was 13.8 ± 4.54). Streams ranged in drainage area from 18 to 221,496 sq km (median = 808.8). Stepwise regression of all sites indicated that decreased riparian condition was associated with increased total nitrogen, suspended sediment, and urban land use, and decreased elevation ($R^2 = 0.30$, $P = 0.0001$). Decreased fish community condition was associated with increased total dissolved solids and percentage of range land use, and decreased riparian condition and percentage of agricultural land use ($R^2 = 0.27$, $P = 0.0001$).

The PCA of all sites produced three significant axes that collectively represented 63.8% of the variation in environmental conditions among the sites. Factors loading $>|0.40|$ on axis 1 included total nitrogen and percentages of agricultural and forested land uses; on axis 2, percentage of rangeland use, elevation, and drainage area; and on axis 3, percentage of urban land use (Table 5). The PCA revealed that the majority of the sites represented an agricultural–forested land-use gradient as indicated by axis 1 (Figure 1), with relatively fewer sites representing rangeland use within relatively larger drainage areas and higher elevations, as indicated by axis 2 (Figure 1). Still fewer sites were characterized by urban land use, as indicated by axis 3 (Figure 2).

Scores for FISHCON were significantly related to scores on PCA axis 1 (FOREST) ($r = 0.18$, $P = 0.016$), scores on PCA axis 2 (RANGE, AREA, ELEV) ($r = -0.33$, $P = 0.001$), and scores on PCA axis 3 (URBAN) ($r = -0.19$, $P = 0.014$). RIPCON scores were signifi-

Table 5. Principal components analysis variable loadings for axes 1, 2, and 3[a]

	Axis 1	Axis 2	Axis 3
Drainage area (AREA)	−0.043	**0.440**	−0.005
Elevation (ELEV)	0.193	**0.479**	−0.064
Agricultural (AG) land use	**−0.464**	−0.195	−0.305
Forest (FOREST) land use	**0.454**	−0.106	−0.295
Urban (URBAN) land use	−0.021	−0.103	**0.849**
Range (RANGE) land use	0.090	**0.614**	−0.013
Total nitrogen (TN)	**−0.458**	−0.06	−0.077
Total phosphorous (TP)	−0.389	0.184	−0.019
Suspended sediment (SED)	−0.252	0.131	−0.207
Total dissolved solids (TDS)	−0.327	0.293	0.216

[a]Bold values are considered high (>|0.40|). Variable abbreviations are in parentheses; $N = 172$ sites.

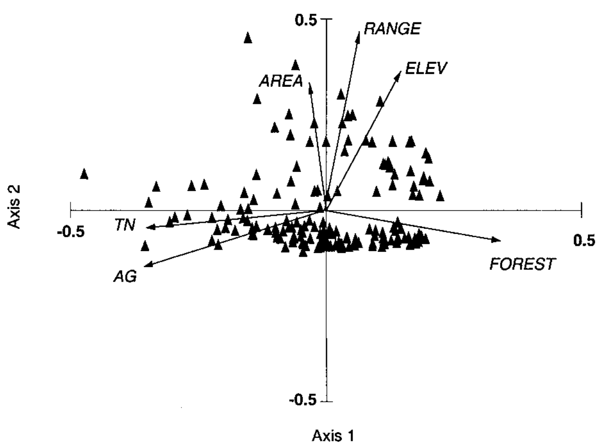

Figure 1. Principal components analysis plot of axis 1 and axis 2. Arrows indicate variable loadings; triangles indicate site loadings. See Table 5 for variable abbreviations.

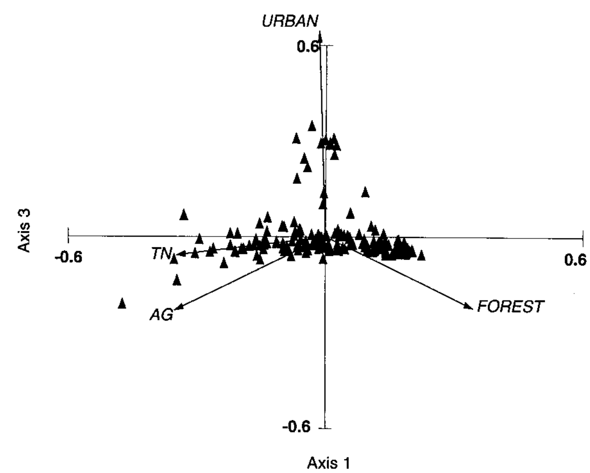

Figure 2. Principal components analysis plot of axis 1 and axis 3. Arrows indicate variable loadings; triangles indicate site loadings. See Table 5 for variable abbreviations.

cantly related to scores on PCA axis 1 ($r = 0.45$, $P = 0.001$), and scores on PCA axis 3 ($r = -0.19$, $P = 0.015$) but were not significantly related to scores on PCA axis 2 ($P = 0.121$).

Cluster analysis identified 6 clusters of 16 or more sites. Two additional clusters of two sites each were identified. ANOVA of individual variables among clusters revealed land-use variables that were significantly greatest within each of four clusters. Land uses in the remaining two clusters were not significantly different but appeared to represent a mix of two dominant land uses. Based on these distinctions, clusters were then classified—cluster 1 was classified as Forest; cluster 2 was classified as Mixed Ag-Forest, cluster 3 was classified as Ag; cluster 4 was classified as Mixed Forest-Range; cluster 5 was classified as Urban; and cluster 6 was classified as Range (Table 6).

Mean RIPCON scores for the Forest and Mixed Forest-Range clusters were significantly higher than those for the Ag and Urban clusters (Figure 3). The mean RIPCON score for the Range cluster, however, was not significantly different from the mean scores for the Mixed Ag-Forest, Forest, and Mixed Forest-Range clusters (Figure 3). The mean Forest FISHCON score was significantly higher than the mean scores for the Ag, Urban, Mixed Forest-Range, and Range clusters, but was not significantly different from the mean scores for the Mixed Ag-Forest cluster (Figure 3). The mean FISHCON score for the Ag cluster also was not significantly different from that of the Mixed Ag-Forest cluster.

Generally, the frequency distribution of riparian condition scores paralleled the frequency distribution of fish community condition scores for sites where forested or agricultural land uses were dominant (Figure 4). Both riparian and fish community condition were high most frequently within areas characterized by for-

Table 6. Mean values of environmental variables in each of six land-use clusters[a]

	Cluster 1 Forest (N = 32)	Cluster 2 Mixed Ag-Forest (N = 49)	Cluster 3 Ag (N = 32)	Cluster 4 Mixed Forest-Range (N = 22)	Cluster 5 Urban (N = 16)	Cluster 6 Range (N = 17)
Drainage area (km^2)	1,969	4,133	2,078	6,894	77.8	**27,039**
Elevation (m)	191.0	123.0	6.4	1,678.5	191.7	1,114.4
Agricultural land use (%)	12.5	56.9	**88.1**	7.8	9.5	18.2
Forested land use (%)	**80.2**	30.7	4.9	56.1	13.4	26.4
Urban land use (%)	1.0	4.1	2.2	1.6	**76.2**	3.7
Range land use (%)	1.6	3.9	2.7	25.8	0.1	**47.5**
Total nitrogen (mg/liter)	0.59	2.14	**4.89**	0.37	1.87	2.14
Total phosphorus (mg/liter)	0.04	0.16	0.39	0.06	0.19	0.38
Suspended sediment (mg/liter)	25.9	67.3	320.9	64.5	99.1	230.8
Total dissolved solids (mg/liter)	96.4	184.9	437.0	164.5	296.7	516.7

[a]Bold values are significantly greatest ($P < 0.05$) across all clusters (ANOVA followed by Tukey tests); N = number of sites in each cluster.

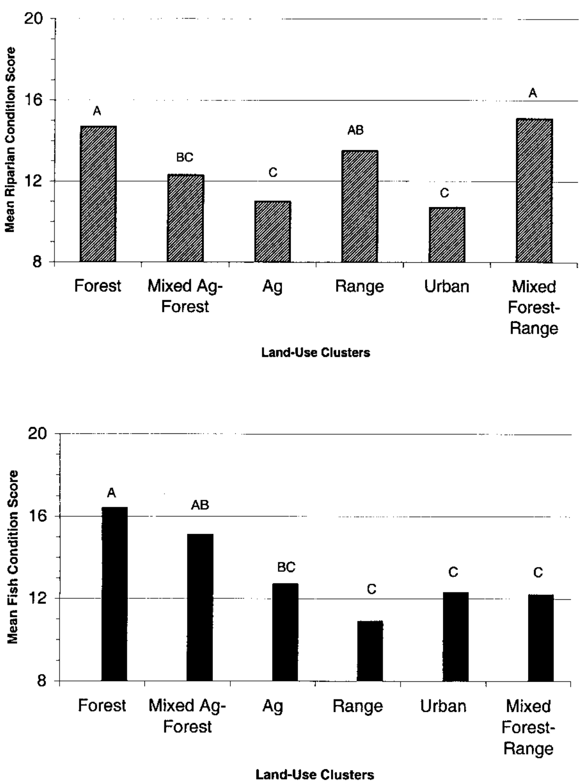

Figure 3. Mean riparian and fish condition scores for each of six clusters of sites, classified by land use. Means with the same letter are not significantly different (ANOVA followed by Tukey tests).

ested land use. Although the actual values of the riparian condition index were low at agricultural sites compared with values at other land-use sites except urban, the fish community condition indices were not (Figure 3), and relatively few agricultural land-use sites had both highly degraded fish communities and riparian zones in poor condition (Figure 4). In urban settings, there was little similarity between riparian condition and fish community condition scores.

In areas where agricultural land use was relatively high (>50% of the basin), 64% of the sites (45 of 70 sites) had FISHCON scores above average (>13.8), indicating relatively low degradation of the fish community despite high agricultural land use. Sites where agricultural land use was >50% were divided into two groups—sites where FISHCON scores were above average and sites where FISHCON scores were below average. Total nitrogen, total phosphorus, suspended sediment, and total dissolved solids were significantly greater at sites where FISHCON scores were below average (Table 7). No other significant differences were detected.

Discussion

The connection between the condition of the riparian zone and the fish community was present throughout the broad range of geographic locations, drainage areas, elevations, and land uses. The level of riparian degradation appears to have increased with the magnitude of change that has occurred since the original land use has evolved to the current land use. Hence, urban and agricultural land uses produce the greatest effects. Riparian condition was higher in forested and rangeland uses and lower in urban and agricultural land uses. Wang and others (1997) determined that for Wisconsin streams, agricultural land use was correlated with bank instability. Increased agricultural land use has been related to decreased water-quality conditions in streams (Omernik and others 1981, Smart and others 1981, Osborne and Wiley 1988). Thus, although the connection between riparian zone and fish community conditions appeared to be ubiquitous, the nature of the

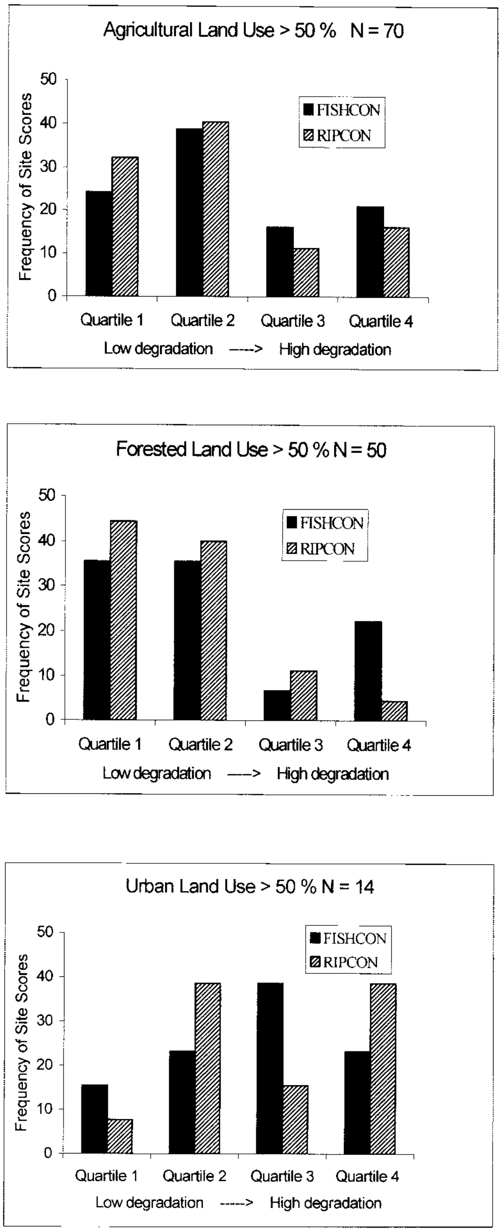

Figure 4. Fish community and riparian condition scores plotted in quartiles by land use (N = number of sites).

relation between riparian condition and fish community structure was dependent upon the land use.

At a broad geographic scale, the relation between fish community structure and agricultural land use is complex. Roth and others (1996) reported that agricultural land use was a primary determinant of fish community structure in streams with the amount of agriculture negatively correlated to IBI scores; thus, a degraded fish community was associated with increasing agricultural land use. However, Wang and others (1997) did not detect a linear relation between IBI scores and agriculture except where agricultural land use was the major land use within the basin. These authors suggested that fish communities may not respond to relatively low levels of agriculture or disturbance within a basin and noted that even when agriculture exceeded 80%, some sites had fish communities in good condition. A number of sites were located where agriculture was at relatively high levels (>50% of the basin), yet fish condition scores were comparatively high, suggesting fish communities in good condition. Thus, it appears that agricultural land use, as determined by the percentage of agricultural land use within a basin, may not be associated necessarily with degraded fish community structure at broad geographic scales.

The observed relation between decreases in degraded fish community structure and increased agricultural land use may be the result of several factors, including the relative disturbances of other land uses within a basin, the location of the particular land use within a basin and its proximity to the stream, and management practices that control water physicochemistry. Wang and others (2000) observed a positive correlation between basinwide agricultural land use and fish biotic integrity—degraded fish communities decreased with increasing agriculture. These authors noted that in their study area in Wisconsin, high levels of agriculture were associated with low levels of urbanization; conversely, low levels of agriculture were associated with high levels of urbanization. Wang and others (2000) concluded that urban land use was more deleterious to stream fishes than agricultural land use on a per-unit-area basis. Although the results of the present investigation were similar to the results reported by Wang and others (2000)—degraded fish communities decreased with increased agriculture—low levels of agricultural land use were not associated with high levels of urbanization.

Any index approach used over a large geographic area at a given point in time has limitations. There may be more comprehensive methods for assessing riparian degradation that are likely to be more sensitive to a wide array of environmental changes in streams at a local scale. The focus of the RIPCON is on degradation of the riparian zone. Thus, the index is designed to reflect large-scale stream modifications that are most likely to be observed over time and is not designed to reflect aspects of instream habitat solely, which are more likely to reflect short-term water levels.

Characteristics of stream channel, bank geomorphology, and riparian vegetation can be used as diagnostic criteria to assess environmental changes in

Table 7. ANOVA where agricultural land use was >50% and FISHCON scores were above (>13.8) and below (≤13.8) average

	FISHCON > 13.8 (N = 45)	FISHCON ≤ 13.8 (N = 25)	P
Drainage area (km^2)	1785	4643	0.410
Elevation (m)	179.9	210.6	0.888
Forested land use (%)	18.2	13.3	0.079
Urban land use (%)	2.4	3.3	0.212
Range land use (%)	2.5	2.3	0.433
Total nitrogen (mg/L)	3.09	4.59	0.029
Total phosphorus (mg/L)	0.23	0.36	0.024
Suspended sediment (mg/L)	201.7	589.8	0.031
Total dissolved solids (mg/L)	241.3	407.6	0.004
Riparian condition	12.3	13.1	0.293

stream systems (Simon and Downs 1995). Environmental changes of sufficient magnitude and extent can initiate responses in the condition of riparian vegetation (Simon and Hupp 1992). Although limitations exist with any index approach in a broad-scale survey with a large number of sampling sites across a variety of geographic settings and stream sizes, such an approach can provide useful information to assess gross environmental changes in stream and bank condition (Kirchhofer 1995).

As with the RIPCON approach, the FISHCON represents an assessment of the fish community at one point in time; thus, interpretations undoubtedly will be coarse. Use of the FISHCON across a broad geographic area also may have limitations. Most applications of such a multimetric approach to assessing fish community structure have involved only small- to medium-sized wadeable warmwater streams (Simon and Lyons 1995). In some studies, attempts have been made to adjust fish community metrics to account for differences in fish communities along the river continuum from headwaters to mouth (Simon and Lyons 1995). Adjustments for cold, headwater streams generally have focused on a reduction in the number of metrics, reflecting the relatively simplified structure and function of fish communities typical of such headwater streams. Similarly, a review of modifications proposed for nonwadeable rivers suggested that no adjustments for large rivers could be made that would improve sensitivity. Although limitations exist with any index approach, the multimetric approach described herein to assess fish community condition provides useful information to assess gross environmental relations across broad geographic areas.

The use of land-use percentages within a basin as a measure of broad-scale disturbance may have limitations. Hughes and others (1998) in an intensive study of the Willamette Valley, Oregon, USA, noted that the percentage of introduced fish increased with agricultural and urban land use. Rathert and others (1999), however, noted that broad categories of land use may not be adequate to observe patterns in local-scale relations between land use and fish species richness. Wang and others (2001), in a study of the effects of urbanization on stream fish in Wisconsin, noted that species richness was negatively correlated with the percentage of urban land use in the basin; however, they also noted that the presence of a 50-m-wide riparian buffer was positively related to species richness. Wang and others (2001) concluded that the location of urbanization within a basin, the distance of urban activities from a stream, and the presence of a riparian buffer may be important determinants of fish community structure. Similarly, the location of agricultural activities within a basin, the distance of agricultural activities from a stream, and the presence of a riparian buffer may affect observed relations between fish community structure and the percentage of agricultural land use within a basin.

Agricultural land use within a basin is a general descriptor that may not adequately characterize local activities affecting water physicochemistry. It appears that degraded fish communities are related to increased nutrients, suspended sediment, and total solids and that these relations may be more important than relations with the percentages of agricultural land use within a basin. In California, the San Joaquin River system has been intensively converted to agricultural land use, with nearly all available flow substantially altered by dams, diversions, and irrigation return flows (Brown 2000). As a result, agricultural land use in the San Joaquin River system is associated with altered flows, commonly containing high concentrations of nutrients and pesticides (Brown and others 1999, Brown 2000). Additionally, agricultural basins with poorly drained soils (high runoff potential) tend to have artificial drainage systems—usually either ditches or buried tile drains. Tile drains with open inlets deliver runoff laden with sediment and nutrients directly to the stream and bypass the riparian zone.

Although results of this study seem to suggest that basinwide rangeland use is related to degraded fish communities, this may not be the case. Reduced vegetative cover on cattle-grazed rangeland increases sediment in runoff and nutrients from cattle-waste products. The delivery of these sediments and nutrients to streams is compounded by the loss of riparian vegetation from grazing and the physical destabilization of the streambanks by the cattle (Platts 1991, Waters 1995). Although total dissolved solids and suspended sediment were relatively high for rangeland use sites, riparian condition did not appear to be a factor. Relatively few sites were examined that were characterized by rangeland use (17 of 172 sites), and these sites were characterized by mean drainage areas in excess of 27,000 sq km. Thus, the relatively few sites and very large streams represented by these sites may have influenced the observed relations between fish community condition and rangeland use.

Results of this study suggest that across large geographic scales, measures of water physicochemistry and riparian condition may be better indicators of fish community condition than basinwide determinations of land use, similar to the conclusions of Lammert and Allan (1999). The results of this study support riparian enhancement, particularly in agricultural and urban basins, to restore fish communities. Whereas numerous studies have indicated that riparian restorations are successful in specific cases, this analysis suggests a universal importance of riparian zones to the maintenance and restoration of diverse fish communities in streams.

Acknowledgments

We thank Paul Angermeier, Terry Maret, Gary Larson, and anonymous reviewers for their helpful comments and suggestions. We also thank the individuals, too numerous to name individually, who spent time and effort collecting data as part of the NAWQA Program.

Literature Cited

Anderson, J. R., E. E. Hardy, J. T., Roach, and R. E. Witmer. 1976. A land use and land cover classification system for use with remote sensor data. US Geological Survey Professional Paper 964, 28 pp.

Allan, J. D., D. L. Erickson, and J. Fay. 1997. The influence of catchment land use on stream integrity across multiple spatial scales. *Freshwater Biology* 37:149–161.

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.

Barling, R. D., and I. D. Moore. 1994. Role of buffer strips in management of waterway pollution: a review. *Environmental Management* 18:543–558.

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.

Brown, L. R. 2000. Assemblages of fishes and their associations with environmental variables, lower San Joaquin River drainage, California. *Environmental Biology of Fishes* 57:251–269.

Brown, L. R., C. R. Kratzer, and N. M. Dubrovsky. 1999. Integrating chemical, water quality, habitat, and fish assemblage data from the San Joaquin River drainage, California. Pages 25–62 *in* K. M. Scow, G. E. Fogg, D. E. Hinton, and M. L. Johnson, (eds.), Integrated assessment of ecosystem health. Lewis Publishers, Boca Raton, Florida.

Chandler, G. L., T. R. Maret, and D. W. Zaroban. 1993. Protocols for assessment of biotic integrity (fish) in Idaho streams. Idaho Department of Health and Welfare, Division of Environmental Quality, Water Quality Monitoring Protocols, Boise, 40 pp.

Clark, G. M., D. K. Mueller, and M. A. Mast. 2000. Nutrient concentrations and yields in undeveloped stream basins of the United States. *Journal of the American Water Resources Association* 36:849–860.

Cross, F. B., and J. T. Collins. 1995. Fishes in Kansas. University of Kansas Museum of Natural History Public Education Series, no. 3, 189 pp.

Devio, J. C., C. A. Couch, and B. J. Freeman. 1997. Use of preliminary index of biotic integrity in urban streams around Atlanta, Georgia. Pages 119–122 *in* K. J. Hatcher, (ed.), Proceedings, 1997 Georgia Water Resource Conference. Athens, Georgia.

Etnier, D. A., and W. C. Starnes. 1993. The fishes of Tennessee. The University of Tennessee Press, Knoxville, Tennessee, 681 pp.

Fausch, K. D., J. Lyons, J. R. Karr, and P. L. Angermeier. 1990. Fish communities as indicators of environmental degradation. *American Fisheries Society Symposium* 8:123–144.

Fitzpatrick, F. A., I. R. Waite, P. D'Arconte, M. R. Meador, M. A. Maupin, and M. E. Gurtz. 1998. Revised methods for characterizing stream habitat in the National Water-Quality Assessment Program. US Geological Survey Water-Resources Investigations Report 98-4052, 64 pp.

Frenzel, S. A., and R. B. Swanson. 1996. Relations of fish community composition to environmental variables in streams of central Nebraska, USA. *Environmental Management* 20:689–705.

Frey, J. W., N. T. Baker, M. J. Lydy, and W. W. Stone. 1996. Assessment of water quality at selected sites in the White River Basin, Indiana, 1993 and 1995 using biological indices. US Geological Survey Fact Sheet 209–96, 4 pp.

Gatz, A. J., Jr., and A. L. Harig. 1993. Decline in the index of biotic integrity of Delaware Run, Ohio, over 50 years. *Ohio Journal of Science* 93:95–100.

Gilliom, R. J., W. M. Alley, and M. E. Gurtz. 1995. Design of the National Water-Quality Assessment Program: Occurrence and distribution of water-quality conditions. US Geological Survey Circular 1112, 33 pp.

Gilliom, R. J., D. K. Mueller, and L. H. Nowell. 1998. Methods for comparing water-quality conditions among National Water-Quality Assessment study units, 1992–1995. US Geological Survey Open-File Report 97–589, 54 pp.

Goldstein, R. M., and T. P. Simon. 1998. Toward a united definition of guild structure for feeding ecology of North American freshwater fishes. Pages 123–202 *in* T. P. Simon, (ed.), Assessing the sustainability and biological integrity of water resources using fish communities. CRC Press, Lewis Publishers, Boca Raton, Florida.

Goldstein, R. M., T. P. Simon, P. A. Bailey, M. Ell, E. Pearson, K. Schmidt, and J. W. Emblom. 1994. Concepts for an index of biotic integrity for the streams of the Red River of the North Basin. Pages 169–180 *in* Proceedings North Dakota Water Quality Symposium, 30–31 March 1994. Fargo, North Dakota.

Hall, L. W., S. A. Fischer, W. D. Killen, Jr., M. C. Scott, and M. C. Ziegenfuss. 1994. Status assessment in acid-sensitive and non-acid-sensitive Maryland coastal plain streams using an integrated biological, chemical, physical, and land-use approach. *Journal of Aquatic Ecosystem Health* 3:145–167.

Halliwell, D. R., 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, (ed.), Assessing the sustainability and biological integrity of water resources using fish communities. CRC Press, Lewis Publishers, Boca Raton, Florida.

Harding, J. S., E. F. Benfield, P. V. Bolstad, G. S. Helfman, and E. B. D. Jones, III. 1998. Stream biodiversity: the ghost of land use past. *Proceedings of the National Academy of Sciences, USA* 95:14843–14847.

Heins, D. C., and G. H. Clemmer. 1975. Ecology, foods, and feeding of the longnose shiner, *Notropis longirostris* (Hay), in Mississippi. *American Midland Naturalist* 94:284–295.

Hlass, L. J., W. L. Fisher, and D. J. Turton. 1998. Use of the index of biotic integrity to assess water quality in forested streams of the Quachita Mountains ecoregion, Arkansas. *Journal of Freshwater Ecology* 13:181–192.

Hughes, R. M., and J. R. Gammon. 1987. Longitudinal changes in fish assemblages and water quality in the Willamette River, Oregon. *Transactions of the American Fisheries Society* 116:196–209.

Hughes, R. M., P. R. Kaufmann, A. T. Herlihy, T. M. Kincaid, L. Reynolds, and D. P. Larsen. 1998. A process for developing and evaluating indices of fish assemblage integrity: a case study for wadeable streams in the Willamette valley ecoregion, Oregon, USA. *Canadian Journal of Fisheries and Aquatic Sciences* 55:1618–1631.

Hurst, H., G. Bass, and C. Hubbs. 1975. The biology of the Guadalupe, Suwannee, and redeye basses. Pages 47–55 *in* R. H. Stroud and H. Clepper (eds.), Black bass biology and management. Sport Fishing Institute, Washington, DC.

Jacobson, R. A. 1994. Application of the index of biotic integrity to small Connecticut streams. Master's thesis. University of Connecticut, Storrs, 24 pp.

Jenkins, R. E., and N. M. Burkhead. 1993. Freshwater fishes of Virginia. American Fisheries Society, Bethesda, Maryland, 1,079 pp.

Johnson, L. B., R. Richards, G. E., Host, and J. W. Arthur. 1997. Landscape influences on water chemistry in midwestern stream ecosystems. *Freshwater Biology* 37:193–208.

Karr, J. R. 1981. Assessment of biotic integrity using fish communities. *Fisheries* 6(6):21–27.

Karr, J. R., and I. J. Schlosser. 1978. Water resources and the land-water interface. *Science* 201:229–234.

Karr, J. R., K. D. Fausch, P. L. Angermeier, P. R., Yant, and I. J. Schlosser. 1986. Assessing biological integrity in running waters: A method and its rationale. Illinois Natural History Survey Special Publication 5, Champaign, Illinois, 28 pp.

Kirchhofer, A. 1995. Morphological variability in the ecotone—an important factor for the conservation of fish species richness in Swiss rivers. *Hydrobiologia* 303:103–110.

Kuehne, R. A., and R. W. Barbour. 1983. The American darters. The University Press of Kentucky, Lexington, Kentucky, 177 pp.

Laerm, J., and B. J. Freeman. 1986. Fishes of the Okefenokee Swamp. The University of Georgia Press, Athens, Georgia, 118 pp.

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.

Langdon, R. W. 1989. The development of fish population-based criteria in Vermont. Pages 12–25 *in* T. P. Simon, L. L. Holst, and L. J. Shepard, (eds.), Proceedings of the first national workshop on biocriteria, US Environmental Protection Agency, EPA 905-9-89-003. Chicago, Illinois.

Lee, D. S., C. R. Gilbert, C. H. Hocutt, R. E. Jenkins, D. E. McAllister, and J. R. Stauffer, Jr. 1980. Atlas of North American freshwater fishes. North Carolina State Museum of Natural History. Raleigh, North Carolina, 867 pp.

Legendre, L., and P. Legendre. 1983. Numerical ecology. Elsevier Scientific Publishing Company, New York, 419 pp.

Leonard, P. M., and D. J. Orth. 1986. Application and testing of an index of biotic integrity in small, coolwater streams. *Transactions of the American Fisheries Society* 115:401–414.

Lyons, J. 1992a. The length of stream to sample with a towed electrofishing unit when fish species richness is estimated. *North American Journal of Fisheries Management* 12:198–203.

Lyons, J. 1992b. Using the index of biotic integrity (IBI) to measure environmental quality in warmwater streams of Wisconsin. US Forest Service General Technical Report NC-149, North Central Forest Experiment Station, St. Paul, Minnesota, 51 pp.

Lyons, J., L. Wang, and T. D. Simonson. 1996. Development and validation of an index of biotic integrity for coldwater streams in Wisconsin. *North American Journal of Fisheries Management* 16:241–256.

Marsh-Matthews, E., and W. J. Matthews. 2000. Geographic, terrestrial and aquatic factors: which most influence the structure of stream fish assemblages in the midwestern United States? *Ecology of Freshwater Fish* 9:9–21.

McCormick, F. H., R. M. Hughes, P. R. Kaufmann, D. V. Peck, J. L. Stoddard, and A. T. Herlihy. 2001. Development of an index of biotic integrity for the Mid-Atlantic Highlands region. *Transactions of the American Fisheries Society* 130:857–877.

Meador, M. R., T. F. Cuffney, and M. E. Gurtz. 1993a. Methods for sampling fish communities as part of the National Water-Quality Assessment Program. US Geological Survey Open-File Report 93–104, 40 pp.

Meador, M. R., C. R. Hupp, T. F. Cuffney, and M. E. Gurtz. 1993b. Methods for characterizing stream habitat as part of the National Water-Quality Assessment Program. US Geological Survey Open-File Report 93–408, 48 pp.

Mettee, M. F., P. E. O'Neill, and J. M. Pierson. 1996. Fishes of Alabama and the Mobile basin. Oxmoor House, Birmingham, Alabama, 820 pp.

Moyle, P. B., and M. P. Marchetti. 1999. Applications of indices of biotic integrity to California streams and watersheds. Pages 367–382 *in* T. P. Simon, (ed.), Assessing the sustainability and biological integrity of water resources using fish communities. CRC Press, Lewis Publishers, Boca Raton, Florida.

Mundahl, N. D., and T. P. Simon. 1999. Development and application of an index of biotic integrity for coldwater streams of the upper Midwest United States. Pages 1383–415 *in* T. P. Simon, (ed.), Assessing the sustainability and biological integrity of water resources using fish communities. CRC Press, Lewis Publishers, Boca Raton, Florida.

Niemela, S., E. Pearson, T. P. Simon, R. M. Goldstein, and P. A. Bailey. 1999. Development of an Index of Biotic Integrity for the species-depauperate Lake Agassiz Plain ecoregions, North Dakota and Minnesota. Pages 339–366 *in* T. P. Simon, (ed.), Assessing the sustainability and biological integrity of water resources using fish communities. CRC Press, Lewis Publishers, Boca Raton, Florida.

North Carolina Department of Environment, Health, and Natural Resources. 1997. Fish community structure. Pages 18–27 *in* Standard operating procedures—biological monitoring. Division of Water Quality, Water Quality Section Ecosystem Analysis Unit, Raleigh, North Carolina.

Ohio Environmental Protection Agency. 1987. Biological criteria for the protection of aquatic life: volumes I–III. Ohio Environmental Protection Agency, Columbus, Ohio.

Olmsted, L. L., and D. G. Cloutman. 1979. Life history of the flat bullhead, *Ictalurus platycephalus*, in Lake Norman, North Carolina. *Transactions of the American Fisheries Society* 108:38–42.

Omernik, J. M., A. R. Abernathy, and L. M. Hale. 1981. Stream nutrient levels and proximity of agricultural and forest land to streams: some relationships. *Journal of Soil and Water Conservation* 36:227–231.

Osborne, L. L., and D. A. Kovacic. 1993. Riparian buffer strips in water-quality restoration and stream management. *Freshwater Biology* 29:243–258.

Osborne, L. L., and M. J. Wiley. 1988. Empirical relationships between land use/land cover and stream water quality in an agricultural watershed. *Journal of Environmental Management* 26:9–27.

Plafkin, J. L., M. T. Barbour, K. D. Porter, S. K. Gross, and R. M. Hughes. 1989. Rapid bioassessment protocols for use in streams and rivers: benthic macroinvertebrates and fish. US Environmental Protection Agency, EPA/444/4-89-001, Washington, DC.

Platts, W. S. 1991. Livestock grazing. Pages 389–423 *in* W. R. Meehan, (ed.), Influences of forest and rangeland management on salmonid fishes and their habitats. American Fisheries Society, Special Publication 19, Bethesda, Maryland.

Rathert, D., D. White, J. C. Sifneos, and R. M. Hughes. 1999. Environmental correlates of species richness for native freshwater fish in Oregon, USA. *Journal of Biogeography* 26:257–273.

Richards, C., L. B. Johnson, and G. E. Host. 1996. Landscape-scale influences on stream habitats and biota. *Canadian Journal of Fisheries and Aquatic Sciences* 53(Suppl. 1):295–310.

Rhode, F. C., R. G. Arndt, D. G. Lindquist, and J. F. Parnel. 1994. Freshwater fishes of the Carolinas, Virginia, Maryland, and Delaware. The University of North Carolina Press, Chapel Hill, North Carolina, 222 pp.

Robison, H. W., and T. M. Buchanan. 1988. Fishes of Arkansas. University of Arkansas Press, Fayetteville, Arkansas, 536 pp.

Roth, N. E., J. D. Allan, and D. L. Erickson. 1996. Landscape influences on stream biotic integrity assessed at multiple spatial scales. *Landscape Ecology* 11:141–156.

Schleiger, S. L. 2000. Use of an index of biotic integrity to detect the effects of land use on stream fish communities in West-Central Georgia. *Transactions of the American Fisheries Society* 129:118–133.

Schrader, L. H. 1986. Testing of the Index of Biotic Integrity in the South Platte River Basin of northeastern Colorado. Master's thesis, Colorado State University, Fort Collins, 120 pp.

Scott, M. C., and L. W. Hall. 1997. Fish assemblages as indicators of environmental degradation in Maryland coastal plain streams. *Transactions of the American Fisheries Society* 126:349–360.

Shelton, L. R. 1994. Field guide for collecting and processing stream-water samples for the National Water-Quality Assessment Program. US Geological Survey Open-File Report 94–455, 42 pp.

Shields, F. D., Jr., S. S. Knight, and C. M. Cooper. 1995. Use of the index of biotic integrity to assess physical habitat degradation in warmwater streams. *Hydrobiologia* 312:191–208.

Simon, A., and P. W. Downs. 1995. An interdisciplinary approach to evaluation of potential instability in alluvial channels. *Geomorphology* 12:215–232.

Simon, A., and C. R. Hupp. 1992. Geomorphic and vegetative recovery processes along modified stream channels in West Tennessee. US Geological Survey Open-File Report 91–502, 142 pp.

Simon, T. P. 1991. Development of ecoregion expectations for the index of biotic integrity. I. Central corn belt plain. US Environmental Protection Agency, EPA 905/9-91/025, Chicago, Illinois, 242 pp.

Simon, T. P. 1992. Development of biological criteria for large rivers with an emphasis on an assessment of the White River drainage, Indiana. EPA 905/R-92/026. US Environmental Protection Agency, Chicago, Illinois, 70 pp.

Simon, T. P., and J. Lyons. 1995. Application of the Index of Biotic Integrity to evaluate water resource integrity in freshwater ecosystems. Pages 245–262 *in* W. S. Davis and T. P. Simon (eds.), Biological assessment and criteria: tools for water resource planning and decision making. CRC Press, Lewis Publishers, Boca Raton, Florida.

Smart, M. M., T. W. Barney, and J. R. Jones. 1981. Watershed impact on stream water quality: a technique for regional assessment. *Journal of Soil and Water Conservation* 36:297–300.

Stauffer, J. C., R. M. Goldstein, and R. M. Newman. 2000. Relationship of wooded riparian zones and runoff potential to fish community composition in agricultural streams. *Canadian Journal of Fisheries and Aquatic Sciences* 57:307–316.

Texas Memorial Museum. 1998. Texas Natural History Collection (TNHC) North American freshwater fishes index: images, maps and information. Accessed October 2001 at http://www.tmm.utexas.edu/tnhc/fish/na/naindex.html.

US Geological Survey. 1993. Conterminous U.S. land cover characteristics data set, version 2[CD-ROM]. U.S. Geological Survey, EROS Data Center, Sioux Falls, South Dakota.

US Geological Survey. 2000. Nonindigenous fish distribution information. Accessed October 2001 at http://nas.er.usgs.gov/fishes/.

Versar, Inc. 1992. An assessment of the feasibility of using an index of biotic integrity (IBI) approach for synthesizing information from a Maryland biological stream survey. Versar, Inc., Columbia, Maryland.

Walsh, S. J., and M. R. Meador. 1998. Guidelines for quality assurance and quality control of fish taxonomic data collected as part of the National Water-Quality Assessment Program. US Geological Survey Water-Resources Investigations Report 98–4239, 33 pp.

Wang, L., J. Lyons, P. Kanehl, and R. Gatti. 1997. Influences of watershed land use on habitat quality and biotic integrity in Wisconsin streams. *Fisheries* 22(6):6–12.

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.

Waters, T. F. 1995. Sediment in streams: sources, biological effects and control. American Fisheries Society, Monograph 7, Bethesda, Maryland, 251 pp.

Yerger, R. W., and K. Relyea. 1968. The flat-headed bullheads (Pisces: Ictaluridae) of the southeastern United States, and a new species of *Ictalurus* from the Gulf coast. *Copeia* 1968: 361–384.

Yoder, C. O., and E. T. Rankin. 1995. Biological response signatures and the area of degradation value: new tools for interpreting multimetric data. Pages 263–286 *in* W. S. Davis and T. P. Simon, (eds.), Biological assessment and criteria: tools for water resource planning and decision making. CRC Press, Lewis Publishers, Boca Raton, Florida.

Zaroban, D. W., M. P. Mulvey, T. R. Maret, R. M. Hughes, and G. D. Merritt. 1999. Proposed classification of species attributes for Pacific Northwest freshwater fishes. *Northwest Science* 73:81–93.