Relationships between Water Quality, Habitat Quality, and Macroinvertebrate Assemblages in Illinois Streams

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The influence of specific stressors, such as nutrient enrichment and physical habitat degradation, on biotic integrity requires further attention in Midwestern streams. We sampled 53 streams throughout Illinois and examined relationships between macroinvertebrate community structure and numerous physicochemical parameters. Streams were clustered into four major groups based on taxa dissimilarity. Habitat quality and dissolved nutrients were responsible for separating the major groups in a nonmetric multidimensional scaling ordination. Furthermore, the alignment of environmental factors in the ordination suggested there was a habitat quality–nutrient concentration gradient such that streams with high-quality habitats usually had low concentrations of nutrients. Discrimination by community measures further validated the major stream groups and indicated that forested streams had generally higher biological integrity than agricultural streams, which in turn had greater integrity than urban streams. Our results demonstrate that physical habitat degradation and nutrient pollution are important and often confounded determinants of biotic integrity in Illinois streams. In addition, we suggest that management of Midwestern streams could benefit from further implementation of multivariate data exploration and stream classification techniques.

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The dramatic impacts that humans have had on lotic ecosystems are exemplified in the midwestern USA, where agricultural and urban activities dominate the landscape. Although anthropogenic impacts in this region have long been known to negatively affect aquatic biota (DeWalt et al., 2005), the impacts of specific stressors, and combinations thereof, that most influence biological integrity at large scales require more attention. For example, physical habitat degradation may be most responsible for impairment of biological integrity in some streams (Master et al., 2000), yet some researchers have found that nutrients have the largest effect on stream organisms (e.g., Wang et al., 2007).

Biological monitoring techniques represent a departure from more standard chemical monitoring of systems, which may not account for stressor variability or nonpoint sources of impairment (Scrimgeour and Wicklum, 1996; Gerritsen et al., 1998). Biomonitoring approaches directly measure biological components of systems, such as community and functional structure. This is advantageous because biota integrate conditions over time and space and are directly linked to ecosystem function and integrity (Fausch et al., 1990; Loeb and Spacie, 1994; Resh et al., 1996).

Relationships between physicochemical characteristics and macroinvertebrates in wadeable streams are generally not well documented (Wang et al., 2007), and this is especially true in Illinois. Degraded physical habitat quality has been linked to decreased diversity and altered stream function (e.g., Newbold et al., 1983; Webster and Ehrman, 1996). Reduced substratum size and stability and decreased organic matter retention have historically been identified as major detrimental characteristics of habitat degradation (e.g., Egglishaw, 1964; Minshall, 1984). Elevated nutrient concentrations have been linked to poor biotic integrity in streams (Miltner and Rankin, 1998), and the mechanism is generally considered to be stimulation of excess primary production, which can degrade habitat, alter food resources, and deplete dissolved oxygen (Wang et al., 2007). However, the interplay of these factors and their ultimate influence on biotic integrity are in need of further study, and understanding these relationships is central to managing and assessing streams.

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Abbreviations: DRP, dissolved reactive phosphorus; NMDS, nonmetric multidimensional scaling; MBI, macroinvertebrate biotic index.

Table 1. Sample stream locations, Illinois Environmental Protection Agency (IEPA) codes, a priori predominant land use codes, and dates sampled for macroinvertebrates in 2005.

Stream	Location	IEPA code	a priori Code†	Date sampled
Apple River	near Hanover	MN-03	ag6	8 Mar.
Bankston Creek	near Harrisburg	ATGC-01	ag13	5 Mar.
Bay Creek	at Nebo	KCA-01	ag16	7 Mar.
Bear Creek	near Marcelline	KI-02	ag33	8 Mar.
Big Bureau Creek	near Princeton	DQ-03	ag15	8 Mar.
Big Creek	near Balcom	IXJ-02	for7	5 Mar.
Big Creek	near Bryant	DJB-18	ag24	7 Mar.
Big Ditch	near Dewey	EZU-01a	ag7	14 Mar.
Big Muddy River	near Mt. Vernon	N-08	ag14	6 Mar.
Black Slough	near Philo	BETA	ag10	15 Mar.
Bonpas Creek	near Browns	BC-02	ag1	15 Mar.
Cache River	near Forman	AD-02	for2	5 Mar.
Cahokia Canal	near Collinsville	JN-02	urb2	7 Mar.
Canteen Creek	near Collinsville	JNA-01	urb3	7 Mar.
Clear Creek	near Elbridge	BM-02	ag3	15 Mar.
Crab Orchard Creek	at Marion	ND-04	ag17	5 Mar.
Crooked Creek	near Newton	BEG-01	for1	15 Mar.
Dupage River	at Shorewood	GB-11	urb12	10 Mar.
E. Fk. Kaskaskia River	near Sandoval	OK-01	ag30	6 Mar.
E. Fk. Lamoine River	near Colchester	DGL-03	for8	8 Mar.
Elkhorn Creek	near Penrose	PH-16	ag23	9 Mar.
Elm River Drainage Ditch	near Tom's Prairie	CD-01	ag2	15 Mar.
Farm Creek	at E. Peoria	DZZP-03	urb4	8 Mar.
Hickory Creek	near Bluff City	ON-01	ag29	6 Mar.
Hurricane Creek	near Mulberry Grove	OL-02	ag26	5 Mar.
Hutchins Creek	near Bald Knob	ICE-01	for4	6 Mar.
Indian Creek	near Wyoming	DJL-01	ag5	8 Mar.
Kankakee River	at Momence	F-02	urb6	13 Mar.
Kickapoo Creek	at Waynesville	EIE-04	ag12	14 Mar.
Kishwaukee River	at Marengo	PQ-10	for6	9 Mar.
Little Kickapoo Creek	near Bloomington	–‡	ag11	14 Mar.
Lusk Creek	near Eddyville	AK-02	for3	5 Mar.
McDonald Creek	at Mt. Prospect	GR-01	urb7	10 Mar.
McKee Creek	near Chambersburg	DE-01	ag25	7 Mar.
Mid. Fk. Vermillion River	near Penfield	BPK-07	for9	13 Mar.
Midlothian Creek	near Midlothian	–	urb1	10 Mar.
N. Fk. Saline River	near Texas City	ATF-04	ag27	5 Mar.
N. Fk. Vermillion River	near Bismark	BPG-09	ag21	13 Mar.
Plum Creek	near Crete	HBE-01	for5	13 Mar.
Rayse Creek	near Waltonville	NK-01	ag31	6 Mar.
S. Br. Kishwaukee River	near Fairdale	PQC-06	ag8	9 Mar.
S. Fk. Saline River	near Carrier Mills	ATH-05	ag27	5 Mar.
S. Fk. Vermillion River	near Fairbury	DS-06	ag22	13 Mar.
Saline Branch Salt Fork	near Mayview	BPJC-06	ag18	14 Mar.
Salt Creek	at Western Springs	GL-09	urb11	10 Mar.
Shoal Creek	near Breese	OI-08	ag32	6 Mar.
Sisinawa River	near Galena	–	ag19	9 Mar.
Skokie River	at Lake Forest	–	urb8	10 Mar.
Spring Creek	at Algonquin	–	urb9	10 Mar.
Spring Creek	at Springfield	EL-01	urb5	7 Mar.
Spring Creek	near Gilman	–	ag9	13 Mar.
Thorn Creek	at Thornton	HBD-04	urb10	10 Mar.
Whitley Creek	in Moultrie County	–	ag4	15 Mar.

† ag, agricultural; for, forested; urb, urban.

‡ A dash indicates that a code does not exist for a stream.

The objectives of this study were to identify stressors that most strongly influence the biotic integrity of stream macroinvertebrate communities and to determine what stressors most influence the macroinvertebrate communities of specific stream types. We used multivariate techniques to identify patterns of macroinvertebrate composition and factors influencing them. We used metrics of community structure, function, and pollution tolerance to assess biotic integrity. We predicted that the degree of nutrient enrichment and the amounts and types of habitat degradation would vary across Illinois in relation to land use and that this would be reflected in macroinvertebrate community assemblages. We further hypothesized that, across all sites, physical habitat quality would be the most important factor controlling biotic integrity among the streams.

Materials and Methods

Sampling and Processing

This study was designed to gather representative data from gauged, wadeable streams from each major physiogeographic region of Illinois (Table 1; Fig. 1). A total of 53 streams were analyzed from May 2004 to March 2005. At each site, water temperature, pH, specific conductance, and dissolved oxygen concentration were measured with calibrated, portable meters. Water samples for analysis of sestonic chlorophyll-*a* and dissolved nutrients were collected from the center of the channel. Water samples were kept in the dark and on ice until they could be processed and preserved according to standard methods (American Public Health Association, 2005). Samples were analyzed for dissolved organic carbon, dissolved reactive phosphorus (DRP), nitrate-nitrogen (NO_3–N), ammonium-nitrogen (NH_4–N), and chloride after filtration through a precombusted 0.45-μm membrane filter. Total P and total N concentrations were determined on unfiltered water samples after appropriate digestion (American Public Health Association, 2005). Chloride and NO_3–N were determined with ion chromatography (DX-120; Dionex, Sunnyvale, CA). Dissolved reactive P, total P, NH_4–N, and total N were determined colorimetrically using a QuikChem 8000 (Lachat, Loveland, CO). Dissolved organic carbon was determined on a Dohrmann total organic carbon analyzer. Additionally, we compiled historical water quality data that were collected between 1972 and 2005 by the Illinois Environmental Protection Agency, Illinois State Water Survey, United States Fish and Wildlife Service, or University of Illinois Department of Natural Resources and Environmental Sciences. The number of samples from each site ranged from 22 to 321, with 150 to 200 samples being most common. A linear regression between pre- and post-1990 N and P showed that concentrations were consistent through both time periods ($r^2 > 0.9$). Therefore, we averaged all available nutrient data for each site to provide a robust assessment of nutrient availability.

For physical habitat assessments, we first quantified the composition of the substrata of each study reach by taking 100 random grabs (Wolman, 1954) and classifying substratum size using a modified Wentworth scale (Cummins, 1962). Second, we conducted semi-quantitative habitat surveys by scoring physical features (riffle-run-pool morphology, sinuosity, substrata, and riparian condition) of the reach to obtain overall habitat scores based on procedures of Barbour et al. (1999). Each of nine habitat features was given a score of 0 to 20, with a total score of 180 being the best possible score.

We sampled benthic macroinvertebrates from 8 to 18 Mar. 2005 following USEPA multihabitat Rapid Bioassessment Protocols for wadeable streams (Barbour et al., 1999). We took 20 jabs with a 0.3 by 0.5 m, 500-μm mesh dip net from each 100-m study reach. A jab consisted of disturbing the substrata for 0.5 m upstream of the net to a depth of ~10 cm in riffle and run habitats. In pools, the net was bumped through ~10 cm of substrata for 0.5 m. The 20 jabs were partitioned among pool, riffle, and run habitats within each study reach (e.g., a reach with 50% riffle habitat had 10 jabs taken from riffles). All jabs were combined into one composite sample, which was immediately rinsed through a 500-μm sieve and preserved in a 10% formalin solution.

A two-stage subsampling procedure was used to remove invertebrates from samples in the laboratory (Vinson and Hawkins, 1996). First, relative abundances were estimated by randomly removing 300 organisms, when available, using a numbered, gridded pan and random number generator. Next, the unsearched portion of the sample was searched for taxa not found in the original 300 to obtain a comprehensive assessment of richness. With the exception of chironomids and several non-insect taxa (e.g., oligochaetes), organisms were identified to genus.

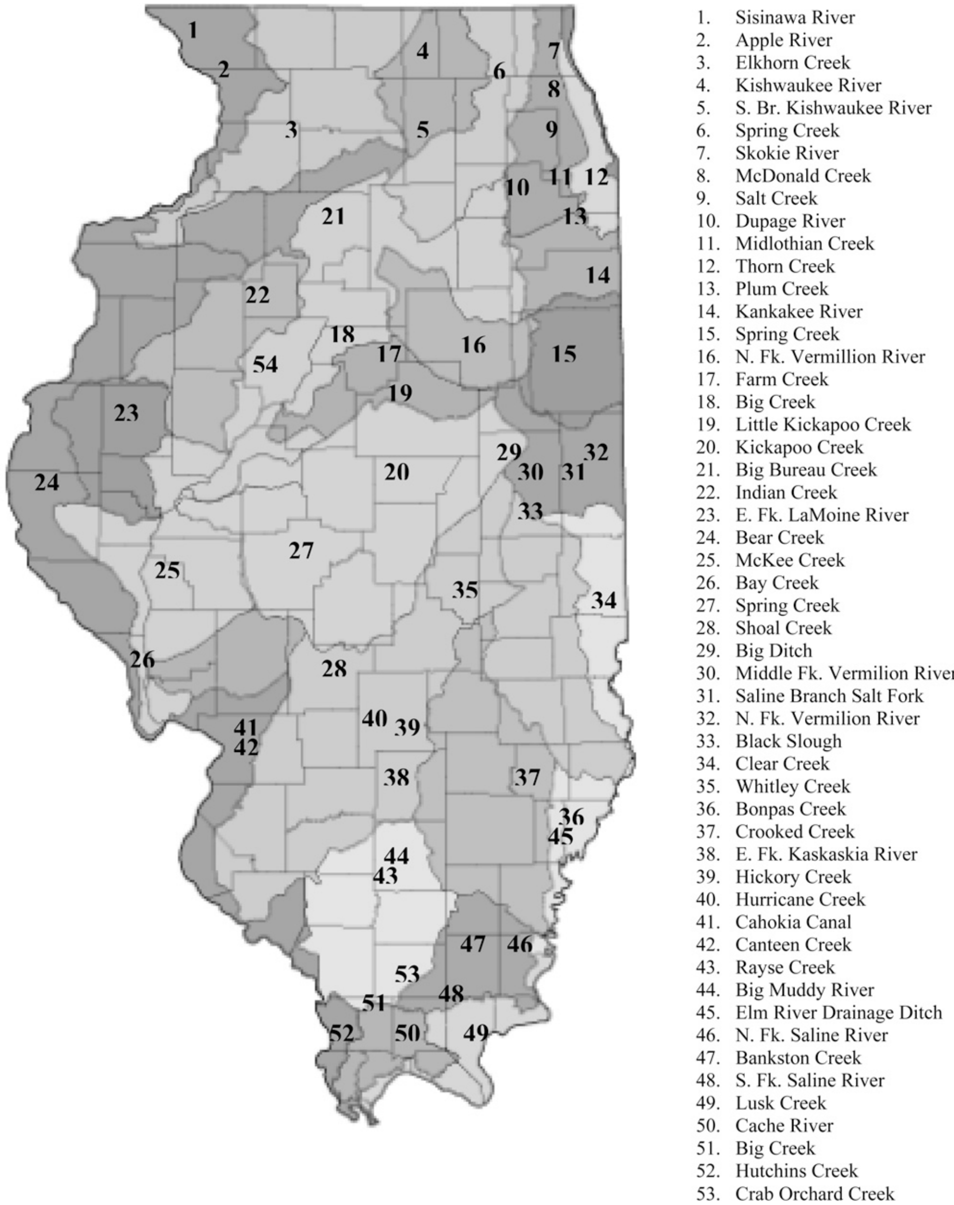

Fig. 1. Map of Illinois showing the approximate locations of the 53 wadeable streams sampled. Colors indicate major watersheds. Map created by Sally A. McConkey and Kathleen J. Brown, Illinois State Water Survey. Map and legend for watersheds can be found at http://www.watershed.uiuc.edu/getting_involved/map.cfm.

Data Analyses

We organized the 53 streams into discrete groups using an agglomerative hierarchical cluster analysis (PC-ORD; MJM Software Design, 2005). Bray-Curtis dissimilarity (Bray and Curtis, 1957) was used for the dissimilarity matrix due to its robustness for community data. The flexible beta method (value = 0.25) was used to link clusters because it is compatible with Bray-Curtis dissimilarity and because space-distorting properties can be controlled (McCune and Grace, 2002). We used Wishart's (1969) distance function to scale the resulting dendrogram based on the amount of information lost at each step in the analysis. We pruned the dendrogram in what in our estimation was the most ecologically meaningful manner.

We compiled a nonmetric multidimensional scaling (NMDS) ordination on presence-absence macroinvertebrate data using DECODA (Minchin, 1989). Ordinations are effective for summarizing multidimensional community data (McCune and Grace, 2002), and NMDS is more robust and effective for ordination of community data than previous methods, such as detrended correspondence analysis or canonical correspondence analysis (Minchin,

Table 2. Macroinvertebrate metrics calculated for Illinois streams and expected direction of change with increasing anthropogenic disturbance.

Metric	Definition	Expected change
Richness	number of distinct taxa	decrease
Shannon diversity	index that reflects richness and evenness of a community	decrease
MBI	macroinvertebrate biotic index, based on Hilsenhoff (1987), modified for Illinois; reflects tolerance to pollution	increase
EPT	number of distinct taxa in the orders Ephemeroptera, Plecoptera, and Trichoptera	decrease
% EPT	relative abundance of members of the orders Ephemeroptera, Plecoptera, and Trichoptera	decrease
% Oligochaeta	relative abundance of aquatic worms	variable
% Chironomidae	relative abundance of midges (Diptera: Chironomidae)	increase
% Gatherer	relative abundance of the functional group gatherers	variable
% Filterer	relative abundance of the functional group filterers	increase
% Scraper	relative abundance of the functional group scrapers	decrease
% Shredder	relative abundance of the functional group shredders	decrease
% Dominance	relative abundance of the most common taxon	increase

1987; Battaglia et al., 2002). The distance matrix was calculated using Bray-Curtis dissimilarity. We chose the best ordination (lowest stress solution) from 100 random starting configurations and 1000 iterative adjustments of each starting configuration. The proper number of dimensions was determined by plotting stress values versus dimensions. We also used the vector-fitting procedure in DECODA. This is a type of correlation analysis that gives a confident estimate of the variables responsible for separating sites in ordination space (Kantlivas and Minchin, 1989).

To evaluate the relative biotic integrity of our groups of streams, we used assemblage-level bioassessment metrics. Metrics were chosen to reflect diversity, richness, tolerance, and functional structure and composition (Table 2). The macroinvertebrate biotic index (MBI) is a version of the Hilsenhoff Biotic Index (Hilsenhoff, 1987) that has been calibrated for Illinois taxa (Fitzpatrick et al., 2004). An ANOVA in conjunction with a Tukey-Kramer honestly significant difference test was used to test for differences in metric scores among the groups of streams.

Results

A total of 14,015 macroinvertebrates, comprising 85 taxa, were identified. We were able to remove 300 macroinvertebrates from the composite samples in most cases. Samples containing less than 300 total macroinvertebrates were typically from highly degraded streams. For example, Farm and Canteen creeks drain predominantly industrial areas, and samples from these sites contained only 42 and 53 macroinvertebrates, respectively. The number of macroinvertebrates identified was not a significant vector-fitting variable to the ordination, indicating that sampling bias was not a factor in our results.

Richness of the study streams ranged from 7 to 29. Taxa present in nearly every stream included *Tubifex* (Oligochaeta: Tubificidae), several subfamilies of midges (Diptera: Chironomidae), *Simulium* (Diptera: Simuliidae), *Stenelmis* (Coleoptera: Elmidae), *Baetis* (Ephemeroptera: Baetidae), and *Cheumatopsyche* (Trichoptera: Hydropsychidae). In contrast, taxa such as *Palaemonetes* (Decapoda: Palaemoniidae), *Nigronia* (Neuroptera: Corydalidae), *Attenella* (Ephemeroptera: Ephemerellidae), *Ptilostomus* (Trichoptera: Phryganeidae), and *Amphinemura* (Plecoptera: Nemouridae) were collected from only one or two streams.

Four groups of streams were assembled according to their compositional similarity (Fig. 2). Most of the sites we examined drained watersheds that were dominated by a single land use, which allowed us categorize the sites by this parameter and examine general patterns in land use among the four groups identified in the dendrogram (Fig. 2). Most of the sites draining agricultural land were associated with Group 1, and most of the forested sites occurred in Group 2. Group 3 contained most of the urban streams, although two highly degraded urban streams were identified as a distinct group (Group 4) and were dissimilar from other streams. Streams in geographic proximity often did not cluster together, although there was tendency for this to happen with forested sites in southern Illinois and with urban sites in the northeastern portion of the state.

Concentrations of nutrients and dissolved ions were generally lowest in Group 2, followed by Groups 1, 4, and 3 (Table 3). Group 3 had a much higher concentration of DRP than the other groups. Taxa richness ($F_{[3,30]} = 9.40$; $P < 0.0001$) was higher in Group 1 and 2 streams than in Groups 3 and 4 (Table 4). Similarly, Shannon diversity ($F_{[3,30]} = 5.03$; $P = 0.004$) was greater in Groups 1 and 2 than in Group 4. Group 2 had a lower MBI score ($F_{[3,30]} = 11.04$; $P < 0.0001$) than Groups 3 and 4, and Group 1 had a lower MBI score than Group 4. The percent Oligochaeta metric ($F_{[3,30]} = 10.97$; $P < 0.0001$) was higher in Group 4 than in Groups 1 and 3, which in turn were higher than in Group 2. Groups 1 and 2 had more Ephemeroptera, Plecoptera, and Trichoptera taxa ($F_{[3,30]} = 2.99$; $P = 0.039$) than Groups 3 and 4. Overall, patterns suggested the highest biotic integrity in forested streams and the lowest in urban streams.

We identified the same NMDS solution for 98 of 100 random starting configurations. Three dimensions were necessary to achieve an acceptable stress level of 0.182. Additional dimensions did not greatly reduce this stress. Of 18 measured environmental variables, seven had significant effects on the ordination (Table 5; Fig. 3). Habitat measures (survey score and substrate stability) as well as nutrient and dissolved ion measures correlated in an antagonistic manner to the ordination (Fig. 3). Groups 1 and 2, determined from the cluster analysis, aligned best with habitat quality and stable substrata, whereas Group 3 aligned best with higher nutrient and dissolved ion concentrations. Group 4 was notably isolated in the ordination and seemed to be aligned best with the percentage of run habitat.

Discussion

As predicted, differences in macroinvertebrate assemblages of Illinois streams seemed to be a function of physical habitat and nutrient concentrations. However, we were unable to support our hypothesis that habitat quality would be the primary factor governing biotic integrity because habitat degradation was generally evident in streams with elevated nutrient concentrations. Our analyses indicate that habitat quality and nutrient concentrations were of equal importance in structuring macroinvertebrate communities across Illinois streams. In a similar study, Wang et al. (2007) found that macroinvertebrate assemblage measures were strongly linked to physical parameters and nutrient concentrations in wadeable Wisconsin streams. Our study provides further evidence that nutri-

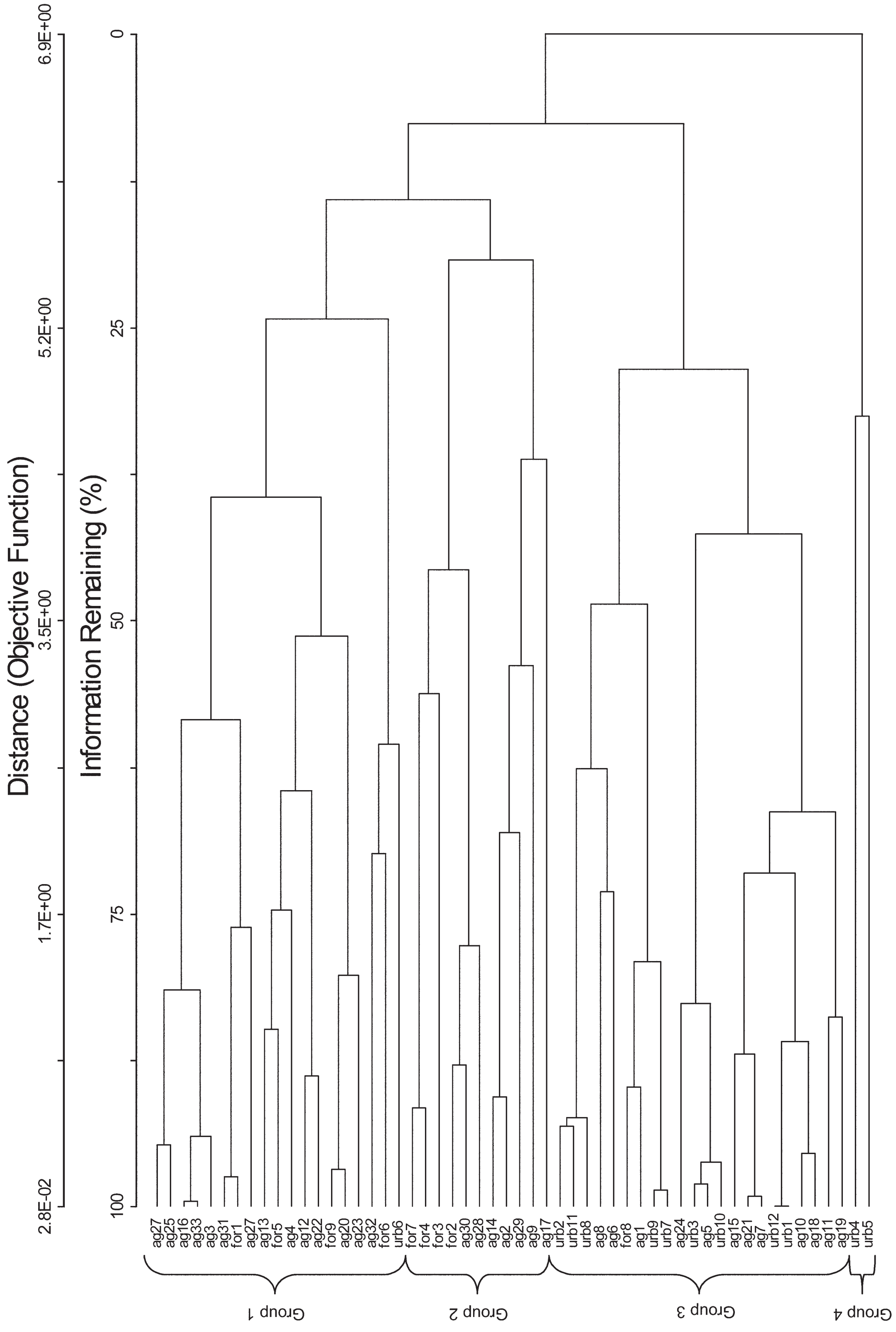

Fig. 2. Dendrogram of Illinois study streams resulting from a hierarchical cluster analysis of benthic macroinvertebrate presence–absence data. Streams were separated into the four groups shown based on Bray-Curtis dissimilarity. Stream site codes are from Table 1.

Table 3. Mean (SE) values of water chemistry parameters for four groups of streams as determined by cluster analysis.

	Group			
Parameter	1	2	3	4
Chloride, mg L^{-1}	40 (12)	35 (16)	92 (17)	56 (17)
Sulfate, mg L^{-1}	38 (14)	12 (4)	44 (7)	NA†
NO_3–N, mg L^{-1}	3.7 (0.8)	1.1 (0.6)	5.1 (0.6)	4.9 (0.8)
NH_4–N, mg L^{-1}	0.2 (0.0)	0.2 (0.0)	0.4 (0.1)	0.2 (0.1)
DRP,‡ mg L^{-1}	0.08 (0.01)	0.06 (0.01)	0.53 (0.18)	0.17 (0.04)
Total P, mg L^{-1}	0.23 (0.04)	0.22 (0.04)	0.64 (0.03)	0.3 (0.01)
DOC,§ mg L^{-1}	3.9 (0.5)	5.8 (1.2)	3.8 (0.5)	NA
Silica, mg SiO_2 L^{-1}	10.4 (1.0)	9.3 (1.7)	8.7 (0.9)	NA
pH	7.5 (0.2)	7.3 (0.1)	7.8 (0.1)	7.9 (0.3)
Specific conductivity, µS cm^{-1}	668 (47)	581 (123)	895 (87)	724 (115)

† NA indicates data were unavailable.

‡ DRP, dissolved reactive P.

§ DOC, dissolved organic C.

ent pollution is related to degraded biotic integrity in streams (e.g., Miltner and Rankin, 1998), although separating the effects of nutrients per se from other factors remains problematic.

The inverse relationship that we observed between habitat quality and nutrient concentrations is intuitive considering the impacts of row crop agriculture, which constitutes ~95% of the land use in the northern two thirds of Illinois (DeWalt et al., 2005). A combination of channelization and tile drainage results in habitat impairment through scouring, sedimentation, habitat homogenization, and altered riparian vegetation (Cooper, 1993). Furthermore, widespread subsurface tile drainage in Illinois enhances transport of N (Gentry et al., 1998; Royer et al., 2006), P (Gentry et al., 2007; Stamm et al., 1998), and numerous agrochemicals (David et al., 2003) into streams that also receive nutrient inputs from surface runoff. Both N and P were identified as predictors of macroinvertebrate assemblages, but in many Illinois streams, N concentrations are sufficiently high to preclude N limitation of algae (Morgan et al., 2006). This suggests P may be the critical nutrient affecting biotic integrity among the sites, particularly in agricultural regions of the state.

Streams with the lowest biotic integrity were located in urban/residential areas. This has major implications for the state of Illinois because nearly seven million people live within a six-county area around Chicago (DeWalt et al., 2005). We also sampled urban streams near Peoria, Springfield, East St. Louis, and numerous smaller population centers. As would be expected in streams draining watersheds with extensive impervious area, most had high concentrations of nutrients (Paul and Meyer, 2001; Meyer et al., 2005; Moore and Palmer, 2005). Channelization is the most common form of habitat modification in urban streams (Tavzes et al., 2006), and the urbanized streams in our study had the poorest physical habitat quality. Among other detrimental effects, channelization may cause habitat homogenization and reduced biodiversity because of reduced niche potential (Beisel et al., 2000). Farm Creek and Spring Creek provided clear examples of high degrees of impairment from urbanization, as indicated by their poor metric values and status as outliers in the ordination. Farm Creek drains much of East Peoria, and Spring Creek drains the city of Springfield. These two streams were dominated by a few pollution-tolerant, non-insect taxa (e.g., sowbugs [Isopoda]), whereas agricultural streams typically were dominated by chironomids and oligochaetes.

Streams in southern Illinois represented the opposite end of the habitat–nutrient gradient. These streams drain primarily forested catchments in the Shawnee National Forest, which experience relatively little anthropogenic disturbance, have high-quality physical habitat, and have much lower nutrient concentrations than streams from other areas in the state. In particular, Lusk and Hutchins creeks have high substratum and habitat heterogeneity, intact riffle-run-pool morphology, and forested riparian zones. These factors likely contributed to the high richness and diversity scores, the high numbers of pollution intolerant taxa, and the low nutrient concentrations observed in Lusk and Hutchins creeks. Wang et al. (1997) found that fish community integrity improved with increasing forested land area in Wisconsin, and our study indicates a similar relationship for stream macroinvertebrates in Illinois.

The pattern we observed of lower nutrient concentrations in streams with better physical habitat quality was in some cases (e.g., streams draining undisturbed forested catchments) clearly related to a lack of human impacts on each factor. However, in some cases this relationship may also be related to direct and indirect interactions between physical habitat structure and nutrient dynamics. In addition to directly increasing biological activity and thus nutrient uptake (e.g., Newbold et al., 1983; Webster and Ehrman, 1996), high physical habitat and substratum quality can indirectly result in decreased nutrient export through enhanced physical retention, which allows for increased biological uptake (Peterson et al., 2001).

An important implication of this study for Illinois stream bioassessment is that that similarity of macroinvertebrate communities in streams was not linked to the geographic proximity of the

Table 4. Mean (SE) values of bioassessment metrics that significantly discriminated among four groups of streams as determined by cluster analysis.

	Group			
Parameter	1	2	3	4
Richness	18.9a† (1.1)	20.8a (1.4)	14.4b (1.0)	7.0b (3.2)
MBI	7.1ab (0.2)	6.2a (0.3)	7.8b (0.2)	9.5c (0.7)
EPT richness	6.1a (0.5)	6.9a (0.7)	2.1b (0.5)	0.5b (1.6)
Shannon diversity	1.8a (0.1)	1.8a (0.1)	1.5ab (0.1)	0.8b (0.3)
% Oligochaeta	17.3abc (4.1)	9.1b (5.4)	31.2c (3.8)	79.9d (12.7)

† Letters after mean values indicate significant differences among stream groups (Tukey-Kramer's honestly significant difference test, $P < 0.05$). Descriptions of metrics appear in Table 2.

Table 5. Significant vectors of the nonmetric multidimensional scaling ordination performed on macroinvertebrate presence-absence data for Illinois streams and correlation coefficients (*r*) and significance (*P*).

Vector	*r*	*P*
Habitat quality	0.64	<0.0001
% stable substrate	0.62	<0.0001
% run habitat	0.45	0.009
NO_3–N concentration	0.60	<0.0001
DRP† concentration	0.50	0.005
pH	0.64	0.001
Silica concentration	0.65	0.012

† DRP, dissolved reactive P.

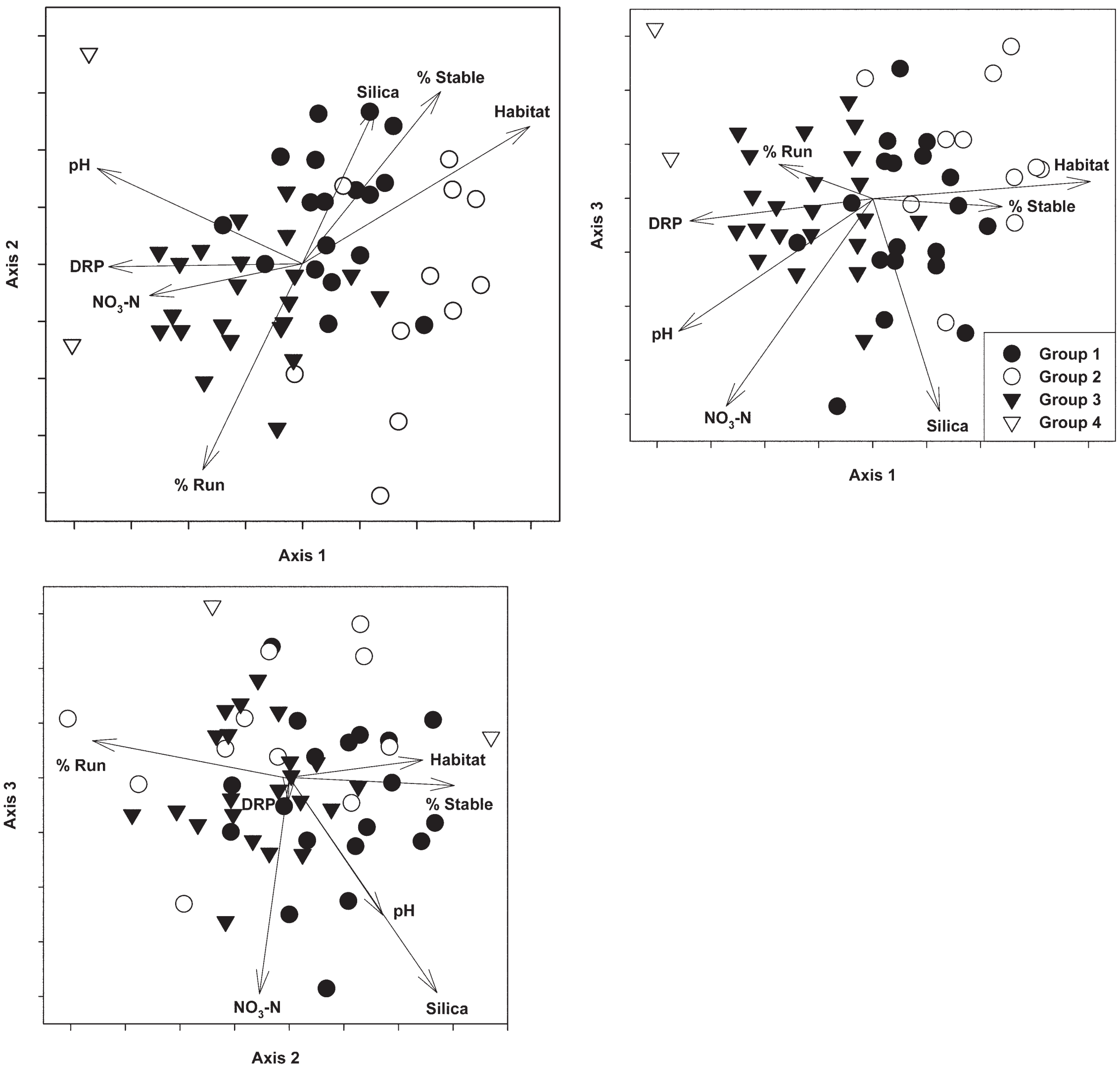

Fig. 3. Nonmetric multidimensional scaling ordination of presence–absence data for benthic macroinvertebrate taxa in Illinois streams. Graphs represent Axis 1 vs. Axis 2, Axis 1 vs. Axis 3, and Axis 2 vs. Axis 3. The stream groups were determined by cluster analysis (Fig. 2). Correlation coefficients and *P* values for vectors are listed in Table 4. DRP, dissolved reactive P.

streams. Previous studies also have reported that streams in the same geographic region do not necessarily contain similar macroinvertebrate assemblages (Parsons and Norris, 1996; Reynoldson et al., 1997; Turak et al., 2000), which is evidence that a strictly ecoregional approach to stream classification and reference choice may prove problematic. We therefore suggest using compositional dissimilarity measures with multivariate analyses to avoid problems that could arise if streams in geographic proximity are assumed to have the same biotic potential.

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