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RESEARCH ARTICLE

Landscape characteristics affecting streams in urbanizing regions of the Delaware River Basin (New Jersey, New York, and Pennsylvania, U.S.)

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Abstract Widespread and increasing urbanization has resulted in the need to assess, monitor, and understand its effects on stream water quality. Identifying relations between stream ecological condition and urban intensity indicators such as impervious surface provides important, but insufficient information to effectively address planning and management needs in such areas. In this study we investigate those specific landscape metrics which are functionally linked to indicators of stream ecological condition, and in particular, identify those characteristics that exacerbate or mitigate changes in ecological condition over and above impervious surface. The approach used addresses challenges associated with redundancy of landscape metrics, and links landscape pattern and composition to an indicator of stream ecological condition across a broad area of the eastern United States. Macroinvertebrate samples were collected during 2000–2001 from forty-two sites in the Delaware River Basin, and landscape data of high spatial and thematic resolution were obtained from photointerpretation of 1999 imagery. An ordination-derived 'biotic score' was positively correlated with assemblage tolerance, and with urban-related chemical characteristics such as chloride concentration and an index of potential pesticide toxicity. Impervious surface explained 56% of the variation in biotic score, but the variation explained increased to as high as 83% with the incorporation of a second land use, cover, or configuration metric at catchment or riparian scales. These include land use class-specific cover metrics such as percent of urban land with tree cover, forest fragmentation metrics such as aggregation index, riparian metrics such as percent tree cover, and metrics related to urban aggregation. Study results indicate that these metrics will be important to monitor in urbanizing areas in addition to impervious surface.

Keywords Urbanization · Macroinvertebrate · Landscape pattern · Land cover · Land use · Water quality · Impervious surface

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Introduction

The amount of developed land, including urban, suburban, and ex-urban areas, grew from 10.1 to 13.3% of the landscape in the coterminous United States from 1980 to 2000, at a rate of approximately 1.6% per year (Theobald 2005). Such urbanization has been shown to negatively affect stream ecosystems (e.g., Paul and Meyer 2001; Luck and Wu 2002). Loss of native biodiversity, and other alterations of biological assemblage structure and function often result from urban-related stressors such as habitat loss, alteration of hydrologic and thermal regimes, and changes in water chemistry (McMahon and Cuffney 2000; Paul and Meyer 2001; Kennen et al. 2005; Meador et al. 2005; Brown et al. 2009). Declines in biological, physical, and chemical indicators of stream health have been linked to increases in generalized indicators of catchment urban intensity such as percent impervious surface, percent urban land use, and road density, and to indices composed of multiple urban characteristics (e.g., Cuffney et al. 2005; Tate et al. 2005; Brown et al. 2009; Coles et al. 2010). Impervious surface, in particular, is an aggregated indicator of urban intensity that has been frequently identified as a key factor associated with declining stream water quality (Schueler 1994; Arnold and Gibbons 1996; FitzHugh 2002). Generalized urban metrics, however, are coarse predictors of ecological condition (Schueler 1994). Urbanization is accompanied by many specific land cover, use, and configuration changes that often have greater relevance to planners and managers because they can be more directly modified (Forman and Godron 1981; Alberti 2005; Kearns et al. 2005; Coles et al. 2010). They also typically have greater ecological relevance because of their more direct link to changes in ecosystem function than general measures of urbanization. For example, Cifaldi et al. (2004) identified spatial pattern characteristics of fragmentation, patch size, and patch interspersion to be important to stream water quality in addition to percent urban land use. Kleppel et al. (2004) related urban typology (small towns with village centers versus sprawling-type growth) to wetland health indicators. Alberti et al. (2007) demonstrated that both configuration and composition of impervious cover and forest land were important to stream biotic integrity in an Oregon watershed. Identifying specific landscape metrics that are functionally linked to indicators of ecological condition would be a great advantage for effective monitoring, assessment, and management (Kearns et al. 2005; Cushman et al. 2008).

It has been challenging to independently examine the effect of landscape patterns such as forest fragmentation on stream condition because of metric redundancy (Griffith et al. 2000; Kearns et al. 2005), or co-linearity with measures of forest loss (e.g., Fahrig 2003; Cushman et al. 2008). In addition, land cover and land use metrics are often highly correlated with each other (e.g., Hargis et al. 1998), confounding both the distinct impact each can have on water quality as well as the distinct management implications each can offer (Alberti et al. 2007). Finally, riparian zone characteristics are often highly correlated with catchment-wide characteristics.

The purpose of our study was to identify specific landscape characteristics that influence stream ecosystem health in a major watershed in the eastern United States. This study took advantage of the substantial increase in multi-scale land use and land cover data made available for the Delaware River Basin (DRB), located in the rapidly-urbanizing eastern United States, through collaboration between U.S. Geological Survey and U.S. Forest Service (Murdoch et al. 2008). We describe major components of landscape variation among the study catchments, and identify characteristics and representative variables that are functionally-related to macroinvertebrate assemblage variation across the study area. Macroinvertebrates are widely used indicators of stream ecological quality (Rosenberg and Resh 1993), and exhibit predictable changes in composition and structure across gradients of urbanization (e.g., Kennen 1999; Roy et al. 2003; Cuffney et al. 2005; Brown et al. 2009; Coles et al. 2010).

Our strategy was to select a set of streams draining catchments with different degrees of urban intensity, quantify and describe macroinvertebrate assemblage variation among these sites, generate a non-redundant set of detailed landscape data, and relate specific landscape characteristics to observed macroinvertebrate patterns. We conduct the study on a set of streams that have generally similar background conditions, and that provide a full range of urban intensity across the study area. Urban intensity can be quantified in various ways. In this study, we use road density as an a priori indicator of urban intensity to aid in site selection, but the analyses use photointerpretated landscape composition and configuration variables. Photointerpretation of relatively high resolution imagery generates landscape data at appropriately fine scales, providing more spatially- and thematically-accurate information regarding landscape composition and configuration characteristics than is generally available across such a broad area.

Methods

Study area

The DRB covers 32893 km^2, and includes parts of Pennsylvania, New Jersey, New York, and Delaware (Fig. 1). It contains eight ecoregions (Omernik Level III classification, Omernik 1987, Fig. 1); erosional streams draining upland catchments are common throughout all except for the Coastal Plain ecoregion. The climate is temperate; average yearly temperature ranges from about 7.2°C in the north to 13.3°C in the south. Average annual precipitation ranges from 127 cm in the north to 107 cm in the south with little seasonal variation (Jenner and Lins 1991), but snowpack results in seasonally-variable runoff. The overall population of the basin has increased 15% since 1970, and the amount of land area classified as urban, suburban or exurban (house densities greater than 6 per sq. km—as used by Radeloff et al. 2005 and Theobald 2005) has increased from 63 to 80%. Agricultural and forested land is rapidly being replaced with housing and commercial developments in large portions of the study area, such as the greater Philadelphia metropolitan area. Forest is being replaced and intermixed with residential development in the Appalachian Plateau ecoregions to the north where some of the most rapidly-growing counties in Pennsylvania are located (e.g., Pike County, with 1990–2000 population growth rates of 66%, U.S. Census Bureau 2000). A relatively broad continuum of urban intensity throughout the study area includes catchments that are almost completely forested as well as those with high-density urban centers.

Site selection

Forty-two wadeable sites were selected, in catchments of 13–287 km^2 (median 47.7 km^2) in area (Fig. 1; Table 1). Catchments were selected to provide a gradient of forest to urban land use across the study area, while minimizing potentially confounding influences such as significant contribution of municipal or industrial effluents, and extensive agricultural land use on low-urban catchments. Selecting low-urban sites that were primarily forested (i.e., not agricultural) was done to avoid obscuring urban effects (Brown et al. 2009; Qian et al. 2010). Site selection criteria also included the presence of riffle habitat with rocky substrate; these habitats are expected to support a high diversity of macroinvertebrates (Moulton II et al. 2002).

Sample collection and processing

Benthic macroinvertebrates were collected from riffles during August–September of 2000 or 2001 by scrubbing rocks in a 500 cm^2 area in front of a modified Slack sampler (with 500 micron mesh) following Moulton II et al. (2002). Each composite sample (from five riffle locations at each site) was preserved in buffered formalin, and sent to the USGS National Water-Quality Laboratory (NWQL, Arvada, CO). Processing methodology (described in Moulton II et al. 2000) included identification of 300 randomly-selected specimens to species or the next lowest-possible taxonomic level (usually genus). Water samples were collected by an equal-area method during spring and summer base flow periods (during the macroinvertebrate collection year), following Shelton (1994). Samples were analyzed at the NWQL for nutrients, pesticides, and ions according to Fishman (1993).

Landscape characterization

Road-density data (calculated as the number of road kilometers per square kilometer of basin area) were obtained for the year 2000 from TIGER (Topologically Integrated Geographic Encoding and Referencing) line data developed for the U.S. Census. Land use, land cover, and landscape configuration data were developed for 32 of the 42 study catchments by photo-interpretation of 1999 black and white, leaf-on, digital orthophoto quarter quads (DOQQ) derived from 1:40,000 National Aerial Photo Program (NAPP) photography.

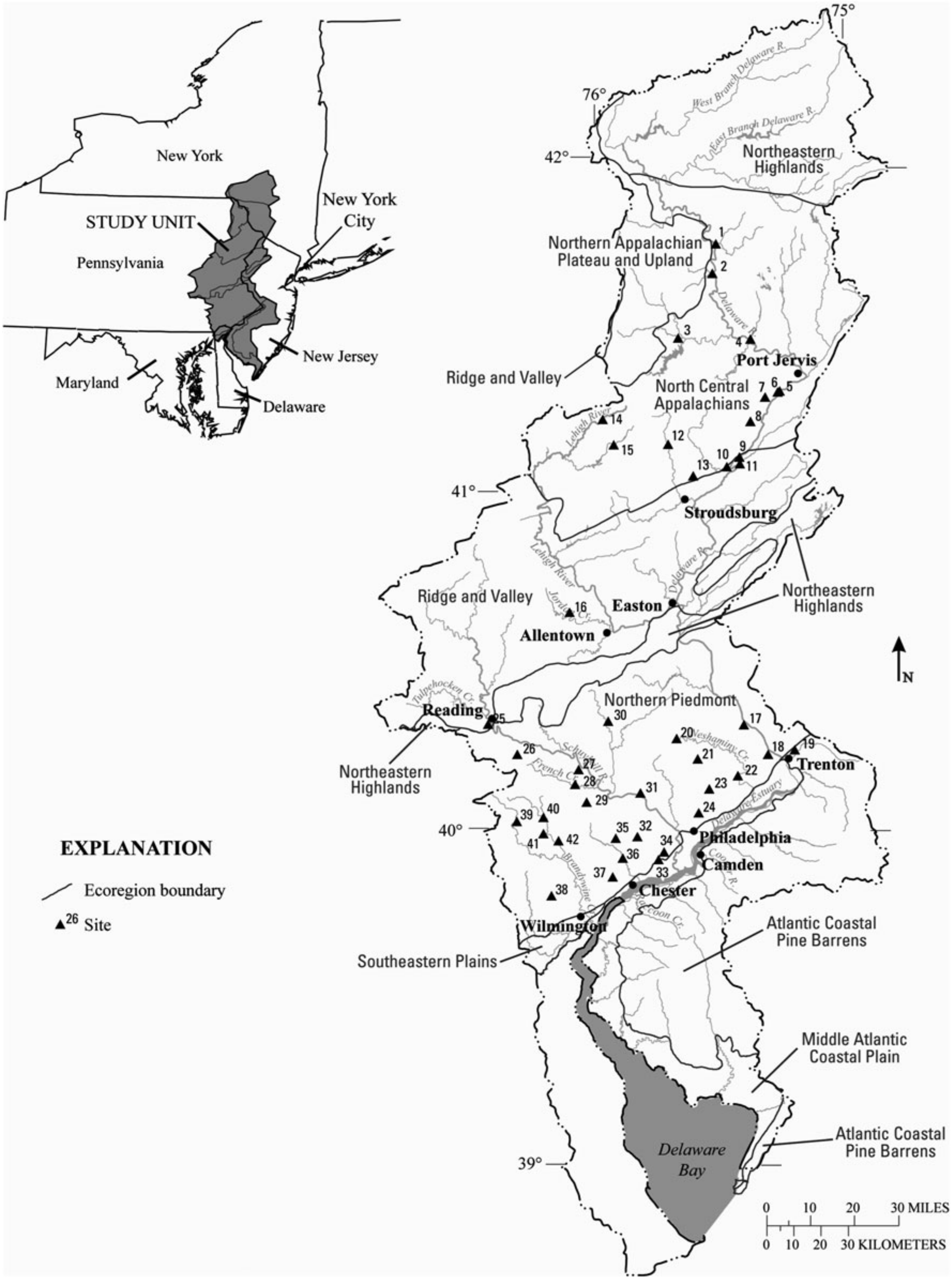

Fig. 1 Map of the DRB in the Eastern US showing sites from which macroinvertebrate, chemical, and physical samples, and field data were collected during 2000–2001. Numbers correspond with site names listed in Table 1

Table 1 List of DRB sites studied during 1999–2001

Map no.	Site name	Drainage area (km^2)	Road density (km/km^2)
1	Calicoon Creek at Callicoon, NY	287	3.0
2	Calkins Creek 1200 ft above mouth at Milanville, PA	114	2.2
3	Middle Creek at Hawley, PA	210	2.4
4	Halfway Brook at Barryville, NY	63	2.2
5	Vandemark Creek at mouth at Milford, PA	13	2.9
6	Sawkill Creek 2000 ft above mouth at Milford, PA	62	3.6
7	Raymondskill Creek below Swale Brook near Silver Springs, PA	57	4.9
8	Dingmans Creek below Fulmer Falls near Dingmans Ferry, PA	36	6.3
9	Toms Creek at Egypt Mills, PA	23	6.1
10	Little Bushkill Creek at Bushkill, PA	85	2.6
11	Flat Brook near Flatbrookville, NJ	168	1.9
12	Brodhead Creek near Mountain home, PA	104	2.9
13	Marshall's Creek near Marshall's Creek, PA	26	4.3
14	Lehigh River near Gouldsboro, PA	44	4.0
15	Tobyhana Creek at Warnertown, PA	54	4.1
16	Jordan Creek near Schnecksville, PA	135	4.8
17	Pidcock Creek near New Hope, PA	34	3.4
18	Buck Creek below Brock Creek at Yardley, PA	18	10.6
19	Shabakunk Creek near Lawrenceville, NJ	31	11.3
20	Pine Run at Chalfont, PA	31	6.4
21	Little Neshaminy Creek at Valley Road near Neshaminy, PA	70	7.3
22	Mill Creek near Langhorne, PA	36	10.2
23	Pennypack Creek at Paper Mill, PA	62	12.6
24	Tacony Creek at Cheltenham, PA	23	16.2
25	Wyomissing Creek at West Reading, PA	41	11.8
26	Hay Creek near Scarlets Mill, PA	49	3.3
27	Pigeon Creek at Parker Ford, PA	36	5.1
28	French Creek near Phoenixville, PA	153	3.6
29	Pickering Creek at Charlestown Rd bridge at Charlestown, PA	73	4.7
30	Macoby Creek at Green Lane, PA	44	4.8
31	Stony Creek at Steriger Street at Norristown, PA	52	8.3
32	Darby Creek at Foxcroft, PA	41	9.6
33	Darby Creek near Darby, PA	98	12.6
34	Cobbs Creek at East Landsdowne, PA	31	18.7
35	Crum Creek at Goshen Road near Whitehorse, PA	34	7.0
36	Ridley Creek near Media, PA	70	5.3
37	West Branch Chester Creek near Chester Heights, PA	47	5.8
38	East Branch Red Clay Creek near Five Points, PA	26	6.3
39	West Branch Brandywine Creek at Cedar Knoll, PA	62	4.1
40	East Branch Brandywine Creek near Dorlan, PA	83	3.9
41	Beaver Creek near Downingtown, PA	44	6.8
42	Valley Creek near Altor, PA	41	6.9

Map number corresponds with location shown on Fig. 1. Sites with photointerpreted data are indicated with bold map numbers

Fourteen land-use classes, and 12 subclasses were generated from visual interpretation of digital aerial photography. Land-cover proportions of tree, grass, and impervious surface were estimated within developed land-use polygons. Forest, rangeland, water, and barren classes were similar to Anderson Level I classes (Anderson et al. 1976); developed land-use classes were interpreted to more detailed levels. Percentages of tree, grass, and impervious cover (houses and roads) were estimated for the land area of each developed land-use polygon using a scaled dot-grid. A minimum mapping unit of 0.40 ha (1 acre) in area and 36.6 m (120 ft) in width was applied for developed land uses occurring within forested land, and a minimum mapping unit of 2.02 ha (5 acres) in area and 100.6 m (330 ft) in width was applied for several land-use combinations that commonly occur together and can be particularly time-consuming to delineate separately (e.g., forest within residential, forest within transportation, commercial within residential). Shrubs were included with trees due to the difficulty of distinguishing these in the non-stereo imagery. Bare ground occurred rarely, and was included with grass. Impervious ground cover was selected over canopy cover where both occurred (i.e., dots on roads were counted as impervious, even if trees blocked the view). This follows guidelines outlined in Philipson (1997). It also represents the most accurate photointerpretation method for impervious surface estimation of those described by Brabec et al. (2002) and provides information most closely aligned to that observed by a manager or planner on the ground.

Photointerpreted land use polygons and their land cover attributes were rasterized to a 30 × 30 m grid. Measures of landscape composition were calculated within each catchment, and within riparian corridors (60 m zone centered on the stream channel) of each catchment, using relatively simple spatial analyses (for land use and land cover composition, and road density data). Land use classes were aggregated into five groups (forest, agriculture, barren, natural non-forest vegetation, and urban-developed) before calculating landscape pattern metrics. The "natural non-forest vegetation" class includes several vegetated types (shrub, grassland, herbaceous and herbaceous wetland) that, in this region, do not represent a primarily human use. Landscape pattern metrics were generated with the Image ANalysis (IAN) program (DeZonia and Mladenoff 2004), and calculated only at the catchment scale. To ensure completeness in characterization of landscape pattern and configuration, metrics falling within each of the following general categories were quantified: (1) patch size distribution, (2) edge and interspersion, (3) connectedness and "clumpiness," and (4) overall heterogeneity, texture, and shape. These are based on metric groups described by Haines-Young and Chopping (1996), Betts (2000), and Lausch and Herzog (2002).

Data analysis

Developing the macroinvertebrate-based biotic score

We used an indirect ordination analysis approach to characterize variation in macroinvertebrate assemblage composition and structure among sites, and to quantify this pattern as a "biotic score." The ordination-derived "biotic score" was then examined in relation to an aggregated urban indicator and to potential chemical stressors, and used as a dependent variable in multiple regression landscape models. The "biotic score" for each site was created by Detrended Correspondence Analysis (DCA) ordination of macroinvertebrate relative abundances. DCA was employed because preliminary analysis by Correspondence Analysis (Hill and Gauch 1980) indicated that detrending was necessary to avoid the arch effect (Gauch Jr 1982). Biotic score represents the similarity of each site's macroinvertebrate assemblage to the other sites, and can be interpreted as an underlying ecological gradient. This approach is often used in examining relations between biological assemblage data and environmental factors such as urbanization (e.g., Roy et al. 2003; Kennen et al. 2009; Coles et al. 2010). We also calculated selected metrics that are broadly used for monitoring and stream assessment, and have been demonstrated in other studies to be sensitive indicators of urbanization in the northeastern United States (e.g., Kennen et al. 2009; Coles et al. 2010). DCA was run using Canonical Community Ordination (CANOCO) software (ter Braak and Šmilaeur 1998) on taxa relative abundances after square-root transformation, and with down-weighting of rare taxa so that they have less influence on the resulting ordination but are not eliminated entirely from the analysis (ter Braak and Šmilaeur 1998). Invertebrate Data Analysis Software (Cuffney 2003) was used to resolve taxonomic ambiguities prior to DCA, and to calculate selected biotic metrics.

Selecting chemical variables

Chemical variables were selected from many available (Fischer et al. 2004) through nonparametric correlation analysis and principal components analysis (PCA). PCA was conducted on rank-transformed data, grouped by season. A Pesticide Toxicity Index (PTI; Munn and Gilliom 2001), was also calculated for each site; the PTI provides a ranking of sites according to the potential toxicity (to macroinvertebrates, in this case) of the mixture of pesticides detected in their water samples. Spearman rank analysis was used to select chemical variables that were correlated with road density at ρ of 0.60 of greater, for analysis with biotic score. Statistical analyses were conducted using SAS 9.1 (SAS Institute Inc., Cary, NC, USA).

Reducing landscape variables

The analytical software used generates more than 100 landscape variables, from which we sought a small set of non-redundant variables in the aforementioned categories. Data reduction was achieved through (1) elimination of highly-skewed variables, (2) inspection of correlation analysis results and elimination of variables that were highly correlated ($\rho > 0.85$) with others, and (3) exploratory ordinations by PCA. The final set of variables was selected from groups of highly correlated variables on the basis of metric accuracy, interpretability, applicability for management, inclusion of different scales (catchment, riparian, and urban-class), and inclusion of each of the four types of landscape pattern metrics. The final set of variables was subjected to PCA with Varimax rotation (after variable transformation to achieve approximate normal distribution, if necessary) in order to group the remaining variables into a small number of interpretable factors that preserve the original data structure. PCA was conducted on the correlation matrix, which ensures scale-independence of the input variables (Legendre and Legendre 1998). Axes were retained on the basis of minimum eigenvalue of 1.0, inspection of scree plot, and interpretability. The landscape gradient represented by each axis was quantified as site scores on that axis.

Relating landscape variables to biological condition

Univariate regression was used to examine relations between biotic score and individual landscape variables. Two multiple linear regression (MLR) approaches were used to generate 2- and 3-variable models of biotic score. First, MLR using best-subset (R^2) selection was performed; these models were evaluated on the basis of coefficient of determination and Aikake's Information Criterion (Burnham and Anderson 2002). Variance Inflation Factors were also used to assess co-linearity. Second, a set of 2-variable MLRs were run with percent impervious cover forced as the first input variable in order to evaluate each of the other variable's contribution and direction of influence to biotic score over and above impervious surface.

Results

Biotic score from ordination of macroinvertebrate data

The primary DCA axis (Axis I, Fig. 2) provided a biotic score for use as a numerical representation of macroinvertebrate assemblage variation among sites. The DCA Axis I eigenvalue of 0.31 indicated that this

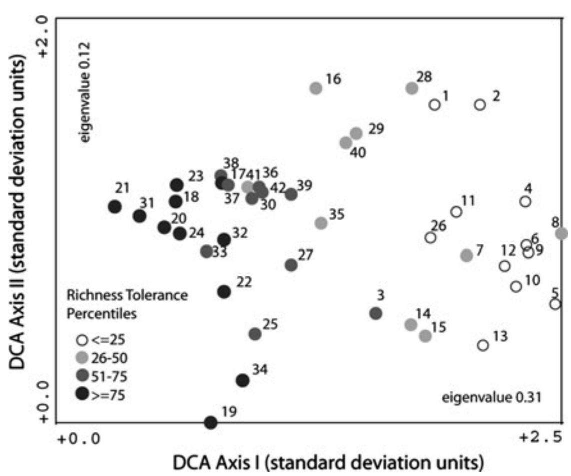

Fig. 2 DCA ordination plot from analysis of macroinvertebrate relative abundances for 42 stream sites in the DRB (eastern US) sampled during 2000–2001. Degree of *shading of symbol* represents richness tolerance, in one of four percentiles. Higher richness tolerance is an indicator of loss of sensitive taxa. Symbol numbers correspond with sites listed in Table 1

Table 2 Correlations (Spearman rank) of biotic score (DCA Axis I score) with selected macroinvertebrate indices and chemical variables

Biological or chemical indicator	Median (min–max)	Correlation with biotic score	
		ρ	P
Macroinvertebrate index or metric			
Abundance Tolerance	4.0 (2.3–5.7)	−0.91	<0.0001
Number of EPT taxa	17 (5–28)	0.84	<0.0001
Percent abundance as omnivores	5.7 (0–36)	−0.86	<0.0001
Richness Tolerance	4.2 (3.1–5.8)	−0.90	<0.0001
Total richness	41 (23–53)	0.72	<0.0001
Spring base flow chemistry			
Chloride (mg/l)	22.8 (6–76.4)	0.70	<0.0001
Dissolved nitrate + nitrite (mg/l)	1.14 (<0.05–3.636)	0.64	<0.0001
PTI (unitless)	0.0004 (0–0.0425)	0.72	<0.0001
Summer base flow chemistry			
Chloride (mg/l; $n = 41$)	20.9 (7.1–97.8)	0.60	<0.0001
PTI (unitless)	0.00003 (1–0.00975)	0.84	<0.0001

EPTr: taxa in the orders Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies)

gradient explained a high percentage of the variation relative to the secondary (eigenvalue 0.12), and subsequent axes. This biotic score was negatively correlated with road density (ρ −0.73, $P < 0.0001$), and with biological and chemical indicators of ecological condition (Table 2). Relatively higher biotic scores (i.e., DCA Axis 1 scores) indicated more diverse assemblages with more sensitive taxa, such as mayflies (Ephemeroptera), stoneflies (Plecoptera) and caddisflies (Trichoptera). In comparison, lower biotic scores indicated less diverse, more tolerant assemblages with more generalized (omnivorous) feeding strategies (Table 2). Biotic score was negatively correlated with chloride, nutrients, and PTI (Table 2). Seasonal differences were evident in the strength of these correlations. Chloride and dissolved nutrient concentrations from spring collections were more highly correlated with biotic score than those from summer collection, whereas the reverse applied to PTI.

Landscape characterization

The 12 landscape variables selected (Table 3) characterized land use composition, land cover composition, and landscape configuration variation among the study sites. Some were highly correlated ($\rho > 0.85$) with other related variables. Although not included in the final analysis data set, examples of these are identified, for reference, in Table 3.

PCA synthesized a large portion (88%) of landscape variation into three main factors (Table 4), denoted here as PCA Axes I, II, and III. PCA Axis I represented increasing edge density, decreasing forest patch size, and increasing patchiness of urban land. PCA Axis II represented increasing impervious cover and commercial/industrial land use with concomitant decreasing forest aggregation. PCA Axis III represented increasing grass cover and decreasing tree cover in the riparian zone, and increasing grass cover on urban land in the catchment.

Relation of landscape variables to biotic score

Site scores on two of the PCA axes were negatively correlated to biotic score (Fig. 3). These are Axis II, which represented increasing impervious cover, commercial land use, and forest fragmentation (ρ −0.62, $P < 0.0001$), and Axis III, which represented replacement of trees with grass in urban areas and in riparian zones (ρ −0.64, $P < 0.0001$). When combined in a multiple regression, PCA Axes II and III together accounted for 82% of the variation in biotic score (Fig. 4), indicating that the influence, on stream biota, of urban intensity in general can be significantly modified by the type of vegetation on urban land and in riparian zones.

Univariate regressions showed that biotic score was significantly correlated at ρ greater than 0.50 to five of the 12 landscape metrics (Table 4). Biotic

Table 3 Landscape land cover, land use, and pattern metrics, units, descriptions, and statistics

Abbreviation (units): description, references, and [selected highly-correlated variables]	Mean SD (min–max)
IMP_C (%): Percent of catchment that is impervious cover such as roads, rooftops, parking lots. [Catchment % forest (−); % residential, %t urban, adjacency of forest to urban patches (+)]	11.03 11.87 (1.3–44)
CI_C (%): Percent of catchment that is commercial and (or) industrial land. [Riparian % commercial/industrial (+), riparian % forest (−)]	3.22 4.17 (0–16)
UGR_C: Percent of urban land use in catchment that is grass-covered. Note: grasses in this study area are primarily turf-grass. [Catchment % grass cover (+)]	31.32 17.28 (2.1–60)
GRA_R (%): Percent of riparian zone with grass cover. [Catchment % grass cover (+)]	18.1 12.1 (0.1–47)
TRE_R (%): Percent of riparian zone with tree cover. [% urban that is tree-cover, riparian % forest, catchment % tree (+)]	73.7 16.3 (38–99)
PDN_U (number per km^2): Urban patch density; number of urban patches per unit catchment area	0.071 0.072 (0.00–0.26)
$P10_F$ (%): Percentage of forestland in catchment occurring in patches less than 4.05 ha (10 acres). [Forest patch density (+)]	1.0 1.2 (0.0–4.9)
ED_C: Edge density; meters of edge between pixels of different land use groups per unit catchment area (not including water)	0.51 0.20 (0.15–0.92)
ED_F (m/m^2): Forest edge density: meters of edge between forest land use pixels and other land use groups per unit catchment area (not including water)	0.003 0.001 (0–0.006)
AI_F (unitless): Aggregation index for forest patches in catchment. The degree of clumping or aggregation of forest patches. Ranges from 0 to 1; AI of 1.0 indicates the entire class is aggregated into a single square patch; AI closer to 0 indicates many long and skinny patches (He et al. 2000; DeZonia and Mladenoff 2004). [% of forest that is core (+) i.e., at least 30 m from patch edge]	0.93 0.04 (0.85–0.99)
AI_U (unitless): Aggregation index for urban patches in catchment; the degree of clumping or aggregation of urban patches. See definition for AI_F. [% urban area that is core (+, i.e., at least 30 m from patch edge)]	0.93 0.04 (0.854–1.00)
DOM: Dominance (unitless); the degree to which a landscape departs from maximal diversity per Shannon and Weaver (1962). Small values indicate variety of cover classes in approximately equal proportions. (Shannon and Weaver 1962; Turner 1990; DeZonia and Mladenoff 2004). [eveness (−)]	0.66 0.27 (0.001–1.14)

Selected highly-correlated variables (Spearman $\rho > 0.85$) are listed after descriptions

score was most strongly correlated with percent tree in the riparian zone. Catchment percent impervious cover accounted for 56% of the variation in biotic score. In two-variable MLR models with percent impervious forced as the first input variable (Table 4), all variables except for commercial-industrial land use contributed significantly. Reduction in biotic score associated with impervious surface was exacerbated by grass cover (on urban land and in the riparian zone), edge density (overall and of forest),

Table 4 Results of landscape variable PCA, showing input variable correlations (loadings) for Axes I–III, and results of variable regressions with biotic score PCA eigenvalues are in parentheses below axis headings

Variable abbreviation, [transformation]	Axis I (4.23)	Axis II (3.11)	Axis III (2.64)	Univariate regression with biotic score R^2 (P)	Contribution to 2-variable MLR model with percent impervious		
					Model R^2	Variable P	Direction of influence
IMP_C [Log_{10}]	−0.26	**0.90**	0.25	−0.56 (<0.0001)	–	–	–
CI_C [$\log_{10}$ X + 1]	−0.02	**0.86**	0.45	−0.69 (<0.0001)	–	ns	–
GRA_U [sq. root]	0.54	0.18	**0.71**	−0.44 (<0.0001)	0.821	<0.0001	−
GRA_R [sq. root]	0.27	0.25	**0.86**	−0.60 (<0.0001)	0.819	<0.0001	−
TRE_R	0.04	−0.49	**−0.86**	0.71 (<0.0001)	0.767	<0.0001	+
DOM	−0.38	−0.09	−0.28	ns	0.625	0.029	+
ED_C	**0.86**	0.20	0.26	−0.21 (0.009)	0.724	0.0002	−
AI_F	−0.47	**−0.75**	−0.37	0.67 (<0.0001)	0.750	<0.0001	+
ED_F	**0.94**	−0.08	−0.01	ns	0.650	0.010	−
$P10_F$	**0.72**	0.50	0.13	−0.29 (0.002)	0.651	0.009	−
AI_U	**−0.72**	0.57	−0.24	ns	0.753	<0.0001	+
PDN_U [4th root]	**0.87**	−0.21	0.18	ns	0.684	0.0019	−

Loadings of 0.60 or higher are in bold. Variable names and descriptions are provided in Table 2

forest patchiness indicators, and urban patch disaggregation. Some variables, such as urban aggregation index, which were not significantly related to biotic score on their own, contributed significantly when combined with percent impervious (Table 4).

The best MLR models from the R^2 selection approach combined impervious cover with urban grass in the catchment or grass in the riparian zone (Table 5). The three-variable model, which yielded only a small improvement in variance explained over the best 2-variable model, showed that the negative effect of impervious cover and urban grass cover on biotic score was exacerbated when the riparian zone was grass-covered instead of tree-covered.

Discussion

This study identified three main gradients of landscape change across the DRB associated with increasing urbanization—(1) edge and urban aggregation, (2) impervious land cover and forest fragmentation, and (3) vegetation cover (grass versus tree) on catchment and riparian scales. Stream biological condition (as represented by biotic score) declined with increased imperviousness and forest fragmentation, and with the replacement of tree cover with grass. Other studies have shown declines in ecological condition of streams with increasing general indicators of urbanization (e.g., Cuffney et al. 2005; Brown et al. 2009), and with forest loss, particularly in the riparian zone (e.g., Kennen 1999). Results of this study, however, suggest that other characteristics can significantly modify how biological condition responds to general indicators of urbanization. The construction of simple, 2-variable models of stream biological integrity using percent impervious as a fixed variable allowed us to investigate specifically the strength and direction of influence of various kinds of landscape changes beyond imperviousness. Our findings indicate that declines in biological condition associated with urbanization in general (and impervious surface in particular) can be exacerbated to varying degrees by the extent to which non-impervious urban land is covered in grass versus trees, the extent of grass cover in riparian zones, amount of forest edge, overall land use heterogeneity, and the proportion of forest in small patches. In contrast, greater forest and urban patch aggregation, and more tree cover in urban and riparian areas were related to better conditions than would be expected from the level of impervious surface present.

Examination of potential chemical stressors revealed that the observed variation in macroinvertebrate assemblages across the portions of the DRB

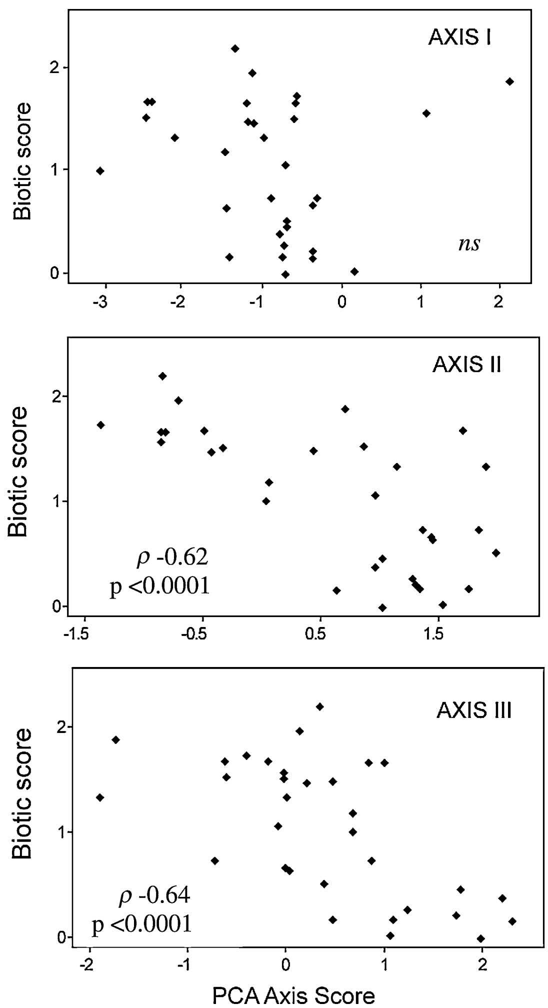

Fig. 3 Scatterplots showing relations of biotic score to landscape PCA axes. Axis I represents decreasing aggregation (forest and urban), and increasing edge, Axis II represents increasing impervious cover, commercial-industrial land use, and forest fragmentation, and Axis III represents loss of tree cover and increase in grass cover in the riparian zone and in urban areas

considered in this study were related to increases in chemical stressors that have been previously linked to urbanization both in the broader DRB (Fischer et al. 2004) and in other metropolitan areas. In our study, biological condition, as indicated by biotic score, declined with increasing potential pesticide toxicity, particularly during the summer. Brown et al. (2009) found that insecticide inputs increased with urbanization in nine other metropolitan study areas distributed across the United States, and Sprague and Nowell (2008), found increasing potential pesticide toxicity

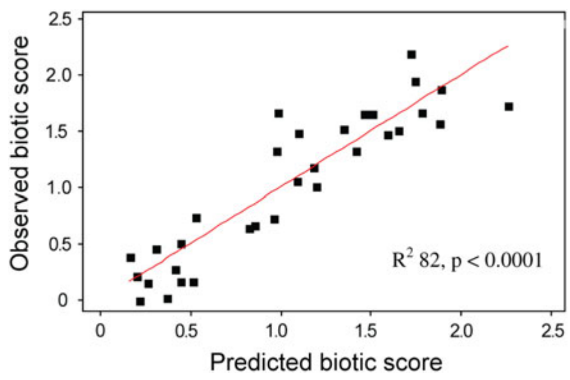

Fig. 4 Plot of observed biotic score versus biotic score predicted from multiple regression of site scores on PCA Axes II and III. Axis II represents increasing impervious cover, commercial-industrial land use, and forest fragmentation, and Axis III represents loss of tree cover and increase in grass cover in the riparian zone and in urban areas

with increasing urbanization in six of these areas. These patterns are likely a result of the typical application of insecticides in urban areas throughout the growing season (Fischer et al. 2004; Gilliom 2006). Biological condition declined with increasing chloride and nutrient concentration, particularly in the spring. Chloride is a robust indicator of urban impact (Herlihy et al. 1998; Paul and Meyer 2001; Kaushal et al. 2005), and has been linked to road de-icing salts that are transported to the stream in surface runoff in the spring (Kaushal et al. 2005; Kelly et al. 2008).

The contribution of many of these landscape characteristics can be explained by their functional relationships to stream ecosystems. Forested riparian zones provide diverse food sources and habitat structure, moderating temperature and hydrological fluctuations, and interfering with the transmission of sediment, nutrients, ions, and contaminants to the stream (Allan 1995; Sweeney and Blaine 2007). Tree cover throughout the catchment indicates less land disturbance than grass cover, or, when planted on highly disturbed land, acts as an additional physical buffer and biological filtering agent. In contrast, grass in urban areas has a negative influence on water quality relative to urban tree cover. For example, grass is often accompanied by fertilizing and pesticide application as well as irrigation, which, together, make lawns a source of nutrients and potentially-toxic chemicals to streams. This corresponds with our findings of declining biotic score with increasing PTI and nutrients.

Table 5 Best multiple regression models relating biotic score to landscape characteristics

Variable abbreviation [transformation]	Akaike's information criterion	Model R^2	Model *P*	Partial R^2	Variable *P*	Variable influence	Variance Inflation Factor
3-variable models							
IMP_C [log_{10}]	−81.6	0.86	<0.0001	0.56	<0.0001	−	1.27
UGR_C [sq. root]				0.28	0.037	−	3.51
GRA_R [sq. root]				0.02	0.042	−	3.99
2-variable models							
IMP_C [log_{10}]	−78.8	0.832	<0.0001	0.56	<0.0001	−	1.04
UGR_C [sq. root]				0.28	<0.0001	−	1.04
IMP_C [log_{10}]	−78.5	0.831	<0.0001	0.56	<0.0001	−	1.18
GRA_R [sq. root]				0.27	<0.0001	−	1.18
TRE_R	−77.6	0.826	<0.0001	0.71	<0.0001	+	1.82
AI_F				0.12	<0.0001	+	1.82
CI_C	−69.4	0.775	<0.0001	0.69	0.0067	−	2.83
TRE_R				0.08	0.0023	+	2.83
TRE_R	−68.9	0.772	<0.0001	0.71	<0.0001	+	1.15
$P10_F$				0.06	0.0086	−	1.15

Variable names and descriptions are given in Table 2. Intercepts for all models are significant at $P < 0.01$

The results of this study indicate that the continuation of sprawl-type growth in the DRB, with its corresponding effect on landscape configuration, will worsen biological condition of streams more than is predicted from models based solely on percent impervious surface. The findings suggest that some of the deleterious effects of impervious surface and other landscape changes associated with increasing urbanization are reduced by increasing aggregation of forest and (or) urban land uses (i.e., less sprawl-type development, more cluster-type development).

Our findings directly support Cushman et al.'s (2008) suggestion that it is important to distinguish class-level structure from landscape level structure, and to distinguish landscape composition from configuration, especially at the class level, when examining how landscape characteristics are related to ecological systems. Relations between biological condition and landscape characteristics were also significantly clarified in this study by using detailed and locally-accurate land use and land cover data. Separate collection of land use and land cover data allowed us to examine the importance of land cover independently of land use, and to examine the effects of tree, grass, and impervious surface cover that occur to varying extents within urban-developed land uses. Locally-accurate data also enabled reasonably effective characterization of riparian areas at the scale of a 60 m-wide zone, allowing us to analyze its role as distinct from catchment wide effects. Future efforts to monitor landscape change in urbanizing areas with respect to water quality should consider both catchment and riparian scales, include both landscape composition and pattern, and include class-specific land cover and configuration metrics.

Many of the landscape pattern characteristics identified in our study as ecologically-relevant were also identified by Cushman et al. (2008) to be both highly universal (globally present) and consistent across a broad range of landscapes. Highly consistent characteristics are relatively stable in their meaning, allowing for more precise ecological interpretation. Landscape characteristics with high consistency also provide more confidence that they can be reliably represented by any one or a subset of the individual metrics with which it is associated. Metrics that are ecologically-relevant, universal and consistent have a much greater likelihood of providing effective information for monitoring, management, and further modeling research.

Important landscape characteristics to consider in monitoring and research studies, in addition to impervious surface, include type of vegetation cover (trees versus grass) in riparian and urban areas, forest and urban patch aggregation, and overall forest cover and landscape heterogeneity in the catchment. Although we necessarily used specific metrics to represent these landscape characteristics in this study, we selected them from a number of highly-correlated and sometimes functionally-related metrics. Other metrics from each group could prove equally or more useful in other areas, with other sites and ranges of conditions, and (or) with other data sources, and should be considered when investigating potential causal relationships. Studies over large areas such as this are needed to identify and validate the broadly-applicable relationships, to enable further refinement of land management practices and protect stream health. Given the relative expense of visual photo-interpretation, broad-area studies will benefit from increased availability of high-quality satellite imagery and further assessment of the accuracy of these and other landscape metrics that are derived from these sources.

By combining environmental factors on several scales (riparian zone and catchment-wide), separating land cover from land use, and using the full gradient of landscape types, from fully forested to highly urbanized, we can better predict and monitor landscape changes that affect streams. By utilizing landscape data that more closely correspond to data used by managers and planners on the ground, and validating the landscape metrics as functional metrics that are explicitly related to stream biological condition, we can better apply the research to planning efforts and management practices. Together these results provide a much clearer understanding of those landscape characteristics that can affect streams in urban settings, and suggest potential avenues for management and planning to reduce, minimize, or correct the detrimental effects of urbanization on stream water quality.

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