J. N. Am. Benthol. Soc., 2006, 25(4):954–976

Macroinvertebrate distribution in relation to land use and water chemistry in New York City drinking-water-supply watersheds

Erika B. Kratzer[1], John K. Jackson[2], David B. Arscott[3], Anthony K. Aufdenkampe[4], Charles L. Dow[5], L. A. Kaplan[6], J. D. Newbold[7], AND Bernard W. Sweeney[6]

Stroud Water Research Center, 970 Spencer Road, Avondale, Pennsylvania 19311 USA

Abstract. Macroinvertebrate communities were examined in conjunction with landuse and water-chemistry variables at 60 sites in the NYC drinking-water-supply watersheds over a 3-y period. The watersheds are in 2 adjacent regions of New York State (east of Hudson River [EOH] and west of Hudson River [WOH]) that are geographically distinct and have unique macroinvertebrate communities. Nonforested land use at EOH sites was mostly urban (4–57%), whereas land use at sites in the rural WOH region was more agricultural (up to 26%) and forested (60–97%). Land use accounted for 47% of among-site variability in macroinvertebrate communities in the EOH region and was largely independent of geological effects. Land use accounted for 40% of among-site variability in macroinvertebrate communities in the WOH region but was correlated with underlying geology. Comparisons among 3 landuse scales emphasized the importance of watershed- and riparian-scale land use to macroinvertebrate communities in both regions. Multivariate and bivariate taxa–environment relationships in the EOH and WOH regions identified specific landuse and water-chemistry gradients and, in general, showed a continuum in conditions across the watersheds. WOH macroinvertebrate communities varied primarily with specific conductance, population density, and agricultural and urban land use, but communities were not classified as impaired along these gradients. EOH macroinvertebrate communities were associated with a wider range of watershed conditions than WOH communities. Conditions ranged from forested to urban, and distinctive communities were associated with point-source discharges, road density, and lake outlets. The severity of the impact gradient in the EOH region resulted in impaired macroinvertebrate communities with decreased total and Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxon richness and increased densities of oligochaetes and chironomids.

Key words: geology, land cover, land use, macroinvertebrates, multivariate statistics, water chemistry.

Maintaining adequate quantities of safe drinking water for a growing human population is a primary concern in the 21st century as human activities on the landscape alter critical ecosystem functions and services of streams and rivers (Allan and Johnson 1997, Paul and Meyer 2001), including those that contribute to potable drinking water (Dudley and Stolton 2003, Fitzhugh and Richter 2004). Landscape alterations can disrupt natural disturbance and flow regimes (Poff et al. 1997, Paul and Meyer 2001); modify channel morphology, substrate (Poff et al. 1997), and temperature regimes (Sweeney 1993); alter delivery and addition of nutrients, sediments, toxins, and other pollutants (Phillips et al. 2002, Meador and Goldstein 2003, Dodds and Whiles 2004); and affect the basal energy source in headwater streams (Wallace et al. 1999, England and Rosemond 2004). Identifying natural and anthropogenic environmental factors that influence biological communities in streams is an important step in effective watershed management (Wang and Kanehl 2003). Bioassessment programs, which typically have contributed to resolving issues

[1] Present address: Department of Biological Sciences, Virginia Polytechnic Institute and State University, Blacksburg, Virginia 24061 USA. E-mail: ekratzer@vt.edu

[2] E-mail address: jkjackson@stroudcenter.org

[3] Present address: National Institute of Water and Atmospheric Research, P.O. Box 8602, Christchurch, New Zealand. E-mail: d.arscott@niwa.co.nz

[4] E-mail addresses: aufdenkampe@stroudcenter.org

[5] cdow@stroudcenter.org

[6] lakaplan@stroudcenter.org

[7] newbold@stroudcenter.org

[8] sweeney@stroudcenter.org

for natural resources management, also can be relevant to management decisions for drinking-water-supply watersheds.

Macroinvertebrates are widely used as biological indicators of water quality (Rosenberg and Resh 1993). The distribution and abundance of lotic macroinvertebrates are controlled by interacting physical, chemical, and biological factors at hierarchical scales (Richards et al. 1996, Poff 1997), such that biogeographic factors at larger spatial scales influence land use/cover (hereafter land use) that, in turn, affects water chemistry and physical habitat for biota at smaller scales (Frissell et al. 1986, Poff 1997, Malmqvist 2002, Townsend et al. 2003). Biota can be influenced ultimately by factors at one or multiple scales. Indicator criteria (bioassessment metrics) aim to minimize response sensitivity at some scales (e.g., ecoregion), while maximizing sensitivity at others (e.g., local habitat) (Barbour et al. 1996, 1999, Weigel 2003).

In our study, multivariate techniques were used to relate macroinvertebrate communities to landuse and water-chemistry patterns among watersheds that are the source of drinking water for >9 million people in the New York City (NYC) metropolitan area and that are used recreationally and personally by thousands living in these watersheds. Macroinvertebrate variation was examined relative to geology, land use, and water chemistry and both broad-scale and regional patterns were identified as part of a large-scale enhanced water-quality monitoring project (the Project; Blaine et al. 2006). Understanding these relationships can contribute directly to watershed management decisions aimed at minimizing stream degradation resulting from human activities.

Methods

Study sites

Macroinvertebrates were collected from 60 sites distributed throughout the NYC drinking-water-supply watersheds (figs 1 and 2 and table 1 in Arscott et al. 2006a). Thirty sites were west of Hudson River (WOH; total watershed area = 4095 km^2) in the Catskill Mountains and Delaware River headwaters ~130 to 200 km northwest of NYC, and 30 sites were east of Hudson River (EOH; total watershed area = 971 km^2) in the Croton/Kensico watersheds ~70 to 105 km north of NYC. WOH sites were in watersheds that are part of either the Delaware River watershed (i.e., the East and West branches of the Delaware and Neversink rivers) or Hudson River watershed (i.e., Rondout Creek, Esopus Creek, and Schoharie Creek). EOH sites were primarily part of the Croton River watershed (which drains to the Hudson River), with 2 sites in the headwaters of the Bronx River (Kensico Reservoir watershed).

The WOH region is in the Northern Appalachian Plateau and Uplands and the North Central Appalachians ecoregions, and the EOH region is in the Northeastern Highlands and Northeastern Coastal Zone ecoregions (Omernik 1987). The WOH region remains largely undeveloped, with watershed forest cover ranging from 60 to 97% at WOH sites. Watershed agricultural land use ranged from 0 to 26% and urban land use ranged from 1 to 11% at WOH sites (fig. 6 in Arscott et al. 2006a). EOH sites were spread across a region with greater urban development (range = 4–57%) and more limited agriculture than the WOH (fig. 7 in Arscott et al. 2006a). Aqueducts and tunnels connect watersheds to move water to NYC, but only 1 WOH site (site 26 on Esopus Creek) and 2 EOH sites (sites 41 and 45 on the West Branch Croton River) were clearly influenced by water transfers from other watersheds (Dow et al. 2006, Kaplan et al. 2006). Detailed site descriptions can be found in Arscott et al. (2006a).

Data collection

Macroinvertebrates.—Sampling occurred once annually in either April or May in 2000, 2001, and 2002. Benthic macroinvertebrates were collected in riffles with a Surber sampler (0.093 m^2, 250-μm mesh) using a quantitative composite sampling regime. Sixteen random samples were collected at each site. A composite macroinvertebrate sample was created in the field by combining 4 random Surber samples in a large bucket and randomly removing ¼ of the material using a quadrate splitting tool. Composite samples were fixed with 5% buffered formalin. Thus, 16 Surber samples covering ~1.5 m^2 of stream bottom were reduced to 4 composite samples that were brought back to the laboratory for processing.

In the laboratory, each composite sample was subsampled to reduce the number of macroinvertebrates examined to 200 to 300 individuals per sample (~800–1200 individuals per site per year). Insects, including the Chironomidae, were identified to the lowest possible taxonomic unit (usually genus or species). Noninsect macroinvertebrates (e.g., oligochaetes, mollusks, nematodes) were identified to higher taxonomic levels (i.e., class or order).

Geology and landuse characteristics.—Thirty-three bedrock and surficial geology variables were summarized across the 60 sites from the 1:250,000 Bedrock Geology or Surficial Geology Maps of New York State (NYS Geological Survey Map and Chart Series Number 15 and Number 40; Dow et al. 2006). Landuse

TABLE 1. Landuse categories and abbreviations. X indicates scale(s) at which landuse variables were included in statistical analyses in our study. A total of 16 variables were measured at >1 scale and 7 variables were measured at the watershed scale only. Fifty-three variables were used in the east of Hudson River (EOH) analyses, and 48 were used in the west of Hudson River (WOH) analyses. See text for explanation of scales.

		Scale		
Landuse variable	Abbreviation	Watershed	Riparian	Reach
% brush	BRSH	X	X	X
% commercial	COMM	X	X	X
% coniferous forest	CONF	X	X	X
% cropland	CROP	X	X	X
% deciduous forest	DECD	X	X	X
% farmstead[a]	FMST	X	X	X
% grassland	GRAS	X	X	X
% industry	INDU	X	X	X
% mixed brush-grassland	MBRH	X	X	X
% mixed forest	MFOR	X	X	X
% orchard	ORCH	X	X	
% other urban	OURB	X	X	X
% residential	RESD	X	X	X
% transportation[b]	TRAN	X	X	X
% water	WTER	X		
% wetland	WETL	X	X	X
Watershed area	WTSD	X		
Road density	RDNS	X	X	X
Point-source discharge	SPDE	X		
Population density	PDNS	X		
Area of first upstream lake[b]	LUPS	X		
Upstream lake density[b]	LDNS	X		
Lake code[b]	LCOD	X		

[a] Not included at the reach scale in EOH analyses
[b] Not included at any scale in WOH analyses

categories for each site were extracted from an existing NYC Department of Environmental Protection (NYC DEP) geographic data set based on 2001 Landsat Enhanced Thematic Mapper Plus satellite imagery, with other data sources incorporated, to obtain a composite landuse data layer at a 10-m resolution. These data resolved 17 different landuse categories equivalent to Anderson Level 2 categories (Anderson et al. 1976). Land use upstream of each site was expressed at 3 spatial scales (Arscott et al. 2006a): 1) watershed, 2) riparian (30 m on each side of entire stream network upstream of a site), and 3) reach (same as riparian, but truncated 1 km upstream of the study site). Five variables that were linked only to the watershed scale also were computed: watershed area (km^2), State Pollution Discharge Elimination System (SPDE) permitted discharges (point-source discharge, normalized by watershed area; Arscott et al. 2006a), area of nearest upstream lake, lake density in the watershed, and lake code (a fuzzy-coded variable defined as 0 if no lake/reservoir was present along the mainstem river, 1 if there was a small [<5 ha] lake/reservoir on the mainstem >3 km upstream of the sampling site, and 2 if a large [>5 ha] lake/reservoir was within 3 km of the sampling site). The lake variables were included only in EOH analyses because lakes/reservoirs did not occur upstream of WOH sites. Overall, 53 measured landscape variables were available for inclusion in EOH analyses and 48 were available for WOH analyses (Table 1).

Water-chemistry and other instream variables.—Water depth and current velocity were measured with each macroinvertebrate Surber sample taken. Periphyton chlorophyll *a* and biomass were estimated from rocks collected in association with each composite sample. For chlorophyll *a* analyses, the chlorophyll present on whole rocks was extracted overnight in alkaline acetone, analyzed spectrophotometrically, and expressed as mg chlorophyll a/m^2 (Lorenzen 1967). The rocks were subsequently scrubbed to remove attached organic material (i.e., algae, fungi, and bacteria), and this organic material was captured on a pre-ashed GF/F filter and combusted at 550°C for 5 h. Organic matter was quantified as g ash-free dry mass (AFDM)/m^2. Rock surface area (m^2) was calculated from rock length–width measurements. Benthic organic matter (BOM) was estimated as g AFDM/m^2 of material associated with each processed subsample (after macroinvertebrates were removed). Wet detritus (organic and inorganic material) was transferred to an aluminum weigh boat, dried at 60°C for >48 h, weighed, combusted at 550°C for 5 h, and then reweighed.

Water-chemistry data for each site consisted of summer baseflow measurements of major nutrients (i.e., N, P), ions (all field and analytical procedures are in Dow et al. 2006), molecular tracers associated with anthropogenic sources of pollution (Aufdenkampe et al. 2006), and organic matter (dissolved organic C [DOC] and suspended material [seston]; Kaplan et al. 2006). Statistical analyses were done using a subset of chemical variables chosen from our larger suite of 52 baseflow chemistry analytes (Aufdenkampe et al. 2006, Dow et al. 2006, Kaplan et al. 2006) and resulted in 14 summary variables (e.g., specific conductance was a summary variable for the 6 ion analytes, the sum of all fecal steroids was a summary variable for the 7 other fecal steroid compounds analyzed). The water-chemistry data set was merged with the 4 instream measures (depth, velocity, chlorophyll *a*, and BOM) to create the chemistry data set used in subsequent analyses (Table 2). Percentage data were arcsin $\sqrt{(x)}$ transformed to remove bimodality, and concentrations were $\log_{10}(x + 1)$ transformed to down-weight high values.

Data analyses

Macroinvertebrate communities.—Measures of community structure measures included total taxon richness, Ephemeroptera, Plecoptera, and Trichoptera (EPT) richness, the Hilsenhoff Biotic Index (HBI) (Hilsenhoff 1988), and Percent Model Affinity (PMA; Novak and Bode 1992) and were calculated for each sample from computer-generated random sampling (1000 iterations) of 100 organisms for each site. These 4 metrics were combined to generate a multimetric index called the Water Quality Score (WQS), a tool developed by the New York State Department of Environmental Conservation (NYS DEC) as part of the NYS biological monitoring program (Bode et al. 2002). The 100 organism resampling program was implemented to best mimic data produced by NYS DEC methods, which call for sorting and identifying 100 organisms. Analysis of variance (ANOVA) was used to test for interregional differences in total density (ind./m^2), total richness, and WQS between WOH and EOH regions.

All statistical analyses focused on spatial relationships, rather than temporal variation, and were conducted with 3-y means. Densities were $\log_{10}(x + 1)$ transformed prior to statistical analyses. Rare taxa were removed from the data set prior to multivariate analyses to focus on common taxa that would be considered characteristic of a site (>100 taxa could be considered relatively rare in that they were collected at only 1 site or on 1 occasion). Taxa present at <5 of 60 sites were removed from interregional analyses, and taxa present at <5 of 30 sites within each region were removed from intraregional analyses (see below).

TABLE 2. Water-chemistry and instream variables with abbreviations.

Variable	Abbreviation
Alkalinity	ALK
Benthic organic matter	BOM
Chlorophyll *a*	Chl *a*
Specific conductance	COND
Depth	DEPT
Dissolved organic C	DOC
Fragrance materials	FM
Fecal sterols	FS
NH_4^+	NH_4-N
NO_3^-	NO_3-N
Polycyclic aromatic hydrocarbon	PAH
pH	pH
Particulate N	PN
Particulate P	PP
Total dissolved P	TDP
Total suspended solids	TSS
Water velocity	v_w
Volatile suspended solids	VSS

Interregional multivariate analyses.—The first step in a series of analyses of environmental gradients across the NYC drinking-water watersheds was to examine the geologic template and broad landuse patterns across all 60 sites. We used Canonical Correspondence Analysis (CCA; CANOCO, version 4.0, Microcomputer Power, Ithaca, New York), a direct gradient ordination technique that uses regression procedures to relate species and environmental data and assumes that species have unimodal (or Gaussian) responses to environmental variables (ter Braak 1986, 1995). In this analysis, a subset of geology variables was selected from Dow et al. (2006) to best describe surficial and bedrock geology differences between EOH and WOH regions. These variables covary with many other natural differences between these regions (e.g., watershed elevation and slope, climate variables, soil characteristics, etc.), and it was our intent to use both surficial and bedrock geology as descriptors of geologic/physiographic differences and as proxies for the other gross natural differences between the regions. In addition, 6 landuse variables were selected from 20 watershed-scale (watershed-scale variables listed in Table 1 minus the 3 lake variables) landuse variables using the automatic forward selection procedure included in the CANOCO statistical package. This analysis clearly separated EOH from WOH sites based on distinctly different macroinvertebrate communities. Therefore, subsequent analyses were conducted separately for each region to avoid confounded and complicated interpretation of patterns within each region.

Within-region multivariate analyses.—Intraregional analyses involved 3 steps. First, the large number of variables in the landuse and chemistry data sets was reduced to the variables that best explained and represented macroinvertebrate variation. Second, variance-partitioning analysis was used to account for variation in macroinvertebrates explained by land use after removing effects of geology because land use and geology are often not independent. Third, a multivariate analysis was used to describe patterns and relationships between macroinvertebrate communities and the environment (land use and water chemistry).

Separate CCA analyses (for each region and data set) using the manual forward selection procedure in CANOCO were run to reduce the landuse data set (48 WOH and 53 EOH variables; Table 1) and the water-chemistry data set (18 variables in each region; Table 2) to those variables that significantly explained spatial variation in macroinvertebrate communities. These parallel analyses also allowed comparison of the ability of each set of variables to explain variation in macroinvertebrate communities. Forward selection effectively reduces the number of predictor variables, but it does not necessarily eliminate colinear variables from the final model. Colinear variables may be added to the final model if they explain additional residual variation and, thus, add information to the model not described by variables selected earlier in the procedure. The forward selection procedure uses a partial Monte Carlo permutation test to assess the usefulness of each potential predictor variable for extending the subset of explanatory variables used in the ordination. Variables that were significant at $p < 0.05$ in Monte Carlo tests were retained in these landuse CCA models and were incorporated into multivariate analysis of taxa–environment relationships (see below).

Geology and land use often are not independent, so variance partitioning CCA was used to aid in interpretation of intraregional landuse patterns. This CCA partitioned macroinvertebrate variation into variation that was: 1) explained solely by land use, 2) explained solely by geology, and 3) explained by the inseparable interaction of geology and land use (Borcard et al. 1992, Richards et al. 1997, Lepš and Šmilauer 2003). Five variables from each data set (geology and land use) were included in the variance partitioning CCA. Geology variables in this analysis were selected from surficial and bedrock geology data sets (Dow et al. 2006) to describe geologic gradients within each region. These variables were selected by inspecting results from a PCA (not shown) done on the suite of surficial and bedrock variables for each region (14 variables for WOH and 23 variables for EOH). Landuse variables were selected from the forward selection CCA above (i.e., the first 5 landuse variables in the model). Equal numbers of independent variables from each data set are important when comparing the amounts of variation explained because each explanatory variable is likely to increase the amount of explained variation by chance alone (Borcard et al. 1992).

Site–site and taxa–environment relationships were assessed within each region using Co-Inertia Analysis (CIA), an unconstrained multivariate technique for relating 2 kinds of data sets (Dolédec and Chessel 1994). Interpretations of site–environment–taxa relationships were based on an unconstrained method rather than a constrained method (such as CCA) to view all macroinvertebrate variance and identify the strongest environmental gradients in the data set. Environmental variables selected in the separate landuse and water-chemistry CCAs were included together in CIA, which first computes separate ordinations of each table. Environmental data (water chemistry and land use) were examined using Principal Components Analysis (PCA), and taxa were examined using row-weighted Correspondence Analysis (CA; row weight = 1). The final step in CIA maximizes covariance between the tables and projects this variance–covariance matrix in n dimensions (only the first 2 factors were projected in our study). CIA output illustrates the co-structure of each table resulting in an ordination of environmental vectors, taxa distributions, and sites ranked by both taxa and environment scores. The proximity of a site when scored by taxa to the site when scored by its environmental variables describes the concordance of the biological (macroinvertebrate) community with its environment. Statistical significance of the co-structure between the species and environment matrices was assessed by a Monte Carlo random permutation test with 1000 random matches of the 2 tables. CIA was done with the ADE-4 software (Thioulouse et al. 1997).

Pearson product–moment correlations were computed between all variables in the greater landuse and water-chemistry data sets to aid in interpretation of environmental gradients (SAS version 9.1, SAS Institute, Cary, North Carolina). In each regional analysis, simple correlation was used to illustrate relationships between the best explanatory variables and macroinvertebrate CIA Factor 1 (F1) scores. Correlations between macroinvertebrate density, total taxon richness, and WQS and macroinvertebrate CIA F1 scores were used to determine if the metrics were related to the separation of sites in the ordination.

Results

Macroinvertebrate communities in the WOH and EOH regions

A total of 543 taxa (464 in the WOH region and 436 in the EOH region) were identified across the 60 sites during the 3-y period. All major aquatic macroinvertebrate groups were represented among the 543 taxa, and oligochaetes and dipterans (primarily Chironomidae) were the most abundant taxa in both regions. The total number of taxa included individuals that could not be identified to species (i.e., taxa identified at a higher classification such as family or genus). Species totals included 43 Ephemeroptera, 19 Plecoptera, 55 Trichoptera, and 117 Chironomidae. After elimination of rare taxa, the final data matrices for multivariate analyses included 261 WOH taxa and 224 EOH taxa.

Mean taxon richness ($p < 0.001$) and WQS ($p < 0.001$) were significantly higher at WOH sites than EOH sites, whereas macroinvertebrate densities were significantly higher at EOH sites ($p < 0.05$). Mean macroinvertebrate density per site in each region ranged from 6825 ind./m^2 (site 16) to 38,896 ind./m^2 (site 15) in the WOH region and 10,848 ind./m^2 (site 54) to 164,319 ind./m^2 (site 49) in the EOH region. Mean taxon richness per 100 individuals ranged from 10 (site 49) to 32 (site 36) taxa in the EOH region and 28 (site 17) to 37 (sites 3 and 10) taxa in the WOH region. WQS highlighted a wider range of conditions in the EOH than in the WOH region. The 3-y mean WQSs for WOH sites ranged from slight impact (6.5 at site 26) to no impact (9.0 at site 9) (Fig. 1A); WQSs for EOH sites ranged from severe impact (1.8 at site 49) to no impact (8.2 at site 34) (Fig. 1B).

Interregional analysis.—CCA of all 60 sites resulted in clear separation of EOH and WOH sites and highlighted regional differences in macroinvertebrate communities (Fig. 2). Landuse and geology variables explained 40% of macroinvertebrate species variation. This regional separation was also evident when macroinvertebrates were examined using genus- and family-level density, relative abundance (%), and presence/absence data matrices (data not shown). The EOH and WOH regions were separated along a trajectory of differing geology (e.g., kame deposits and Lower Walton formation in the WOH region and biotite gneiss and Fordham formation in the EOH region) and a gradient in population density and % water land use (both were higher in the EOH than the WOH region). Sites in both regions formed a landuse gradient (orthogonal to geology) from sites influenced by point-source discharges and % commercial land use to sites with greater % deciduous forest in the EOH, and from sites with agricultural land use (% farmstead) to forested sites in the WOH. EOH sites spanned a longer gradient than WOH sites, suggesting that macroinvertebrate communities were more dissimilar among EOH sites than among WOH sites.

Intraregional selection of landuse and chemistry variables.—CCA models of WOH sites selected 7 landuse variables and 6 chemistry variables based on Monte Carlo significance (Table 3). Percent of total inertia explained was similar between the 2 models. Population density explained 12% of the total inertia in the WOH landuse model, and specific conductance explained 14% in the chemistry model. CCA models of EOH sites selected 8 landuse and 6 water-chemistry variables based on Monte Carlo significance (Table 4). The 2 models explained similar amounts of variation when equal numbers of explanatory variables were considered (i.e., the first 6 landuse variables explained 40% of total inertia and the 6 chemistry variables explained 37% of total inertia). Road density explained 11% of total inertia in the landuse analysis, and NH_4-N explained 15% in the chemistry model. No reach-scale landuse variables were selected in either EOH or WOH landuse models.

Variance partitioning of geology and land use.—The influence of land use on macroinvertebrate variation was not completely separate from spatial geological effects in the WOH region. Approximately 40% of the macroinvertebrate variation explained by land use also could be explained by geology (Fig. 3). Geology and land use independently explained similar amounts of macroinvertebrate variance (17 and 18% for geology and land use, respectively; Fig. 3). Thirteen percent of macroinvertebrate variation was explained by redundant land use and geology information; therefore, the total amount of variation explained by land use was 30% and by geology was 31%. The influence of land use and geology were more independent in the EOH region. Less than ¼ of the variance explained by land use also could be explained by geology. Furthermore, land use (26%) independently explained more variation than geology (15%) in the variance partitioning analysis (Fig. 3), and the % of redundant variation explained by both land use and geology was relatively low (8%).

CIA and macroinvertebrate–environment relationships

WOH.—Co-structure between the environment and taxa was highly significant (Monte Carlo test: $p <$

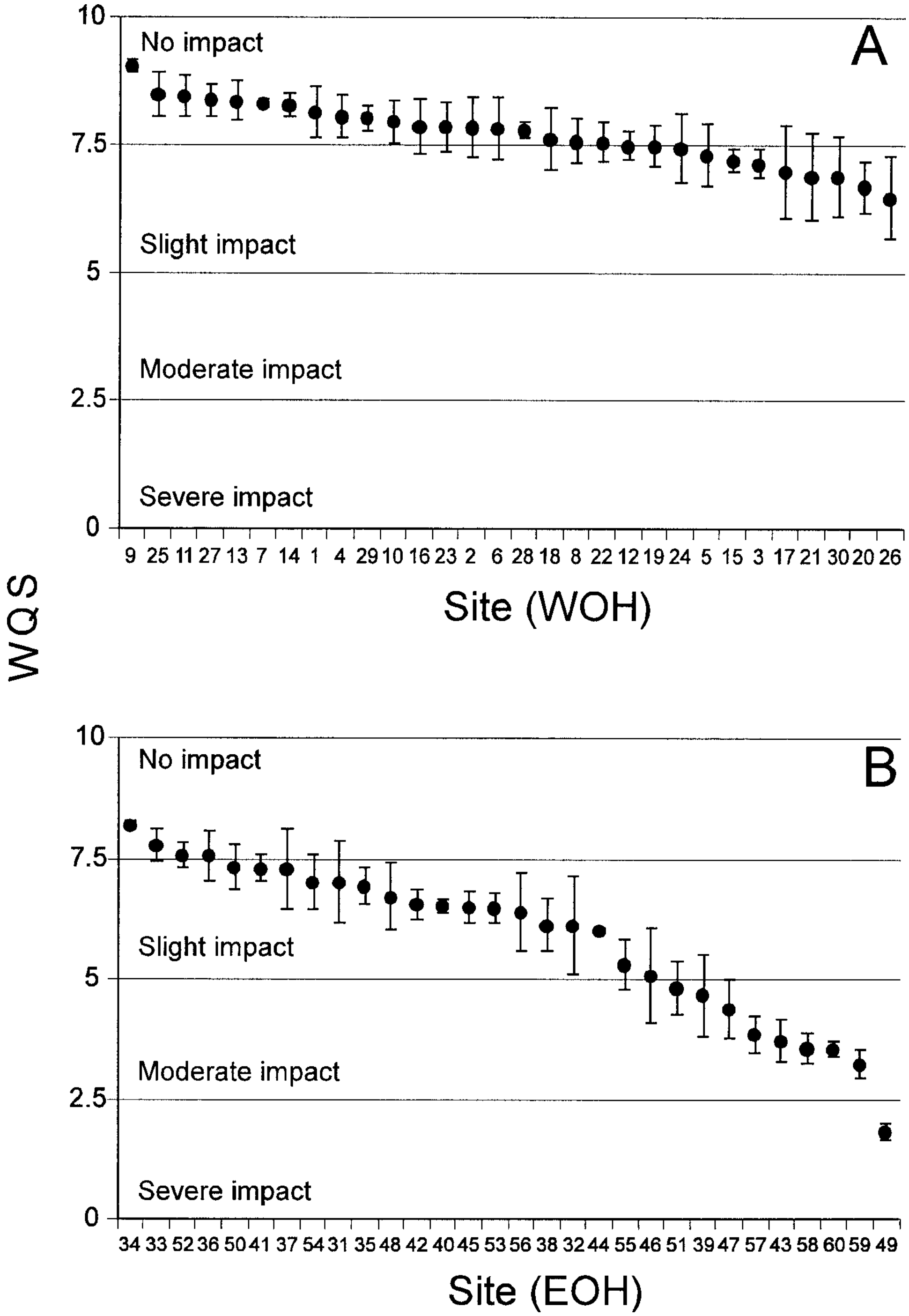

FIG. 1. Three-year mean (±1 SE; $n = 3$) macroinvertebrate water quality scores (WQS) for west of Hudson River (WOH: A) and east of Hudson River (EOH: B) sites. Horizontal lines indicate the cut-off values for 4 water-quality categories: no impact, slight impact, moderate impact, and severe impact.

0.001) with high correlations between the 1st and 2nd factors (F1 and F2) of both ordinations (r = 0.98 and 0.96; Fig. 4A). F1 and F2 of the CIA explained 65% of the total variation in environment variables and 24% of the spatial variation in macroinvertebrate communities.

Specific conductance and population density explained the greatest variation on F1 followed by alkalinity, % grassland, total dissolved P, and riparian-scale % residential land use (Fig. 4B, Fig. 5A, B). Urban and agricultural land uses, although different in their respective ranges of land use, had similar

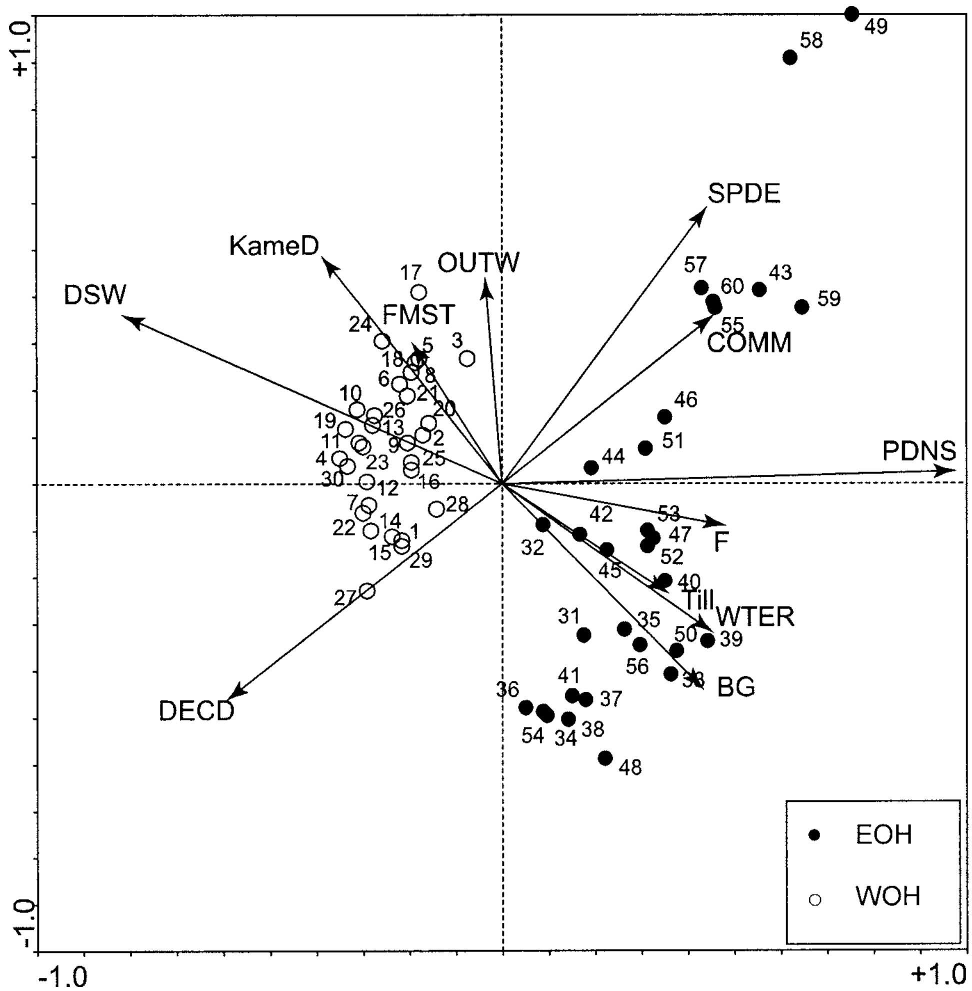

FIG. 2. Canonical Correspondence Analysis (CCA) of 3-y mean macroinvertebrate densities constrained by selected surficial/bedrock geology and landuse variables at all east of Hudson River (EOH, $n = 30$; closed circles) and west of Hudson River (WOH, $n = 30$; open circles) sites. See Tables 1 and 2 and table 1 of Dow et al. (2006) for variable names and abbreviations.

relationships with macroinvertebrate F1 (both at $r = 0.69$; Fig. 5C, D). However, WQS was not significantly related to specific conductance, population density, or urban and agriculture land uses (Fig. 5A–D). F2 was primarily related to watershed area, wetland land cover, and total suspended solids (Fig. 4B). Variables included in the CIA were significantly correlated with other water-chemistry, nutrient, and landuse variables (Table 5), helping to further define these gradients.

Sites within the same watershed tended to cluster together along gradients in chemistry and land use (Fig. 4A, B). For example, West Branch Delaware sites (1–9) and Neversink/Rondout sites (27–30) generally represented opposite ends of an environmental gradient indicated by F1. West Branch Delaware sites on the positive end of F1 generally had higher conductivity, population density, % grassland, % residential land use, and nutrients. They also had higher % agricultural and % urban land use than other WOH watersheds. Furthermore, West Branch Delaware sites with larger watershed areas (5, 6, and 8) were separated from sites with smaller watershed areas (1, 7, 2, and 9) on F2. Neversink/Rondout sites (27, 28, 29, and 30) on the negative end of F1 were associated with low specific conductance, population density, benthic chlorophyll *a*, and nutrients. East Branch Delaware (10–15), Schoharie (16–21), and Esopus sites (22–26) had intermediate conditions. Generally, East Branch Delaware and Schoharie sites had greater specific conductance, population density, and nutrients than Esopus sites, and East Branch Delaware sites had higher agricultural land use than Schoharie, Esopus, and Neversink/Rondout sites. Schoharie sites had higher % urban land use than Esopus and Neversink/Rondout sites.

Major taxonomic groups (Ephemeroptera, Trichop-

TABLE 3. Results of west of Hudson River (WOH) canonical correlation analysis (CCA) models for landuse and water-chemistry data sets. The 7 landuse and 6 water-chemistry variables were identified by manual forward selection and were significant based on Monte Carlo permutation tests ($p < 0.05$). Landuse variables with b- were quantified at the riparian scale (30-m buffer on each side of stream for entire stream length) and all other variables were quantified at the watershed scale. See Tables 1 and 2 for variable names and abbreviations.

	WOH CCA model	
	Land use	Chemistry
Percent of total inertia explained	40	37
Cumulative (axes 1+ 2) variance of taxonomic data	21	22
Taxa–environment correlations	0.95	0.95
Total inertia	0.649	0.649
Manual forward-selection variables	PDNS	COND
	GRAS	TDP
	RDNS	ALK
	WTSH	Chl *a*
	WETL	TSS
	b-MBRH	PN
	b-RESD	

TABLE 4. Results of east of Hudson River (EOH) canonical correlation analysis (CCA) models for landuse and water-chemistry data sets. The 8 landuse and 6 water-chemistry variables were identified by manual forward selection and were significant based on Monte Carlo permutation tests ($p < 0.05$). Landuse variables with b- were quantified at the riparian scale (30-m buffer on each side of stream for entire stream length) and all other variables were quantified at the watershed scale. See Tables 1 and 2 for variable names and abbreviations.

	EOH CCA model	
	Land use	Chemistry
Percent of total inertia explained	47	37
Cumulative (axes 1+ 2) variance of taxonomic data	24	21
Taxa–environment correlations	0.94	0.89
Total inertia	1.102	1.102
Manual forward-selection variables	RDNS	NH_4-N
	LCOD	DEPT
	b-DECD	FS
	LDNS	TDP
	TRAN	PN
	SPDE	TSS
	b-COMM	
	MBRH	

tera, Chironomidae, etc.) were spread among environmental conditions in the WOH (Fig. 4C). Chironomid taxa were distributed among all gradient endpoints. In addition to chironomids, several species of elmid beetles (e.g., *Optioservus trivittatus*, *Stenelmis mera*, *Stenelmis crenata*, and *Dubiraphia* sp.), a few Ephemeroptera and Plecoptera, and noninsect taxa (e.g., leeches and bivalves) were most abundant at sites with higher nutrient and ion concentrations (West Branch Delaware sites). Small West Branch Delaware sites were characterized by baetid ephemeropterans (e.g., *Baetis intercalcaris/flavistriga* and *Diphetor hageni*), perlid plecopterans (e.g., *Agnetina capitata* and *Paragnetina immarginata*), and a few trichopterans in the families Uenoidae, Odontoceridae, and Limnephildae. Low specific conductance and low population density sites (Neversink/Rondout) were characterized by more Trichoptera (e.g., rhyacophild, glossosomatid, and brachycentrid taxa) and Plectoptera taxa (e.g., perlodids and chloroperlids). Ephemeroptera generally did not discriminate among WOH sites as well as taxa in other groups. Numerous macroinvertebrate taxa (e.g., oligochaetes) were uniformly distributed along environmental gradients and contributed little to site separation (i.e., unlabelled points in Fig. 4C).

EOH.—Co-structure between the environment and taxa was highly significant (Monte Carlo test: $p < 0.001$) with high correlations between both factors ($r = 0.93$ and 0.95 for F1 and F2, respectively; Fig. 6A). F1 and F2 of the CIA explained 51% of the total variation in environment variables and 28% of the spatial variation in macroinvertebrate communities.

F1 represented a forested-to-urban gradient (Fig. 6B). Riparian-scale % deciduous forest increased in the negative F1 direction. NH_4-N, particulate N (PN), road density, point-source discharges, and total fecal steroids increased along the positive F1 dimension. NH_4-N, watershed-scale road density, watershed-scale % urban land use, and PN were strongly correlated with macroinvertebrate F1 and strongly negatively correlated with WQS (Fig. 7A–D). NH_4-N concentrations were more than an order of magnitude higher at site 49 than the other sites. When site 49 was removed from the correlation, the relationship of NH_4-N with F1 weakened considerably (from $r = 0.75$ to $r = 0.63$). Road density and point-source discharges were positively correlated with several variables (e.g., watershed-scale population density, Cl^-, specific conductance, total dissolved N) not included in the CIA (Table 6), and these variables further described the forested-to-urban gradient. F2 described distinctive conditions related to lake outlets (i.e., including lake density, lake code, total suspended solids, depth, and velocity) in the negative direction on F2 and watershed-scale % transportation land use in the positive direction (Fig. 6B).

Sites 43, 49, and 58 defined the urban end of the

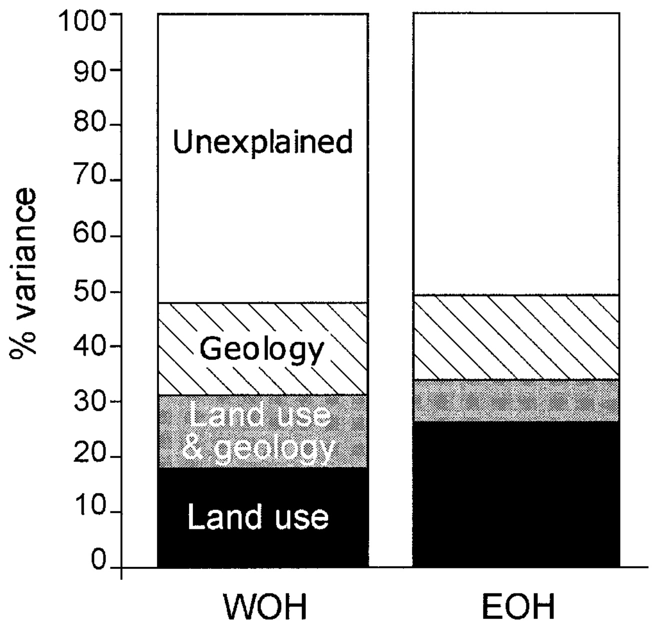

FIG. 3. Variance in macroinvertebrate communities explained by geology, land use, geology and land use, and unexplained variance from CCA analyses for the west of Hudson River (WOH) and east of Hudson River (EOH) regions. WOH geology variables were kame deposits, outwash sand and gravel, till, Oneonta formation, and Lower Walton formation, and landuse variables were population density, % grassland, road density, watershed area, and % wetland. EOH geology variables were outwash sand and gravel, till, biotite granite gneiss, Fordham formation, and Manhattan formation, and landuse variables were road density, lake code, % deciduous forest, lake density, and % transportation.

disturbance gradient (NH_4-N, watershed-scale road density, PN, point-source discharges, and watershed-scale % urban land use; Fig. 6A, B). Sites at the forested end (sites 54, 37, 50, 53, 32, 41, 48, and 34) had greater riparian-scale % deciduous forest and were less influenced by anthropogenic factors, as indicated by lower nutrient concentrations, road density, and point-source discharges. Sites 59 and 60 were very small urban streams draining watersheds with high % transportation and high % mixed brush-grassland. Sites draining larger watersheds, at the opposite end of the gradient (sites 38, 45, 41, 36 and 44), had the greatest upstream lake densities, were in closer proximity to lakes/reservoirs, had greater seston concentrations (e.g., PN), and had greater riparian-scale % commercial land use.

Chironomid taxa loaded on all gradient endpoints, but numerous chironomid taxa (e.g., *Micropsectra polita*, *Polypedilum scalaenum*, *Orthocladius dubitatus*, *Eukiefferiella coerulescens*, *Robackia* sp., and *Nanocladius crassicornis*) particularly defined the urban end of the disturbance gradient and were associated with high point-source discharges and high watershed-scale road density and % transportation (e.g., *Thienemanniella boltoni* and *Zavrelimyia* sp.; Fig. 6C). Ephemeroptera, Trichoptera, and Plecoptera (e.g., *Epeorus pleuralis*, *Polycentropus remotus*, *Acroneuria abnormis*, and peltoperlids) were more common at the forested end of the disturbance gradient (Fig. 6C). Larger sites below lake outlets were characterized by Trichoptera (Fig. 6C), including several species of hydropsychids, in addition to chironomids. As in the WOH, numerous macroinvertebrate taxa were uniformly distributed along environmental gradients and contributed little to site separation and discrimination.

CIA factors, density, richness, and WQS

In the WOH, density ($r = 0.20$, $p = 0.28$), taxon richness ($r = 0.30$, $p = 0.10$), and WQS ($r = 0.04$, $p = 0.83$) were not related to CIA F1 (Fig. 8A). In the EOH, CIA site scores based on macroinvertebrate communities (i.e., F1 from CIA) were positively related to invertebrate densities ($r = 0.75$, $p < 0.0001$) and negatively related to average taxon richness/100 ind. and WQS ($r = 0.78$ and $r = 0.94$, respectively; Fig. 8B).

Discussion

Biogeographic distribution of macroinvertebrates and regional distinction

Macroinvertebrate communities can differ greatly across large-scale regional distances as a result of natural geologic and geographic features (Feminella 2000, Li et al. 2001, Townsend et al. 2003, Weigel 2003). The ability to distinguish macroinvertebrate community variation caused by natural landscape or biogeographic differences is important for evaluating macroinvertebrate response to various stressors (Hawkins and Norris 2000). The adjacent EOH and WOH regions had differing underlying geophysical templates, and benthic macroinvertebrate communities exhibited striking differences between the 2 regions. Community differences were evident at species, genus, and family levels of taxonomic resolution and with presence/absence data (EBK, unpublished data). Such strong differences have not been shown in other studies of neighboring physiographic regions or ecoregions (Hawkins and Vinson 2000, Marchant et al. 2000, Waite et al. 2000); however, differences in physiography (e.g., montane vs valley) and the geographic distance between ecoregions (e.g., neighboring regions may be more similar) may affect whether differences are detected (Feminella 2000,

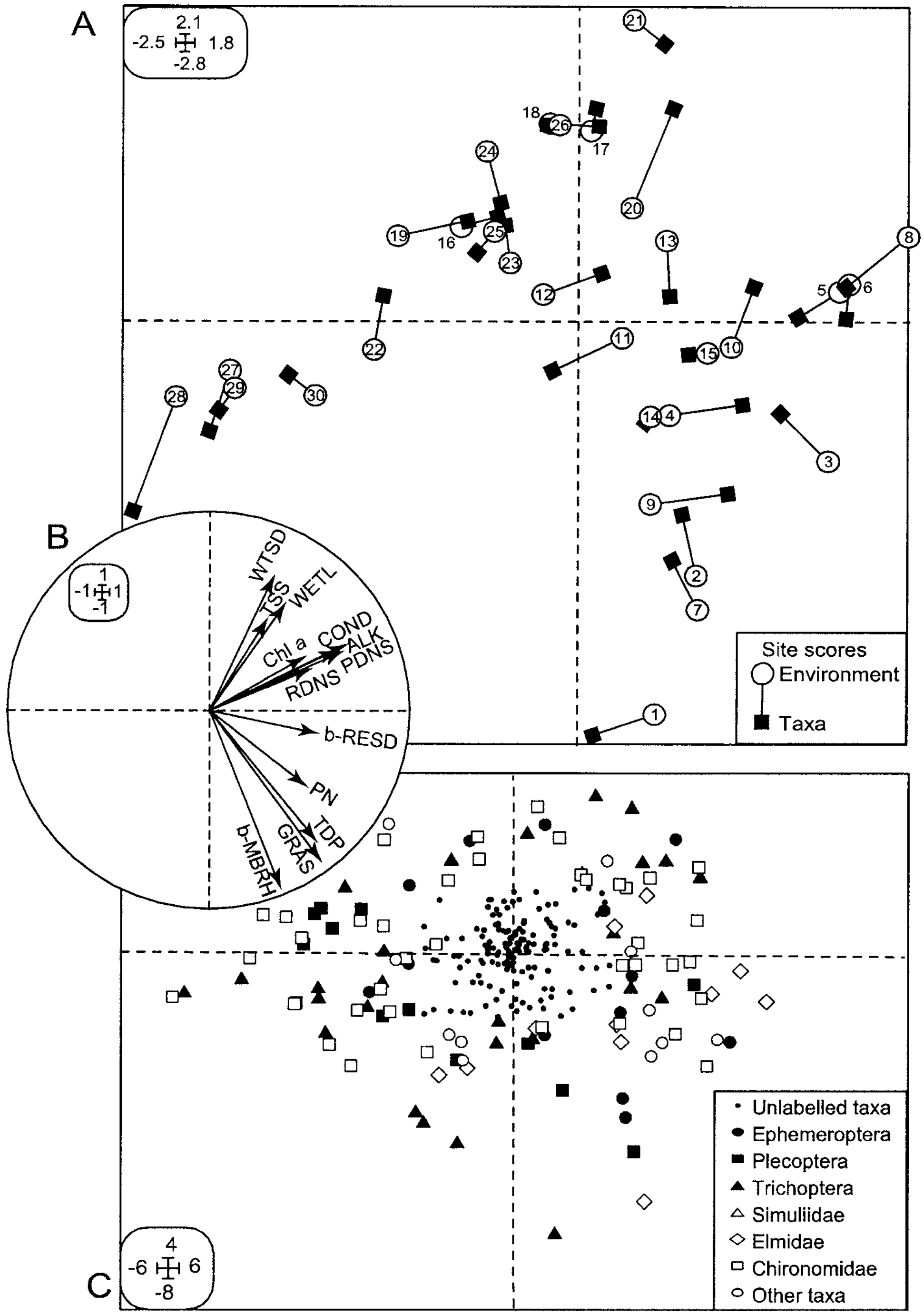

FIG. 4. Co-Inertia Analysis (CIA) of west of Hudson River (WOH) sites. Ordination diagrams are for site scores by species densities and environmental variables (A), environmental variables (B), and macroinvertebrate species densities coded by taxonomic group (C). In (A), the shorter the distance between the species and environment site scores, the better the agreement between the 2 analyses. Landuse variables with b- were quantified at the riparian scale (30-m buffer on each side of stream for entire stream length). All other landuse variables were quantified at the watershed scale. See Tables 1 and 2 for variable names and abbreviations. Insets show axis lengths.

Hawkins et al. 2000, Marchant et al. 2000, Rabeni and Doisy 2000). Furthermore, taxonomic resolution, sampling design, and choices of statistical methods can increase or decrease sensitivity in detecting biogeographic effects relative to other factors (Hawkins et al. 2000, Arscott et al. 2006b).

Geographic differences between and within the WOH and EOH regions have presumably influenced

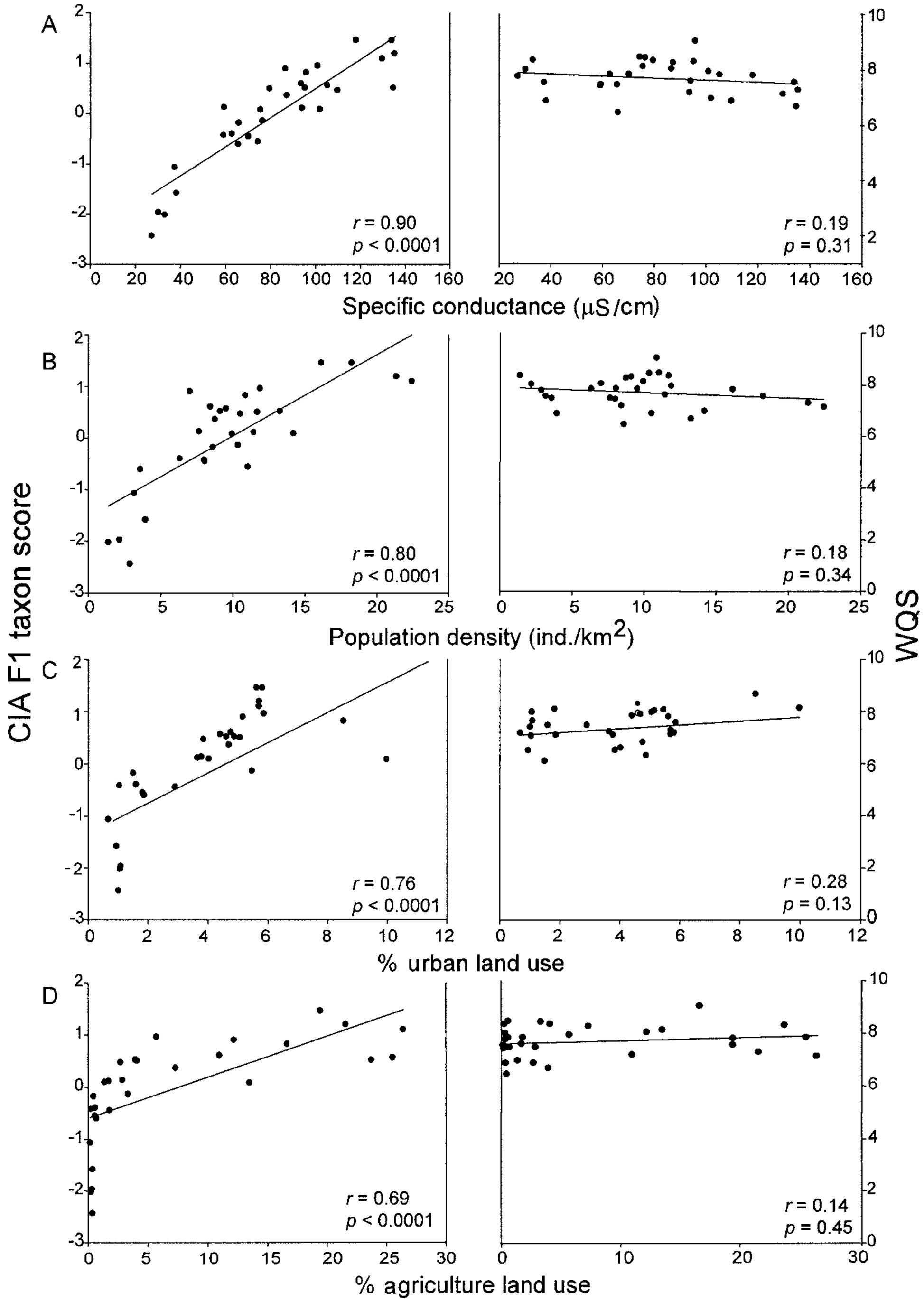

FIG. 5. Relationships between specific conductance (A), population density (B), urban land use (C; = sum of RESD, COMM, INDU, TRAN, and OURB in table 2 of Dow et al. 2006), and agriculture (D; = sum of CROP, ORCH, FMST, and GRAS in table 2 of Dow et al. 2006) for west of Hudson River (WOH) sites and either the Co-Inertia Analysis (CIA) Factor 1 site scores based on the macroinvertebrate data matrix or the macroinvertebrate water-quality score (WQS). Lines were drawn to aid in interpretation of relationships.

human activities and landuse patterns (Mehaffey et al. 2001, Arscott et al. 2006a) that have, in turn, amplified regional distinctions (sensu Allan 2004). For example, sites in the EOH are closer to NYC, and more people live in these watersheds. The degree of biological degradation at EOH sites relative to WOH sites (macroinvertebrate density, average richness per site, and WQS showed a wider range in EOH conditions) may have intensified regional macroinvertebrate differences. Even though one might assume that differ-

Table 5. Pearson product–moment correlation coefficients (significant at $p < 0.05$) between variables from the west of Hudson River (WOH) Co-Inertia Analysis (CIA) and other environmental variables not included in the CIA. Landuse variables with b- were quantified at the riparian scale (30-m buffer on each side of stream for entire stream length). All other landuse variables were quantified at the watershed scale. See Tables 1 and 2 for variable names and abbreviations.

Landuse category	CIA variable	Related variables	*r*
Urban	b-RESD	RESD	0.91
		OURB	0.70
	RDNS	b-DECD	–0.85
		PDNS	0.84
		Mg	0.83
		Na	0.83
		COND	0.82
		Cl	0.81
		K	0.81
		RESD	0.79
		OURB	0.78
		GRAS	0.77
		b-COMM	0.75
		b-OURB	0.75
		DOC	0.75
		b-GRAS	0.74
		CROP	0.74
		Ca	0.74
		BRSH	0.73
		FARM	0.73
		PP	0.73
		b-CROP	0.72
		b-FARM	0.71
		COMM	0.70
		b-WTER	0.70
		DECD	−0.70
		b-WTER	0.70
		ALK	0.70
		FS	0.70
	PDNS	COND	0.92
		Na	0.92
		Ca	0.88
		Cl	0.86
		b-COMM	0.85
		ALK	0.84
		COMM	0.83
		Mg	0.82
		RESD	0.77
		PP	0.75
		FS	0.73
		K	0.72
		INDU	0.71
		TP	0.70
		b-DECD	−0.70

Table 5. Continued.

Landuse category	CIA variable	Related variables	*r*
Agriculture	GRAS	CROP	0.94
		K	0.92
		OURB	0.91
		Mg	0.89
		FARM	0.84
		TDP	0.84
		DOC	0.80
		PP	0.79
		TDN	0.78
		RDNS	0.77
		COND	0.74
		DECD	−0.73
		Ca	0.71

ences between the 2 regions may be related to a paucity of sensitive taxa in response to development in the EOH region, a natural signal was evident at the least- and the most-impaired sites. Macroinvertebrate communities did not overlap in multivariate space among the least-impaired sites in the 2 regions; likewise, macroinvertebrate communities did not converge among the most-impaired sites in the 2 regions as a result of anthropogenic activity.

Macroinvertebrate variation in relation to WOH environmental gradients

Macroinvertebrate communities in the WOH region generally varied in relation to a landuse gradient from forested to agricultural and urban land uses and with water-chemistry gradients (nutrients and specific conductance associated with these landuse gradients; Figs 4 and 5). Bivalves, leeches, ceratopogonids, the burrowing mayfly *Ephemera*, and a number of elmid beetle taxa were more common at agricultural and urban sites than elsewhere in the WOH. Conversely, several species of perlodid and chloroperlid stoneflies and glossosomatid and brachycentrid caddisflies were among the taxa that differentiated the more forested, low-specific conductance sites in the Neversink/Rondout and Esopus watersheds. Despite these differences in macroinvertebrate distribution and abundance, the macroinvertebrate WQS showed no significant relationships with environmental gradients (e.g., specific conductance, population density, urban and agriculture land uses). The presence of particular taxa suggests some subtle effects of land use and water chemistry that parallel more intense responses observed in other studies, although the macroinvertebrate response to impairment in the WOH region was not great enough to result in moderately to severely impaired classifications. For example, Dovciak and Perry (2002) observed an abundance of elmids and the burrowing mayfly *Hexagenia* in agricultural streams with modified channels and poor riparian condition in Minnesota. Plecoptera did not dominate communities in the Neversink/Rondout and Esopus watersheds, but they are possibly indicative of more acidic conditions in the Catskills portion of the WOH (Simpson et al. 1985, Rosemond et al. 1992, Griffith et al. 1993, Earle 2004).

Macroinvertebrate communities and environmental conditions varied among the 5 WOH watersheds such that sites within the same watershed tended to have more similar communities and environmental conditions than sites from different watersheds. Watershed patterns suggest that natural spatial factors that covary with land use underlie these relationships among macroinvertebrates, land use, and water chemistry (Allan 2004, King et al. 2005) and, thus, may lead to misinterpretation of the influence of land use vs spatial effects. In the WOH region, landuse and water-chemistry gradients strongly corresponded to the spatial pattern among the 5 watersheds, and, as is often the case, a tight interaction between geology and land use confounded our ability to clearly assign spatial variance in macroinvertebrate communities to either effect. However, studies of water chemistry in the WOH region provide strong evidence that agricultural and urban land uses have influenced, independent of geology, concentrations of anthropogenically derived chemical compounds (Aufdenkampe et al. 2006) as well as nutrient and major ion concentrations (Mehaffey et al. 2001, Dow et al. 2006) and dissolved and particulate organic C concentrations (Kaplan et al. 2006).

The multimetric WQS was not related to variation in macroinvertebrate communities or changes in environmental conditions among WOH sites. We believe that changes in land use (forested to urban or agricultural) were not intensive or extensive enough to cause sufficient stress to completely remove sensitive taxa and reduce taxon richness (2 very important components of the WQS metric). Lenat and Crawford (1994) found that declines in taxon richness within intolerant groups (EPT) were partially offset by increases in tolerant groups at agricultural sites when agriculture was not especially intense. This shift may be occurring among West and East Branch Delaware sites, but it was not reflected in changes in pollution tolerance through HBI (Hilsenhoff 1988) or PMA (Novak and Bode 1992), both components of the WQS.

Evidence from other studies suggests that agriculture at higher levels (i.e., >30% land use) than in the WOH watersheds can have a measurable effect on biotic communities (Lenat and Crawford 1994, Delong and Brusven 1998). For example, Illinois headwater streams with 80 to 90% row-crop agriculture had low EPT richness and diversity and a dominance of noninsects (Stone et al. 2005). However, 80% of the actively farmed agricultural tax parcels in the West and East Branch Delaware watersheds were livestock operations, primarily dairy farms, with pastoral grazing fields (National Research Council 2000), and this type of agriculture may have less impact on biota than similar levels of row-crop agriculture (Meador and Goldstein 2003, Allan 2004).

Another consideration is the plethora of implemented farm best-management practices (BMPs) that could have reduced agricultural effects in these watersheds. More than 90% of eligible farms in the Delaware River watersheds participate in an agricultural BMP program. Farm BMPs reduced P loading to small tributaries of the West Branch Delaware (Bishop et al. 2005, Hively et al. 2005) and may have other ameliorative effects on agricultural streams not yet evaluated (National Research Council 2000). Urban land use (2–11%) at WOH sites was relatively low and consisted primarily of residential single-family housing (population density ranged from 2–23 ind./km^2) that may have a smaller impact on stream biota than other types of urban land use. Most of the WOH sites were <6% urban and, at the 2 sites approaching 11% urban, most urban land use consisted of managed turf (lawns and recreational fields) (EBK, unpublished data). Agriculture and urbanization in the WOH did not correspond to changes in the WQS, but macroinvertebrate community variation along the landuse and chemistry gradients in the WOH could reflect initial community changes that may eventually lead to changes in the WQS, especially if anthropogenic land uses intensify without effective watershed management infrastructure.

Macroinvertebrate variation in relation to EOH environmental gradients

Macroinvertebrate communities at EOH sites varied dramatically in relation to a broad range of factors associated with human development. Endpoints along these gradients were primarily forested sites, sites affected by reservoir outlets, and more urbanized sites (e.g., with high road density and % transportation land use, and with or without more nutrients, waste indicators, and point-source discharges). Macroinvertebrate communities at sites below reservoir outlets were distinctly different from communities at forested or urbanized sites because numerous Trichoptera, especially filter-feeding taxa in the Hydropsychidae, Brachycentridae, and the polycentropodid *Chimarra* nr *obscura* that occurred at sites below the reservoir outlets were not found elsewhere. The relative abundance of chironomids and oligochaetes increased along the forested-to-urban gradient, and this increase

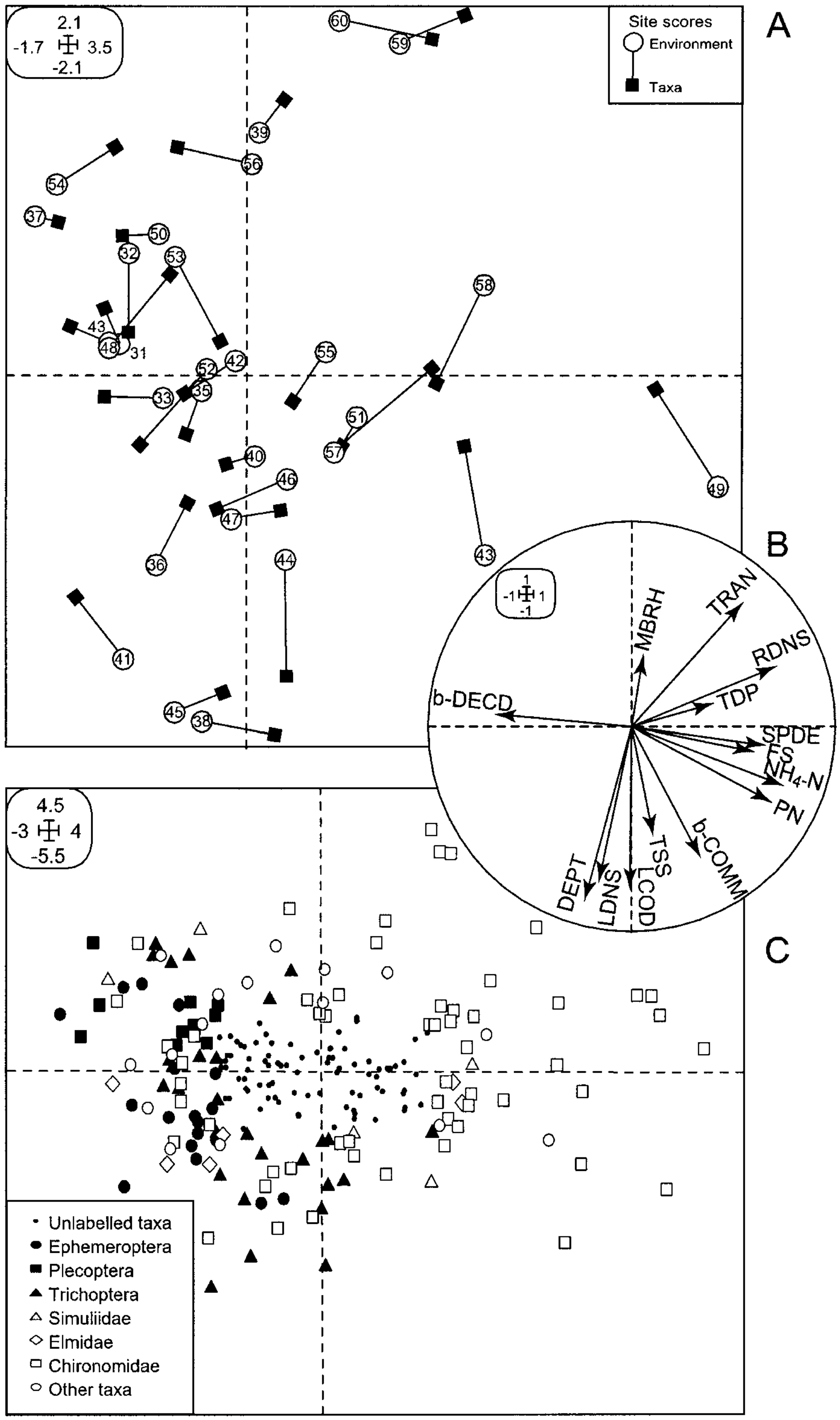

FIG. 6. Co-Inertia Analysis (CIA) of east of Hudson (EOH) sites. Ordination diagrams are for site scores either by species densities or environmental variables (A), environmental variables (B), and macroinvertebrate species densities coded by taxonomic group (C). In (A), the shorter the distance between the species and environment site scores, the better the agreement between the 2 analyses. Landuse variables with b- were quantified at the riparian scale (30-m buffer on each side of stream for entire stream length). All other landuse variables were quantified at the watershed scale. See Tables 1 and 2 for variable names and abbreviations. Insets show axis lengths.

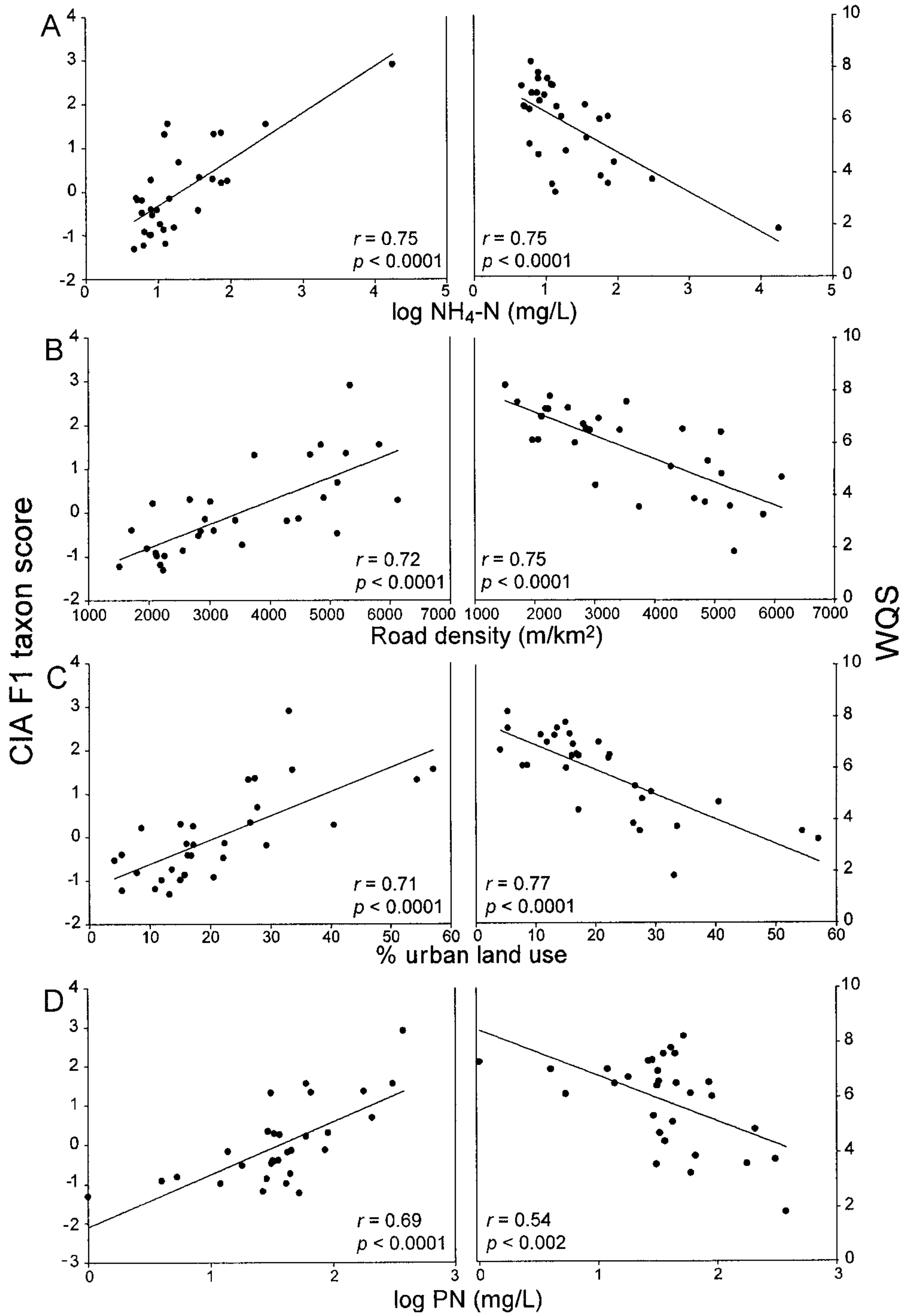

FIG. 7. Relationships between NH_4-N (A), road density (B), % urban land use (C; see definition in Fig. 5 caption), and particulate N (PN: D) for east of Hudson River (EOH) sites and either the Co-Inertia Analysis (CIA) Factor 1 site scores based on the macroinvertebrate data matrix or the macroinvertebrate water-quality score (WQS). Lines were drawn to aid in interpretation of relationships.

contributed to higher total densities at most of these sites. The relative abundance of chironomids also was high at sites 59 and 60 along the urban gradient, but these 2 sites did not have elevated densities of chironomids and oligochaetes. In fact, sites 59 and 60 had among the lowest total densities and WQSs in the EOH.

When described with the WQS, community structure changes also were strongly related to the urban landuse gradient (Figs 6 and 7) and were significant

TABLE 6. Pearson product–moment correlation coefficients (significant at $p < 0.05$) between variables from the east of Hudson River (EOH) Co-Inertia Analysis (CIA) and other environmental variables not included in the CIA. Landuse variables with b- were quantified at the riparian scale (30-m buffer on each side of stream for entire stream length). All other landuse variables were quantified at the watershed scale. See Tables 1 and 2 for variable names and abbreviations.

Land use category	CIA variable	Related variables	*r*
Urban	RDNS	b-RDNS	0.83
		PDNS	0.83
		b-PDNS	0.82
		RESD	0.75
		COND	0.73
		Cl	0.70
	SPDE	TDN	0.71

enough that 9 of 30 sites were classified as moderately or severely impacted. The change from a unimpacted (7.5–10) to a moderately impacted (5–7.5) classification represented an average 46% reduction in total species richness and 83% reduction in EPT richness. Furthermore, Ephemeroptera often were absent or present in low densities at moderately impacted sites (SWRC 2003). Streams that score in the moderately or severely impacted categories may be 303(d)-listed (Bode et al. 2002) and would, therefore, be targeted for remediation.

Reservoirs, lakes, and ponds, common landscape features in the EOH, were connected to various degrees to several of our EOH sites. Four sites in particular (38, 41, 44, 45) were downstream of large lakes/reservoirs and had macroinvertebrate communities that appeared to be exploiting the high concentrations of plankton that presumably occur

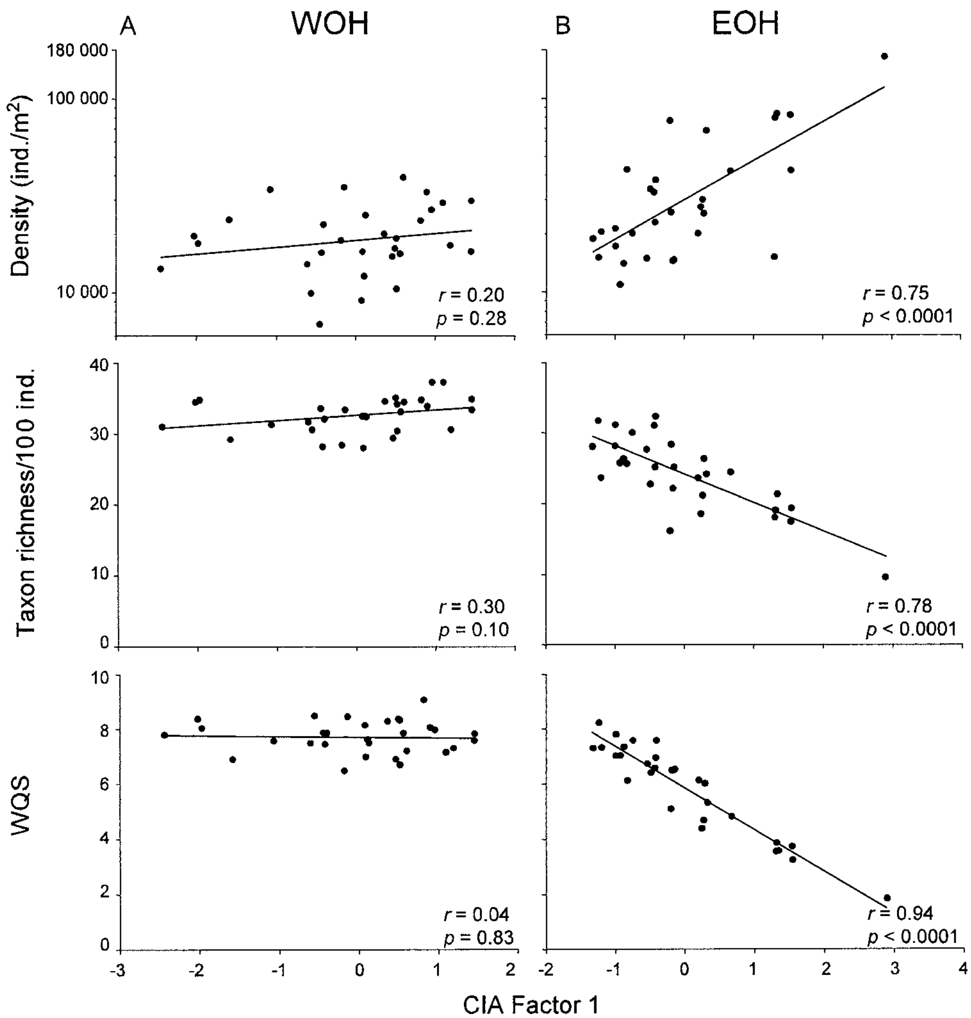

FIG. 8. Relationships between macroinvertebrate density, taxon richness/100 ind., and macroinvertebrate water-quality score (WQS) and Factor 1 macroinvertebrate Co-Inertia Analysis scores for sites west of Hudson River (WOH: A) and east of Hudson River (EOH: B).

immediately downstream of these productive reservoirs (e.g., Bott et al. 2006). Filtering caddisflies can be common below impoundments because of the availability and delivery of food (i.e., plankton and other suspended fine particles; Vannote et al. 1980, Valett and Stanford 1987, Richardson and Mackay 1991, Harding 1994). Previous studies also have noted the distinctiveness of lake-outlet macroinvertebrate communities in this region (Riva-Murray et al. 2002, Passy et al. 2004).

Most of the highly urbanized sites sampled in the EOH had macroinvertebrate communities characteristic of sites with significant organic and nutrient pollution (Wiederholm 1984, Hilsenhoff 1988, Lenat and Crawford 1994). Chironomids and oligochaetes were abundant, whereas pollution-sensitive EPT taxa were almost absent. The densities of macroinvertebrates suggest that hydrologic or toxic stressors were not great enough to prevent the high densities that can result from nutrient and organic enrichment. Macroinvertebrate communities in urban streams typically have low density, diversity, and EPT richness, and increased relative abundance of pollution-tolerant taxa relative to forested or agricultural streams (Hilsenhoff 1988, Novak and Bode 1992, Paul and Meyer 2001, Walsh et al. 2001, Stepenuck et al. 2002, Wang and Kanehl 2003). This condition was observed at only 2 of our highly urbanized sites in the EOH (sites 59 and 60, tributaries of the Kensico Reservoir).

Sites 59 and 60 were somewhat distinctive among the EOH sites in that they were highly urbanized but did not have known point-source discharges, and their nutrient and organic concentrations were not elevated (SWRC 2003). Transportation land use at site 60 was >50% of watershed-scale land use (i.e., Westchester County Airport), whereas development at site 59 was primarily transportation (~30%) and residential (~30%) land uses. Few studies have examined the specific effects of roads or transportation land use on aquatic macroinvertebrate communities (Paul and Meyer 2001, Ourso and Frenzel 2003), but pathways of disturbance are thought to occur through toxic effects of chemicals and heavy metals, hydrologic alterations, and sedimentation (Forman and Alexander 1998). Specific molecular tracers (i.e., PAHs) associated with crude and lubricating oils and other petroleum products were correlated with transportation land use (EBK, unpublished data), suggesting that runoff from road surfaces entered the stream channel. We do not have data on metals and toxic chemicals, but it is likely that toxins associated with roads and transportation also enter the stream. Also, both sites 59 and 60 probably have been affected, albeit in different ways, by altered flow regimes. Typically, high % impervious surface cover in watersheds results in high runoff volumes that damage stream habitat (Booth and Jackson 1997, Paul and Meyer 2001). This scenario is probably the case at site 59, but surface-water runoff from the airport adjacent to site 60 has been minimized with a stormwater abatement system (Westchester County Department of Transportation 2001). The relatively high amount of BOM and silt in the streambed suggests that site 60 had a reduced number of pulsed flow or flushing events to move fine material downstream. The contrast between sites 59 and 60 vs the other highly urbanized sites suggests that most of the highly urbanized sites sampled in the EOH were not typical urban streams in that macroinvertebrates at these sites were responding to elevated nutrient and organic concentrations rather than just to runoff from roads and other impervious surfaces. This complex response demonstrates the multivariate nature of disturbance in urban/suburban environments (Paul and Meyer 2001), but additional information (e.g., comparisons of lawn-care practices, waste-water collection, treatment, and release) is necessary before we can understand the differences between most of the urbanized EOH sites and typical urban streams from other studies.

The overall response of macroinvertebrates to urban land use, as measured by the WQS and represented by F1 in the CIA, was positive and linear (i.e., increased impairment with increased urban land use; Fig. 7C) with no obvious threshold among EOH sites. Cuffney et al. (2005) found a linear macroinvertebrate response to increasing urban intensity and suggested that macroinvertebrate communities begin to degrade as forest is converted to urban land use. Many other studies have identified metric thresholds within a range of 10 to 20% urban land use (Schueler 1994, Roy 2003, Allan 2004), though thresholds as low as 6% and 7% have also been suggested (Morse et al. 2003, Wang and Kanehl 2003). It is not clear why a macroinvertebrate response threshold was not found in our study, but the reason(s) could be related to metric components of the WQS, the level of taxonomy included in these metrics (species vs family), sampling methods, differences in quantification of the urban landuse category (= sum of RESD, COMM, INDU, TRAN, and OURB in table 2 of Dow et al. 2006), or variability in the age of the urban landscape. Threshold responses commonly have been examined with % impervious surface cover as a surrogate for urban land use. However, multiple urban stressors that contribute to impairment, such as combined sewer outflows, wastewater treatment plants, and other point sources may not be directly related to urban land use or % impervious surface cover.

Landuse scale and land use vs water chemistry

Watershed- and riparian-scale landuse variables were better predictors of macroinvertebrate communities than reach-scale landuse variables. Our results should not be interpreted as suggesting that local (streamside) land use is unimportant in these watersheds. Rather, our results indicate strong relationships between watershed conditions and macroinvertebrate community structure through the influence of watershed land use on instream conditions. Also, our study sites did not have severe local habitat degradation that would have intensified local landuse effects. The influence of the scale at which land use is quantified on macroinvertebrate communities can depend on the relatedness of variables among landuse scales, variation within scales, resolution of the landuse data set, proximity to the stream, the relationship between anthropogenic and natural gradients, and study design (Allan 2004). Different land uses may have unequal influences on stream ecosystem processes across scales (Black et al. 2004); thus, macroinvertebrate responses may vary accordingly. Allan (2004) suggested that watershed-scale land use may be the best predictor of macroinvertebrate community structure when flow instability, nutrients, or other factors related to the entire landscape are the primary mechanisms affecting the biota. Other researchers also have shown that watershed- or riparian-scale landuse variables are better predictors of macroinvertebrate communities than fine-scale landuse variables (Morley and Karr 2002, Strayer et al. 2003), although definitions of fine- or local-scale land use vary among studies. Results are mixed, however, because fine-scale (immediately adjacent to the stream) land use was the best predictor of macroinvertebrate communities in other studies (Sponseller et al. 2001, Lammert and Allan 1999, Stewart et al. 2001, Townsend et al. 2003).

Water chemistry and land use are tightly linked in the NYC drinking-water watersheds (Dow et al. 2006) and both sets of variables explained macroinvertebrate variation equally well. Results of previous landuse–water-chemistry investigations are mixed (Roy et al. 2003, Townsend et al. 2003, Wang and Kanehl 2003, and Death and Joy 2004), and the various suites of environmental variables included in different studies confound comparisons across studies. Furthermore, the relative influence of land use, water chemistry, or habitat condition on macroinvertebrates may vary among ecoregions (Weigel 2003) and may differ depending on how the macroinvertebrate community is described. Furthermore, the relative influence of land use, water chemistry, or habitat condition on macroinvertebrates may vary among ecoregions (Weigel 2003) and may differ depending on how the macroinvertebrate community is described. For example, Weigel et al. (2003) showed that watershed-scale land use and reach-scale habitat variables explained macroinvertebrate community metrics equally well, but reach-scale habitat variables were better than watershed-scale landuse variables at describing relative abundances and presence/absence of macroinvertebrates. These results illustrate the complex nature of analyses that attempt to elucidate causal relationships between anthropogenic activities and macroinvertebrate communities.

Metrics and multivariate analyses: implications for detecting and understanding impairment

Tributaries to the NYC drinking-water-supply reservoirs represent a continuum of conditions from mostly forested sites, predominantly in the WOH region, to sites with >50% urban land use in the EOH. Metrics such as WQS and total richness illustrated a severe impact gradient in the EOH, but these metrics were not responsive to less dramatic changes in environmental conditions in the WOH (i.e., WQS, EPT richness, and density were not correlated with CIA F1 in the WOH). Metrics reduce complex data sets to univariate measures, but they may eliminate variability that could be useful in detecting relationships between macroinvertebrate communities and environmental variables (Brooks et al. 2002). Our results suggest that this feature of metrics may be of particular concern when discerning differences in macroinvertebrate communities among sites representing a relatively small, but ecologically meaningful, range in environmental conditions, as in the WOH. Despite the lack of measurable impairment at WOH sites, multivariate analyses showed that taxonomic composition of sites did vary with changing environmental conditions and that macroinvertebrate communities responded not only to moderate-to-severe changes in land use, but to slight changes as well.

Stream biotic communities generally can withstand higher levels of agricultural than of urban land use before becoming impaired (Allan 2004). Thus, the current rate at which agricultural land is being converted to urban land is of increasing concern (Paul and Meyer 2001). No impairment was detected at 26% agricultural land use in the WOH, but the relationship between urban land use and WQS in the EOH indicated slightly to moderately impacted macroinvertebrate WQSs when urban land use reached 26%. Moore and Palmer (2005) showed that macroinvertebrate diversity decreased along an agricultural-to-urban gradient, and richness at agricultural sites was almost 2× the richness at urban sites; Fitzpatrick et al. (2004) observed that macroinvertebrate indices decreased as agriculture was replaced by urbanization. WOH urban land use was not severe enough to cause significant impairment in WOH communities, but development pressures, including proposed resorts, seasonal residences, and their associated infrastructure (NYC DEP 2004), are present in the region, and the impairment status of stream communities and quality of drinking water could change with increasing development.

Acknowledgements

We gratefully acknowledge the many people who contributed to the macroinvertebrate collection and identification portions of this research: Juliann Battle, Andrew Byler, William Crouch, David Funk, John Johansen, Bryan Lees, David Lieb, Amy MacCausland, William Milliken, David Montgomery, Sally Peirson, Roberta Weber, and numerous summer interns. Nicholas Principe and Heather Brooks assisted with SAS analyses. Comments by James Blaine, Pamela Silver, David Allan, and an anonymous referee greatly improved earlier versions of this manuscript. This research was funded by a Safe Drinking Water grant from the New York State Department of Environmental Conservation.

Literature Cited

ALLAN, J. D. 2004. Landscapes and riverscapes: the influence of land use on stream ecosystems. Annual Review of Ecology, Evolution, and Systematics 35:257–284.

ALLAN, J. D., AND L. B. JOHNSON. 1997. Catchment-scale analysis of aquatic ecosystems. Freshwater Biology 37: 107–111.

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. Professional Paper 964. US Geological Survey, Washington, DC.

ARSCOTT, D. B., C. L. DOW, AND B. W. SWEENEY. 2006a. Landscape template of New York City's drinking-water-supply watersheds. Journal of the North American Benthological Society 25:867–886.

ARSCOTT, D. B., J. K. JACKSON, AND E. B. KRATZER. 2006b. Role of rarity and taxonomic resolution in a regional and spatial analysis of stream macroinvertebrates. Journal of the North American Benthological Society 25:977–997.

AUFDENKAMPE, A. K., D. B. ARSCOTT, C. L. DOW, AND L. J. STANDLEY. 2006. Molecular tracers of soot and sewage contamination in streams supplying New York City drinking water. Journal of the North American Benthological Society 25:928–953.

BARBOUR, M. T., J. GERRITSEN, G. E. GRIFFITH, R. FRYDENBORG, E. MCCARRON, J. S. WHITE, AND M. L. BASTIAN. 1996. A framework for biological criteria for Florida streams using benthic macroinvertebrates. Journal of the North American Benthological Society 15:185–211.

BARBOUR, M. T., J. GERRITSEN, B. D. SNYDER, AND J. B. STRIBLING. 1999. Rapid bioassessment protocols for use in streams and wadeable rivers: periphyton, benthic macroinvertebrates and fish. 2nd edition. EPA 841-B-99–002. Office of Water, US Environmental Protection Agency, Washington, DC.

BISHOP, P. L., W. D. HIVELY, J. R. STEDINGER, M. R. RAFFERTY, J. L. LOJPERSBERGER, AND J. A. BLOOMFIELD. 2005. Multivariate analysis of paired watershed data to evaluate agricultural best management practice effects on stream water phosphorus. Journal of Environmental Quality 34:1087–1101.

BLACK, R. W., M. D. MUNN, AND R. W. PLOTNIKOFF. 2004. Using macroinvertebrates to identify biota–land cover optima at multiple scales in the Pacific Northwest, USA. Journal of the North American Benthological Society 23:340–362.

BLAINE, J. G., B. W. SWEENEY, AND D. B. ARSCOTT. 2006. Enhanced source-water monitoring for New York City: historical framework, political context, and project design. Journal of the North American Benthological Society 25:851–866.

BODE, R. W., M. A. NOVAK, L. E. ABELE, D. L. HEINTZMAN, AND A. J. SMITH. 2002. Quality assurance work plan for biological stream monitoring in New York State. New York State Department of Environmental Conservation, Albany, New York. (Available from: New York State Department of Environmental Conservation, 625 Broadway St., Albany, New York, 12233.)

BOOTH, D. B., AND C. R. JACKSON. 1997. Urbanization of aquatic systems: degradation thresholds, stormwater detection, and the limits of mitigation. Journal of the American Water Resources Association 33:1077–1090.

BORCARD, D., P. LEGENDRE, AND P. DRAPEAU. 1992. Partialling out the spatial component of ecological variation. Ecology 73:1045–1055.

BOTT, T. L., D. S. MONTGOMERY, D. B. ARSCOTT, AND C. L. DOW. 2006. Primary productivity in receiving reservoirs: links to influent streams. Journal of the North American Benthological Society 25:1045–1061.

BROOKS, S. S., M. A. PALMER, B. J. CARDINALE, C. M. SWAN, AND S. RIBBLETT. 2002. Assessing stream ecosystem rehabilitation: limitations of community structure. Restoration Ecology 10:156–168.

CUFFNEY, T. F., H. ZAPPIA, E. M. P. GIDDINGS, AND J. F. COLES. 2005. Effects of urbanization on benthic macroinvertebrate assemblages in contrasting environmental settings: Boston, Massachusetts; Birmingham, Alabama; and Salt Lake City, Utah. American Fisheries Society Symposium 47:361–407.

DEATH, R. G., AND M. K. JOY. 2004. Invertebrate community structure in streams of the Manawatu-Wanganui region, New Zealand: the roles of catchment versus reach scale influences. Freshwater Biology 49:982–997.

DELONG, M. D., AND M. A. BRUSVEN. 1998. Macroinvertebrate community structure along the longitudinal gradient of an agriculturally impacted stream. Environmental Management 22:445–457.

DODDS, W. K., AND M. R. WHILES. 2004. Quality and quantity of suspended particles in rivers: continent-scale patterns in the United States. Environmental Management 33: 355–367.

DOLÉDEC, S., AND D. CHESSEL. 1994. Co-inertia analysis: an alternative method for studying species-environment relationships. Freshwater Biology 31:277–294.

DOVCIAK, A. L., AND J. A. PERRY. 2002. In search of effective scales for stream management: does agroecoregion, watershed, or their intersection best explain the variance in stream macroinvertebrate communities? Environmental Management 30:365–377.

DOW, C. L., D. B. ARSCOTT, AND J. D. NEWBOLD. 2006. Relating major ions and nutrients to watershed conditions across a mixed-use, water-supply watershed. Journal of the North American Benthological Society 25:887–911.

DUDLEY, N., AND S. STOLTON. 2003. Running pure: the importance of forest protected areas to drinking water. World Bank/WWF Alliance for Forest Conservation and Sustainable Use, Gland, Switzerland. (Available from: World Bank/WWF Alliance for Forest Conservation and Sustainable Use, Ave. du Mont Blanc, CH-1196, Gland, Switzerland.)

EARLE, J. I. 2004. Stonefly (Plecoptera) species of an acidic intermittent stream in southwestern Pennsylvania, USA, comparison with a circumneutral perennial stream and consideration of survival strategies. Archiv für Hydrobiologie 159:97–114.

ENGLAND, L. E., AND A. D. ROSEMOND. 2004. Small reductions in forest cover weaken terrestrial-aquatic linkages in headwater streams. Freshwater Biology 49:721–734.

FEMINELLA, J. W. 2000. Correspondence between stream macroinvertebrate assemblages and 4 ecoregions of the southeastern USA. Journal of the North American Benthological Society 19:442–461.

FITZHUGH, T. W., AND B. D. RICHTER. 2004. Quenching urban thirst: growing cities and their impacts on freshwater ecosystems. BioScience 54:741–754.

FITZPATRICK, F. A., M. A. HARRIS, T. L. ARNOLD, AND K. D. RICHARDS. 2004. Urbanization influences on aquatic communities in northeastern Illinois streams. Journal of the American Water Resources Association 40:461–475.

FORMAN, R. T. T., AND L. E. ALEXANDER. 1998. Roads and their major ecological effects. Annual Review of Ecology and Systematics 29:207–231.

FRISSELL, C. A., W. J. LISS, C. E. WARREN, AND M. D. HURLEY. 1986. A hierarchical framework for stream habitat classification: viewing streams in a watershed context. Environmental Management 10:199–214.

GRIFFITH, M. B., S. A. PERRY, AND W. B. PERRY. 1993. Growth and secondary production of *Paracapnia angulata* (Plecoptera, Capniidae) in Appalachian streams affected by acid precipitation. Canadian Journal of Zoology 71:735–743.

HARDING, J. S. 1994. Variations in benthic fauna between differing lake outlet types in New Zealand. New Zealand Journal of Marine and Freshwater Research 28:417–427.

HAWKINS, C. P., AND R. H. NORRIS. 2000. Performance of different landscape classifications for aquatic bioassessments: introduction to the series. Journal of the North American Benthological Society 19:367–369.

HAWKINS, C. P., R. H. NORRIS, J. GERRITSEN, R. M. HUGHES, S. K. JACKSON, R. K. JOHNSON, AND R. J. STEVENSON. 2000. Evaluation of the use of landscape classifications for the prediction of freshwater biota: synthesis and recommendations. Journal of the North American Benthological Society 19:541–556.

HAWKINS, C. P., AND M. R. VINSON. 2000. Weak correspondence between landscape classifications and stream invertebrate assemblages: implications for bioassessment. Journal of the North American Benthological Society 19:501–517.

HILSENHOFF, W. L. 1988. Rapid field assessment of organic pollution with a family-level biotic index. Journal of the North American Benthological Society 7:65–68.

HIVELY, W. D., R. B. BRYANT, AND T. J. FAHEY. 2005. Phosphorus concentrations in overland flow from diverse locations on a New York dairy farm. Journal of Environmental Quality 34:1224–1233.

KAPLAN, L. A., J. D. NEWBOLD, D. J. VAN HORN, C. L. DOW, A. K. AUFDENKAMPE, AND J. K. JACKSON. 2006. Organic matter transport in New York City drinking-water-supply watersheds. Journal of the North American Benthological Society 25:912–927.

KING, R. S., M. E. BAKER, D. F. WHIGHAM, D. E. WELLER, T. E. JORDAN, P. F. KAZYAK, AND M. K. HURD. 2005. Spatial considerations for linking watershed land cover to ecological indicators in streams. Ecological Applications 15:137–153.

LAMMERT, M., AND J. D. ALLAN. 1999. Assessing biotic integrity of streams: effects of scale in measuring the influence of land use/land cover and habitat structure on fish and macroinvertebrates. Environmental Management 23: 679–694.

LENAT, D. R., AND J. K. CRAWFORD. 1994. Effects of land use on water quality and aquatic biota of three North Carolina Piedmont streams. Hydrobiologia 294:185–199.

LEPŠ, J., AND P. ŠMILAUER. 2003. Multivariate analysis of ecological data using CANOCO. Cambridge University Press, Cambridge, UK.

LI, J., A. HERLIHY, W. GERTH, P. KAUFMANN, S. GREGORY, S. URQUHART, AND D. P. LARSEN. 2001. Variability in stream macroinvertebrates at multiple spatial scales. Freshwater Biology 46:87–97.

LORENZEN, C. J. 1967. Determination of chlorophyll and pheopigments: spectrophotometric equations. Limnology and Oceanography 12:343–346.

MALMQVIST, B. 2002. Aquatic invertebrates in riverine landscapes. Freshwater Biology 47:679–694.

MARCHANT, R., F. WELLS, AND P. NEWALL. 2000. Assessment of an ecoregion approach for classifying macroinvertebrate assemblages from streams in Victoria, Australia. Journal of the North American Benthological Society 19:497–500.

MEADOR, M. R., AND R. M. GOLDSTEIN. 2003. Assessing water quality at large geographic scales: relations among land use, water physicochemistry, riparian condition, and fish community structure. Environmental Management 31: 504–517.

MEHAFFEY, M. H., M. S. NASH, T. G. WADE, C. M. EDMONDS, D. W. EBERT, K. B. JONES, AND A. RAGER. 2001. A landscape assessment of the Catskill/Delaware watersheds 1975–1998: New York City's water supply watersheds. EPA/600/R-01/075. Office of Research and Development, National Exposure Research Laboratory, Environmental Sciences Division, US Environmental Protection Agency, Las Vegas, Nevada.

MOORE, A. A., AND M. A. PALMER. 2005. Invertebrate biodiversity in agricultural and urban headwater streams: implications for conservation and management. Ecological Applications 15:1169–1177.

MORLEY, S. A., AND J. R. KARR. 2002. Assessing and restoring the health of urban streams in the Puget Sound Basin. Conservation Biology 16:1498–1509.

MORSE, C. C., A. D. HURYN, AND C. CRONAN. 2003. Impervious surface area as a predictor of the effects of urbanization on stream insect communities in Maine, U.S.A. Environmental Monitoring and Assessment 89:95–127.

NATIONAL RESEARCH COUNCIL. 2000. Watershed management for potable water supply: assessing the New York City strategy. National Academy Press, Washington, DC.

NOVAK, M. A., AND R. W. BODE. 1992. Percent model affinity: a new measure of macroinvertebrate community composition. Journal of the North American Benthological Society 11:80–85.

NYC DEP (NEW YORK CITY DEPARTMENT OF ENVIRONMENTAL PROTECTION). 2004. The New York City Department of Environmental Protection's comments on the draft environmental impact statement and draft state pollutant discharge elimination system permits for the proposed Belleayre Resort at Catskill Park Project. New York City Department of Environmental Protection, Valhalla, New York. (Available from: New York City Department of Environmental Protection, 465 Columbus Ave, Valhalla, New York, 10595 USA.)

OMERNIK, J. M. 1987. Ecoregions of the conterminous United States. Map Supplement (scale 1: 7,5000,000). Annals of the Association of American Geographers 77:118–125.

OURSO, R. T., AND S. A. FRENZEL. 2003. Identification of linear and threshold responses in streams along a gradient of urbanization in Anchorage, Alaska. Hydrobiologia 501: 117–131.

PASSY, S. I., R. W. BODE, D. M. CARLSON, AND M. A. NOVAK. 2004. Comparative environmental assessment in the studies of benthic diatom, macroinvertebrate, and fish communities. International Review of Hydrobiology 89: 121–138.

PAUL, J. P., AND J. L. MEYER. 2001. Streams in the urban landscape. Annual Review of Ecology and Systematics 32:333–365.

PHILLIPS, P. J., D. A. ECKHARDT, D. A. FREEHAFER, G. R. WALL, AND H. H. INGLESTON. 2002. Regional patterns of pesticide concentrations in surface waters of New York in 1997. Journal of the American Water Resources Association 38: 731–745.

POFF, N. L. 1997. Landscape filters and species traits: towards mechanistic understanding and prediction in stream ecology. Journal of the North American Benthological Society 16:391–409.

POFF, N. L., J. D. ALLAN, M. B. BAIN, J. R. KARR, K. L. PRESTEGAARD, B. D. RICHTER, R. E. SPARKS, AND J. C. STROMBERG. 1997. The natural flow regime: a paradigm for river conservation and restoration. BioScience 47:769–784.

RABENI, C. F., AND K. E. DOISY. 2000.Correspondence of stream benthic invertebrate assemblages to regional classification schemes in Missouri. Journal of the North American Benthological Society 19:419–428.

RICHARDS, C., R. J. HARO, L. B. JOHNSON, AND G. E. HOST. 1997. Catchment and reach-scale properties as indicators of macroinvertebrate species traits. Freshwater Biology 37: 219–230.

RICHARDS, C., L. B. JOHNSON, AND G. E. HOST. 1996. Landscape-scale influences on stream habitats and biota. Canadian Journal of Fisheries and Aquatic Sciences 53(Supplement 1):295–311.

RICHARDSON, J. S., AND R. J. MACKAY. 1991. Lake outlets and the distribution of filter feeders: an assessment of hypotheses. Oikos 62:370–380.

RIVA-MURRAY, K., R. W. BODE, P. J. PHILLIPS, AND G. L. WALL. 2002. Impact source determination with biomonitoring data in New York State: concordance with environmental data. Northeastern Naturalist 9:127–162.

ROSEMOND, A. D., S. R. REICE, J. W. ELWOOD, AND P. J. MULHOLLAND. 1992. The effects of stream acidity on benthic macroinvertebrate communities in the southeastern United States. Freshwater Biology 27:193–209.

ROSENBERG, D. M., AND V. H. RESH, (EDITORS). 1993. Freshwater biomonitoring and benthic macroinvertebrates. Chapman and Hall, New York.

ROY, A. H., A. D. ROSEMOND, M. J. PAUL, D. S. LEIGH, AND J. B. WALLACE. 2003. Stream macroinvertebrate response to catchment urbanisation (Georgia, U.S.A.). Freshwater Biology 48:329–346.

SCHUELER, T. R. 1994. The importance of imperviousness. Watershed Protection Technology 1:100–111.

SIMPSON, K. W., R. W. BODE, AND J. R. COLQUHOUN. 1985. The macroinvertebrate fauna of an acid-stressed headwater stream system in the Adirondack Mountains, New York. Freshwater Biology 15:671–681.

SPONSELLER, R. A., E. F. BENFIELD, AND H. M. VALETT. 2001. Relationships between land use, spatial scale and stream macroinvertebrate communities. Freshwater Biology 46: 1409–1424.

STEPENUCK, K. F., R. L. CRUNKILTON, AND L. Z. WANG. 2002. Impacts of urban land use on macroinvertebrate communities in southeastern Wisconsin streams. Journal of the American Water Resources Association 38:1041–1051.

STEWART, J. S., L. Z. WANG, J. LYONS, J. A. HORWATICH, AND R. BANNERMAN. 2001. Influences of watershed, riparian-corridor, and reach-scale characteristics on aquatic biota in agricultural watersheds. Journal of the American Water Resources Association 37:1475–1487.

STONE, M. L., M. R. WHILES, J. A. WEBBER, K. W. J. WILLIARD, AND J. D. REEVE. 2005. Macroinvertebrate communities in agriculturally impacted southern Illinois streams: patterns with riparian vegetation, water quality, and instream habitat quality. Journal of Environmental Quality 34:907–917.

STRAYER, D. L., R. E. BEIGHLEY, L. C. THOMPSON, S. BROOKS, C. NILSSON, G. PINAY, AND R. J. NAIMAN. 2003. Effects of land cover on stream ecosystems: roles of empirical models and scaling issues. Ecosystems 6:407–423.

SWEENEY, B. W. 1993. Effects of streamside vegetation on macroinvertebrate communities of White Clay Creek in Eastern North America. Proceedings of the Academy of Natural Sciences of Philadelphia 144:291–340.

SWRC (STROUD WATER RESEARCH CENTER). 2003. Water quality monitoring in the source water areas for New York City: an integrative approach. A report on the first phase of monitoring. Stroud Water Research Center, Avondale, Pennsylvania. (Available from: http://www.stroudcenter.org/research/nyproject.htm)

TER BRAAK, C. J. F. 1986. Canonical correspondence analysis: a new eigenvector technique for multivariate direct gradient analysis. Ecology 67:1167–1179.

TER BRAAK, C. J. F. 1995. Ordination. Pages 91–173 *in* R. H. Jongman, C. J. F. ter Braak, and O. F. R. van Tongeren (editors). Data analysis in community and landscape ecology. Cambridge University Press, Cambridge, UK.

THIOULOUSE, J., D. CHESSEL, S. DOLÉDEC, AND J.-M. OLIVIER. 1997. ADE-4: a multivariate analysis and graphical display software. Statistics and Computing 7:75–83.

TOWNSEND, C. R., S. DOLÉDEC, R. NORRIS, K. PEACOCK, AND C. J. ARBUCKLE. 2003. The influence of scale and geography on relationships between stream community composition and landscape variables: description and prediction. Freshwater Biology 48:768–785.

VALETT, H. M., AND J. A. STANFORD. 1987. Food quality and hydropsychid caddisfly density in a lake outlet stream in Glacier National Park, Montana, U.S.A. Canadian Journal of Fisheries and Aquatic Sciences 44:77–82.

VANNOTE, R. L., G. W. MINSHALL, K. W. CUMMINS, J. R. SEDELL, AND C. E. CUSHING. 1980. The river continuum concept. Canadian Journal of Fisheries and Aquatic Sciences 32: 130–137.

WAITE, I. R., A. T. HERLIHY, D. P. LARSEN, AND D. J. KLEMM. 2000. Comparing strengths of geographic and nongeographic classifications of stream benthic macroinvertebrates in the Mid-Atlantic Highlands, USA. Journal of the North American Benthological Society 19:429–441.

WALLACE, J. B., S. L. EGGERT, J. L. MEYER, AND J. R. WEBSTER. 1999. Effects of resource limitation on a detrital-based ecosystem. Ecological Monographs 69:409–442.

WALSH, C. J., A. K. SHARPE, P. F. BREEN, AND J. A. SONNEMAN. 2001. Effects of urbanization on streams of the Melbourne region, Victoria, Australia. I. Benthic macroinvertebrate communities. Freshwater Biology 46:535–551.

WANG, L. H., AND P. KANEHL. 2003. Influences of watershed urbanization and instream habitat on macroinvertebrates in cold water streams. Journal of the American Water Resources Association 39:1181–1196.

WEIGEL, B. M. 2003. Development of stream macroinvertebrate models that predict watershed and local stressors in Wisconsin. Journal of the North American Benthological Society 22:123–142.

WEIGEL, B. M., L. WANG, P. W. RASMUSSEN, J. T. BUTCHER, P. M. STEWART, T. P. SIMON, AND M. J. WILEY. 2003. Relative influence of variables at multiple spatial scales on stream macroinvertebrates in the Northern Lakes and Forest ecoregion, U.S.A. Freshwater Biology 48:1440–1461.

WESTCHESTER COUNTY DEPARTMENT OF TRANSPORTATION. 2001. Study of current and future aircraft deicing practices and procedures at Westchester County Airport. Westchester County Department of Transportation, White Plains, New York. (Available from: Westchester County Airport, 240 Airport Rd, Suite 202, White Plains, New York 10604 USA.)

WIEDERHOLM, T. 1984. Responses of aquatic insects to environmental pollution. Pages 508–557 *in* V. H. Resh and D. M. Rosenberg (editors). The ecology of aquatic insects. Praeger Publishers, New York.

Received: 21 November 2005
Accepted: 13 July 2006