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Effects of Urbanization on Benthic Macroinvertebrate Assemblages in Contrasting Environmental Settings: Boston, Massachusetts; Birmingham, Alabama; and Salt Lake City, Utah

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Abstract.—Responses of invertebrate assemblages along gradients of urban intensity were examined in three metropolitan areas with contrasting climates and topography (Boston, Massachusetts; Birmingham, Alabama; Salt Lake City, Utah). Urban gradients were defined using an urban intensity index (UII) derived from basin-scale population, infrastructure, land-use, land-cover, and socioeconomic characteristics. Responses based on assemblage metrics, indices of biotic integrity (B-IBI), and ordinations were readily detected in all three urban areas and many responses could be accurately predicted simply using regional UIIs. Responses to UII were linear and did not indicate any initial resistance to urbanization. Richness metrics were better indicators of urbanization than were density metrics. Metrics that were good indicators were specific to each study except for a richness-based tolerance metric (TOLr) and one B-IBI. Tolerances to urbanization were derived for 205 taxa. These tolerances differed among studies and with published tolerance values, but provided similar characterizations of site conditions. Basin-scale land-use changes were the most important variables for explaining invertebrate responses to urbanization. Some chemical and instream physical habitat variables were important in individual studies, but not among studies. Optimizing the study design to detect basin-scale effects may have reduced the ability to detect local-scale effects.

Introduction

Urban lands represent only a small component of human engendered landscape alteration in the United States (U.S. Environmental Protection Agency 2000), but these lands have a disproportionate effect on stream condition. It is estimated that 1 km^2 of urbanized basin impairs three times (0.15 km) the length of stream that would be impaired by a similar amount of agricultural land (National Resources Conservation Service 2000; U.S. Environmental Protection Agency 2000). The extent of urbanized land is also increasing rapidly (about 101,000 km^2 between 1987 and 1997). Consequently, urbanization is a significant source of stream impairment in the United States that will be steadily increasing for the foreseeable future. Understanding how

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urbanization affects physical, chemical, and biological characteristics of streams and the similarities and differences in these effects throughout the United States is important for managing aquatic resources.

The number of urban stream studies has increased substantially in recent years and the effects of urbanization are well documented for selected urban areas (Paul and Meyer 2001) and for invertebrate assemblages (Klein 1979; Jones and Clark 1987; Schueler and Galli 1992; Lenat and Crawford 1994; Yoder and Rankin 1996; Horner et al. 1997; Kemp and Spotila 1997; Kennen 1999; Yoder et al. 1999; Beasley and Kneale 2002; Huryn et al. 2002; Kennen and Ayers 2002; Morley and Karr 2002; Morse et al. 2003; Ourso and Frenzel 2003; Roy et al. 2003; Vølstad et al. 2003). While the effects of urbanization have been well established, the intensity of development that brings about ecological changes, the rate and form of these changes, and regional differences in responses are less clearly understood (Karr and Chu 1999, 2000). Single variable surrogates for urban intensity, such as population density or measures of impervious surface (Arnold and Gibbons 1996), are often used to represent urban intensity and interpret responses to urbanization. However, a comprehensive understanding of the ecological effects of urbanization (i.e., rates and forms of responses) requires an understanding of the interactions of a large variety of physical, chemical, and biological factors that change along gradients of urbanization and that vary locally and regionally. These, along with differences in study design and sampling methods, hinder extrapolating study results from one region of the country to another. Multiple regional urban studies using a common design and sample collection techniques are needed to provide a comprehensive understanding of regional responses to urbanization that are comparable among different environmental settings.

In 1999, the U.S. Geological Survey's (USGS) National Water-Quality Assessment (NAWQA) Program initiated a series of studies that used a common design to examine the regional effects of urbanization on aquatic biota (fish, invertebrates, and algae) and chemical and physical habitat in three metropolitan areas in different environmental settings. These urban gradient studies were conducted in the Boston, Massachusetts (BOS), Birmingham, Alabama (BIR), and Salt Lake City, Utah (SLC) metropolitan areas. A multimetric urban intensity index (UII) was used to identify representative gradients of urbanization within relatively homogeneous environmental settings (McMahon and Cuffney 2000; Tate et al. 2005, this volume) associated with each urban area. The objectives of these studies were to (1) determine if physical, chemical, and biological characteristics of streams responded to urban intensity as defined by the UII; (2) describe the form and rate of these responses; (3) determine which characteristics are useful indicators of urbanization; (4) identify characteristics of urbanization that are most strongly associated with biological responses; and (5) compare responses among urban areas.

This paper describes the responses of invertebrate assemblages. Responses of algae, fish, and physical habitat structure are described in Potapova et al. (2005, this volume), Meador et al. (2005, this volume), and Short et al. (2005, this volume). Tate et al. (2005) describe the design of these urban land-use gradient studies.

Methods

Site Selection

Sampling sites were chosen from populations of candidate basins (2nd–5th order) defined using 30-m digital elevation models (U.S. Geological Survey 2003). Natural and anthropogenic basin characteristics were derived from publicly available information sources. A UII was used to select 30 study basins that represented a gradient of urban intensity from low (0) to high (100). The UII was derived from a combination of land use, infrastructure, population, and socioeconomic variables (McMahon and Cuffney 2000; Tate et al. 2005) that were associated with changes in population density. Study sites were chosen to minimize differences in natural basin features (e.g., ecoregion, climate, topography, stream size) and local disturbances (e.g., major point sources or modifications to riparian vegetation, channels, banks, or beds) as a means of maximizing the ability to detect basinwide urban effects as opposed to local-scale effects (Morley and Karr 2002). The different urban intensities represented by the spatially distributed sampling network are intended to represent changes in urbanization through time (i.e., substitute space for time).

Urban intensity indexes were developed individually for each study area to take full advantage of the unique land-use, land-cover, infrastructure, population, and socioeconomic data available in each. These UIIs represent the range of urban intensity in each study area, but the variables comprising the index differed among study areas. A common urban intensity index (CUII) was also calculated based on a set of five urban indicator variables that were common to all three study areas (Tate et al. 2005). The CUII pro-

vides a measure of urban intensity that is consistent among the study areas, whereas the UII provides a measure of urban intensity that makes maximum use of local indicators of urbanization.

Water Samples

Water column chemistry data (i.e., nitrogen species, phosphorus species, major ions, and pesticides) were collected during summer low flows (BOS August, BIR May, SLC July) in 2000. Additional samples were collected in BOS (April 2000) and BIR (May 2001). These samples were collected from equal-width increments across the stream channel, composited, and processed on site in accordance with standard NAWQA Program protocols (Shelton 1994). Water samples were sent to the USGS National Water Quality Laboratory in Denver, Colorado, for analysis. Dissolved oxygen, pH, specific conductance, and alkalinity were measured during each site visit (every 2–4 weeks). Water temperature was measured continuously every 15 min at each site using temperature probes and data recorders. Streamstage measurements were recorded at 15-min intervals at each site using either existing USGS streamgages or a stage transducer. Temperature and stage recorders were removed from SLC during winter. Water chemistry, temperature, and stage measurements were collected for approximately 1 year prior to collecting biological samples. Because of equipment malfunctions, the temperature data collection in BOS was extended through 2001 to characterize the annual thermal cycle. Chemical characteristics were summarized as the mean of all values collected during the study.

Stream Physical Habitat Characterization

Stream physical habitat characteristics were quantified during summer low flows using standard NAWQA Program protocols (Fitzpatrick et al. 1998). Data were collected from 11 transects at each site, and 65 habitat metrics were calculated from these data. Details on habitat sampling and the derivation of habitat metrics are given in Short et al. (2005).

Invertebrate Samples

Standard NAWQA Program sampling protocols were used to collect benthic macroinvertebrates (Cuffney et al. 1993) during summer low flows in 2000: BOS August, BIR June, and SLC July. Two types of macroinvertebrate samples were collected—a quantitative sample collected from multiple representatives of the stream habitat that contained the richest assemblage of invertebrates (richest targeted habitat, RTH) and a qualitative multihabitat (QMH) sample that collected invertebrates from as many habitats in the stream reach as were accessible. The RTH sample consisted of Slack (Cuffney et al. 1993) samples (0.25 m^2, 425-μm-mesh net) collected from five separate riffle areas in the sampling reach and combined to form a single composite sample of 1.25 m^2. One SLC site (Kays Creek at Layton; Tate et al. 2005) did not have enough riffle habitat, so the RTH sample was collected from at least two woody snags at each of five locations along the stream reach. The QMH sample collected invertebrates using a 212-μm-mesh dip net supplemented with hand-picking of substrates. Sampling effort (time) was apportioned as equally as possible among accessible habitats in the sampling reach. Data from the QMH and RTH samples were combined to form a qualitative composite sample (QRC) that provided a comprehensive list of invertebrates in each sampling reach. Samples were preserved in 10% buffered formalin and sent to the USGS National Water Quality Laboratory in Denver, Colorado for taxa identification and enumeration. Invertebrate samples were processed using standard NAWQA Program protocols (Moulton et al. 2000) for RTH (randomized 300-organism count) and QMH (fixed processing time designed to maximize the number of taxa enumerated) samples.

Resolving Taxonomic Ambiguities

Taxonomic ambiguities arise when results (abundance or presence) are reported at multiple taxonomic levels. For example, an ambiguity would exist in a sample when data are reported for the species *Hydropsyche sparna* and *H. betteni*, the genus *Hydropsyche*, and the family Hydropsychidae. In this example, *Hydropsyche* and Hydropsychidae are ambiguous parent taxa because they may belong to either *betteni, sparna*, or to another unidentified child species or genus in the case of Hydropschidae. Taxa richness in this sample could range from 1 (Hydropsychidae) to 4 depending on how the analyst decides to resolve ambiguities. The method used to resolve ambiguous taxa can strongly influence the analysis and interpretation of assemblage data.

Taxonomic ambiguities were resolved using the Invertebrate Data Analysis System software (IDAS; Cuffney 2003). Ambiguities in RTH samples were resolved separately for each study area using option RC3. This option combines RTH samples from a study area, identifies ambiguities in the combined data, and

determines whether to delete or combine ambiguous taxa depending on their abundances. If the combined abundance of the children is greater than the abundance of the parent, the parent is deleted and the children are retained. If the abundance of the parent is greater than the combined abundance of the children, the abundances of the children are added to the parent and the children are dropped. The decision of which taxa to keep, combine, or delete are then applied to each of the samples individually.

Ambiguities in qualitative samples (QMH or QRC) were resolved separately for each study area using method RC1. This method combines samples from a study area, identifies ambiguous parents in the combined data, and tags them for deletion. Decisions on which taxa to delete are then applied to each sample individually. If a sample contains an ambiguous parent but no associated children, the most commonly occurring child is substituted for the ambiguous parent. This method is appropriate for qualitative samples because the presence of children implies the existence of parents and there is no quantitative information to lose.

Data Analysis

Invertebrate responses were analyzed using a combination of multivariate (ordination) and multimetric analyses that reduced assemblage data into a series of simple response variables. The correspondence between response variables and the UII was examined to determine if assemblages were responding to urbanization and, if so, the form and rate of the response. Invertebrate responses were then compared to land-use, land-cover, topographic, lithologic, soils, population, socioeconomic, habitat, and chemical variables to ascertain which variables were important in explaining responses and how responses and explanatory variables compared among studies. Quantitative data were converted to densities (number/m^2) and taxonomic ambiguities were resolved prior to calculating metrics or conducting ordinations.

Multivariate analysis.—Indirect gradient analysis (ter Braak 1995) was used to investigate the relations between invertebrate responses and explanatory variables separately for each urban study area. This is a two-part procedure that uses ordination to derive response gradients (latent environmental variables) that summarize the assemblages and then relates these gradients to explanatory variables using correlation and/or regression analysis. The response gradients derived from ordination are the site scores whose position along the axis is determined by dissimilarity among assemblages at each site. That is, sites with similar assemblages are located close together on an ordination axis and sites with dissimilar assemblages are located far apart.

Ordinations were conducted for both qualitative (QRC) and quantitative (RTH) data using correspondence analysis (CA) and detrended correspondence analysis (DCA; CANOCO v. 4.0, ter Braak and Šmilauer 1998). Rare taxa were down weighted in all ordinations using the methods of ter Braak and Šmilauer (1998). Ordination of the BOS and BIR RTH samples required transformation of the data ($\log x + 1$) and detrending (DCA). RTH data from SLC were square-root transformed but did not require detrending. Data from QRC samples did not require transformation or detrending. Scaling was focused on intersample distances and detrending was by segments.

The ordination site scores (derived response gradients) for ordination axes 1–4 were correlated (Spearman rank correlation, ρ) with the UII to determine which axis was most strongly associated with urban intensity, the strength of the association, and the sign of the association. If the association was positive, the site scores were transformed (i.e., individual site scores were subtracted from the maximum site score for each study area) to produce responses that were similar and consistent with the response of the EPTr metric (i.e., decrease in value as urbanization increases, Paul and Meyer 2001; Morse et al. 2003). If the ordination contained negative values for site scores, they were adjusted to positive values by subtracting the minimum site score from all site scores. These transformations do not alter the relation (ecological distances) among sites, but are required to produce consistent and comparable responses among ordinations derived using different procedures (i.e., DCA, CA) and data transformations. If an axis was at least moderately associated with UII ($|\rho| \geq 0.5$), regression analysis was used to determine if there was a statistically significant response (slope, $b \neq 0$, $P \leq 0.05$)). Analysis of covariance (ANCOVA) and Newman-Keuls multiple range test (Zar 1974) were used to determine if there were significant differences in responses (slopes) among urban areas.

Associations between invertebrate responses and explanatory variables (e.g., land-use, land-cover, topography, elevation, chemistry, soils, and habitat variables) were identified based on Spearman rank correlations (ρ). These correlations provided an effective means of summarizing relations between variables even when the underlying responses were not linear

or contained outliers. Spearman rank correlations and regression analyses were calculated using SYSTAT 9 (SPSS 1999). Spearman correlations were considered to be strong when $|\rho| \geq 0.7$ and moderate when $0.7 > |\rho| \geq 0.5$, after rounding correlations to the nearest tenth. These criteria provided an effective and efficient mechanism for selecting a subset of variables that merited more intensive investigation as potential explanatory variables.

Assemblage metrics.—The IDAS program was used to calculate 137 invertebrate metrics (Table 1) that are commonly used in bioassessment (Rosenberg and Resh 1993; Davis and Simon 1995; Barbour et al. 1999). Tolerance and functional group metrics were calculated using data from Barbour et al. (1999), supplemented with tolerance data from the North Carolina Department of Environment and Natural Resources (NCDENR 2003). The tolerance metrics reported herein were based on averages of regional values reported in Barbour et al. (1999). Tolerances were calculated on the basis of richness (average of tolerance values assigned to each taxon) and density (density-weighted tolerances; Cuffney 2003).

Associations between metrics and explanatory variables were investigated using correlation and regression analysis. Spearman rank correlations were used to reduce the large number of comparisons to a manageable number that could be investigated using regression analysis. The significance and strength of responses were determined in the same manner as for ordinations.

Multimetric analysis.—Invertebrate responses were evaluated using a multimetric response index (B-IBI) that combined all metrics (excluding tolerance-based metrics and diversity measures) that were at least moderately correlated ($|\rho| \geq 0.5$) with urban intensity. The response of each metric was adjusted so that the value of all metrics decreased as urban intensity increased (i.e., $M_{\text{adj}-i} = M_{\max} - M_i$ when $\rho > 0$). The component metrics were range standardized ($[M_i - M_{\min}]/[M_{\max} - M_{\min}] \times 100$) over all sites so that all metrics were equally weighted. The standardized metrics were averaged over all sites, and the resulting values were again range standardized to produce a B-IBI that varied from 100 (minimum urban) to 0 (maximum urban). This is the same procedure that was

TABLE 1. Abbreviations used to identify invertebrate assemblage metrics based on density. Abbreviations for taxonomy-based and functional-group metrics that are based on other units of measurement are designated by appending lowercase letters to these abbreviations: *r*, richness; rp, % richness; *p*, % density. For example, EPT density, % density, richness, and % richness are abbreviated as EPT, EPTp, EPTr, and EPTrp, respectively.

Metric	Abbreviation	Metric	Abbreviation
Taxonomy-based metrics		Taxonomy-based metrics	
Total	RICH	Amphipoda	AMPHI
Ephemeroptera, Plecoptera, Trichoptera	EPT	Isopoda	ISOPOD
Ratio of EPT to Chironomidae	EPT_CH	Oligochaeta	OLIGO
Ephemeroptera	EPEM	Functional-group metrics	
Plecoptera	PLECO	Parasites	PA
Pteronarcidae	PTERY	Predators	PR
Trichoptera	TRICH	Omnivores	OM
Odonata	ODONO	Collector-gatherers	CG
Coleoptera	COLEOP	Filtering-collectors	FC
Diptera	DIP	Scrapers	SC
Chironomidae	CH	Shredders	SH
Orthocladiinae	ORTHO	Piercers	PI
Orthocladiinae/Chironomidae	ORTHO_CH	Dominance metrics	
Tanytarsanii	TANY	Most abundant taxa	DOM1
Tanytarsanii/Chironomidae	TANY_CH	Two most abundant taxa	DOM2
Nonchironomid Diptera	NCHDIP	Three most abundant taxa	DOM3
Noninsects	NONINS	Four most abundant taxa	DOM4
Nonchironomid Diptera and noninsects	ODIPNI	Five most abundant taxa	DOM5
Mollusca and Crustacea	MOLCRU	Tolerance metrics	
Gastropoda	GASTRO	Average tolerance of taxa	TOLr
Bivalvia	BIVALV	Density-weighted tolerance	TOL
Corbicula	CORBIC	Shannon diversity	SHANND

used to calculate the UII (McMahon and Cuffney 2000; Tate et al. 2005). This B-IBI is analogous to other multimetric indices (e.g., IBI, B-IBI, and ICI) that have proven to be valuable tools for assessing biological responses to changes in water quality (Karr 1981; Kerans and Karr 1994; Fore et al. 1996; Barbour et al. 1999; Morley and Karr 2002).

Three versions of the B-IBI were calculated. The "full" B-IBI (B-IBI-f) used all metrics that were at least moderately correlated with UII in a study area. The "common" B-IBI (B-IBI-c) used only those metrics that at least moderately correlated with UII in all three study areas. The "reduced" B-IBI (B-IBI-r) used the "common" model metrics but eliminated metrics that differ only in units of measurement: richness, percent richness, density, or percent density. The common and reduced IBIs were constructed to investigate the possibility of establishing a nationally consistent B-IBI.

Response thresholds.—Locally weighted least squares smoothing (LOWESS) was used to identify possible thresholds (i.e., points of abrupt change) in the invertebrate responses and to identify the approximate UII value that corresponded to the threshold (Coles et al. 2004). Once a potential threshold was identified, a two-slope linear regression analysis was used to determine if the threshold corresponded to a statistically significant ($P \leq 0.05$) change in the rate of response. This involved dividing the data into two subsets associated with the different sides of the threshold. A linear regression then was calculated for each subset of data and the slopes were compared using ANCOVA. LOWESS and regression analyses were done with SYSTAT 9.0 (SPSS 1999).

Urban tolerance values for invertebrates.—Tolerance values specific to urbanization were derived using weighted-averaging (WA) calibration (Juggins 2003) to estimate the optimum urban intensity (CUII or UII) for the occurrence of each taxon. Optima were derived separately for each sample type (QRC, QMH, and RTH) within a study area for taxa that occurred at five or more sites. Tolerance values were calculated by range standardizing the optima (CUII or UII) and multiplying by 10 to produce a tolerance index with a range that matched tolerance values reported in Barbour et al. (1999): 0 (very intolerant) to 10 (very tolerant of urbanization). Tolerance values derived from UII were range standardized separately for each study area. Tolerances based on CUII were range standardized using the range of optima encompassed by the three study areas combined.

Results

Taxonomic Richness and Composition

A total of 423 invertebrate taxa were collected from the three urban study areas (Table 2; Appendix 1). BOS had the most taxa (240) of which 85% were insects, BIR had 208 taxa (86% insects), and SLC had 185 taxa (88% insects). Very few taxa were common to all three study areas (50 of 423), and most taxa (263 of 423) were found in only one of the three study areas. The greatest commonality in taxonomic composition among study areas was in the Chironomidae, where 54% of the taxa were common to all study areas.

Richness metrics at background sites (UII ≤ 10) were substantially higher for BOS than for BIR and SLC, though percent richness metrics were comparable among study areas (Figure 1). Density and percent density metrics were highly variable, particularly in SLC, and did not show consistent differences among study areas (Figure 1; Appendix 2).

Invertebrate Responses Based on Ordination

Most variation in invertebrate assemblages was accounted for in the first two ordination axes (Table 3) for all sample types and study areas. Analyses of the RTH and QRC data gave similar results in terms of the strength of the ordination axes (eigenvalues), amount of assemblage variation explained (13–17%), and correlation with UII. The UII was strongly correlated with the first ordination axis derived from QRC data for each study area and with RTH data from BOS and BIR. In contrast, the second ordination axis derived from SLC RTH data were most strongly correlated with UII and the first axis with elevation (r = 0.77).

The best-fit relation between ordination site scores and UII was linear for both the RTH and QRC data (Figure 2). The UII accounted for a very high proportion of the variability in ordination site scores for BOS (86–87%) and BIR (74–80%) and a modest amount for SLC (45–51%). In all cases the slopes were statistically significant ($P \leq 0.05$, analysis of variance (ANOVA), Zar 1974). The response rates (slopes) were not statistically different among the study areas (ANCOVA, $P > 0.05$) for either RTH or QRC data. Despite large regional differences in environmental conditions and in the composition of the invertebrate assemblages, the rate of change associated with increasing urbanization was the same for all three study areas (common slope = –0.019 for RTH samples and –0.015 for QRC samples).

TABLE 2. Assemblage richness characteristics (number of taxa in QRC samples) summarized by major taxonomic groupings for the Boston (BOS), Birmingham (BIR), and Salt Lake City (SLC) urban study areas.

	Taxa richness				Taxa unique to a study area				Taxa common to all
Taxon	BOS	BIR	SLC	Total	BOS	BIR	SLC	Total	study areas
Cnidaria	1	0	1	1	0	0	0	0	0
Platyhelminthes	1	1	1	1	0	0	0	0	1
Nemertea	1	0	1	1	0	0	0	0	0
Nematoda	1	1	1	1	0	0	0	0	1
Gastropoda	10	9	5	17	5	4	2	11	1
Bivalvia	3	3	1	5	2	2	0	4	1
Annelida	10	7	7	12	3	1	0	4	4
Arachnida	1	1	1	1	0	0	0	0	1
Decapoda	2	2	1	5	2	2	1	5	0
Isopoda	1	2	1	2	0	1	0	1	1
Amphipoda	4	3	3	5	1	1	0	2	2
Collembola	1	1	1	1	0	0	0	0	1
Ephemeroptera	21	28	17	51	9	17	12	38	2
Odonata	18	17	4	30	9	9	3	21	0
Plecoptera	10	4	12	23	7	2	11	20	0
Hemiptera	12	10	6	22	7	5	4	16	0
Megaloptera	4	4	0	5	1	1	0	2	0
Trichoptera	39	25	30	73	22	9	23	54	2
Lepidoptera	1	1	0	2	1	1	0	2	0
Coleoptera	28	25	21	54	11	5	18	34	0
Non-midge Diptera	13	15	24	36	2	7	16	25	5
Chironomidae	58	49	47	75	9	5	10	24	28
Total insects	205	179	162	372	78	61	97	236	38
Total invertebrates	240	208	185	423	91	72	100	263	50

Ordination analysis indicated that invertebrate responses to urbanization can be predicted using UII and that a comprehensive qualitative depiction of the assemblages (QRC) provided a representation of response that was at least as good as an intensive single-habitat quantitative sample (RTH).

Invertebrate Responses Based on Assemblage Metrics

A variety of invertebrate assemblage metrics were strongly ($|\rho| \geq 0.7$) associated with UII for both RTH and QRC data (Appendices 3 and 4). More metrics were strongly correlated with urban intensity in BOS (34 richness, 16 density) than in BIR (15 richness, 2 density) or SLC (3 richness, 1 density). Richness metrics were, in all cases, much more frequently correlated with UII than were metrics based on density. There was little correspondence among the three study areas in terms of the assemblage metrics that were correlated with the UII either when considering all metrics with $|\rho| \geq 0.7$ or the 12 metrics most strongly correlated with urban intensity in each study area (Appendices 3 and 4). Only one metric (richness-based tolerance, TOLr) appeared in the top 12 metrics for each urban study area. This metric was strongly related to the UII in the BOS urban study, but less strongly related in BIR and SLC (Figure 3). There was little correspondence between metrics that were strongly correlated with UII based on RTH and QRC data (i.e., less than half of the 12 metrics most strongly correlated with UII were similar for RTH and QRC data). Generally speaking, metrics based on EPT and its components, Coleoptera, noninsects, oligochaetes, mollusks plus crustaceans, and tolerances were the best indicators of changes in urban intensity.

Effects of urbanization were qualitatively similar when differences between mean values of assemblage metrics from background (UII ≤ 10) and and highly urbanized (UII ≥ 70) sites were compared (Table 4). Total, EPT, Ephemeroptera, Plecoptera, Trichoptera, and Diptera taxa richness decreased in all three urban

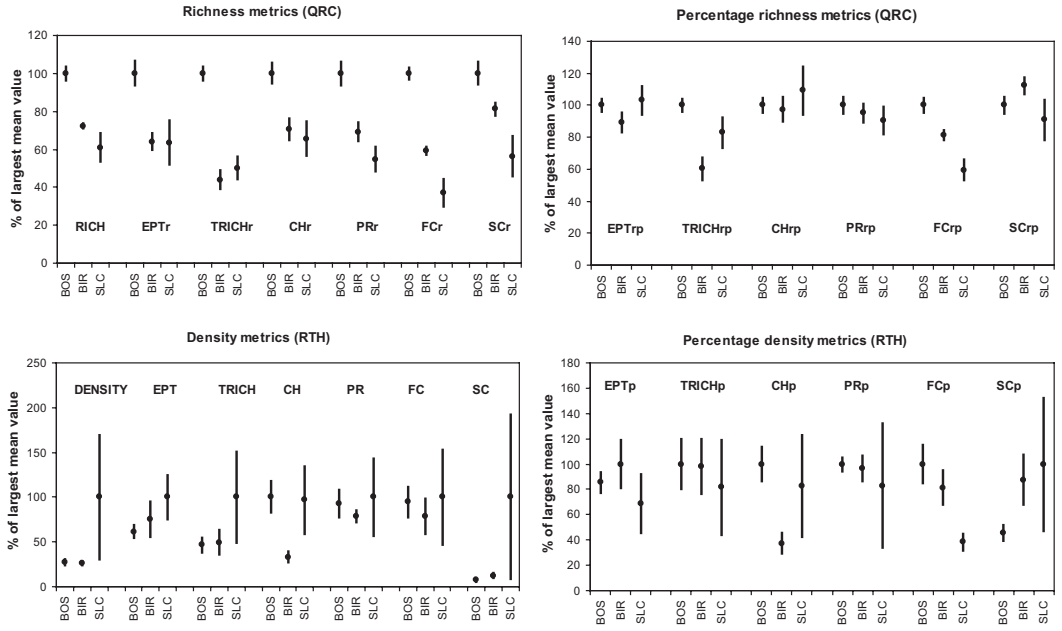

FIGURE 1. Mean values with standard errors for selected metrics at sites with low (UII ≤ 10) urban intensities (background sites) for the Boston (BOS), Birmingham (BIR), and Salt Lake City (SLC) urban study areas. Values are expressed as a percentage of the largest mean associated with the three study areas for each metric. Metric abbreviations are explained in Table 1.

TABLE 3. Correspondence analysis results for invertebrate assemblages and Spearman rank correlations between site scores and the urban intensity index (UII) for the Boston (BOS), Birmingham (BIR), and Salt Lake City (SLC) urban study areas.

Study	Parameter	Axis 1	Axis 2	Axis 3	Axis 4
Quantitative (RTH) samples					
BOS	Eigenvalues	0.272	0.079	0.056	0.037
	Variance explained (%)	17.4	5.0	3.6	2.4
	Correlation with UII	–0.91	–0.29	0.08	–0.04
BIR	Eigenvalues	0.262	0.171	0.085	0.059
	Variance explained (%)	15.5	10.1	5.0	3.5
	Correlation with UII	–0.78	0.01	–0.03	–0.02
SLC	Eigenvalues	0.322	0.307	0.231	0.161
	Variance explained (%)	13.7	13.0	9.8	6.9
	Correlation with UII	–0.12	–0.73	–0.07	–0.05
Qualitative (QRC) samples					
BOS	Eigenvalues	0.242	0.124	0.084	0.080
	Variance explained (%)	14.8	7.6	5.2	4.9
	Correlation with UII	–0.92	–0.11	–0.02	–0.16
BIR	Eigenvalues	0.236	0.164	0.116	0.111
	Variance explained (%)	13.0	9.0	6.3	6.1
	Correlation with UII	–0.84	–0.01	–0.02	0.17
SLC	Eigenvalues	0.259	0.179	0.127	0.112
	Variance explained (%)	13.7	9.4	6.7	5.9
	Correlation with UII	–0.54	–0.29	–0.32	0.12

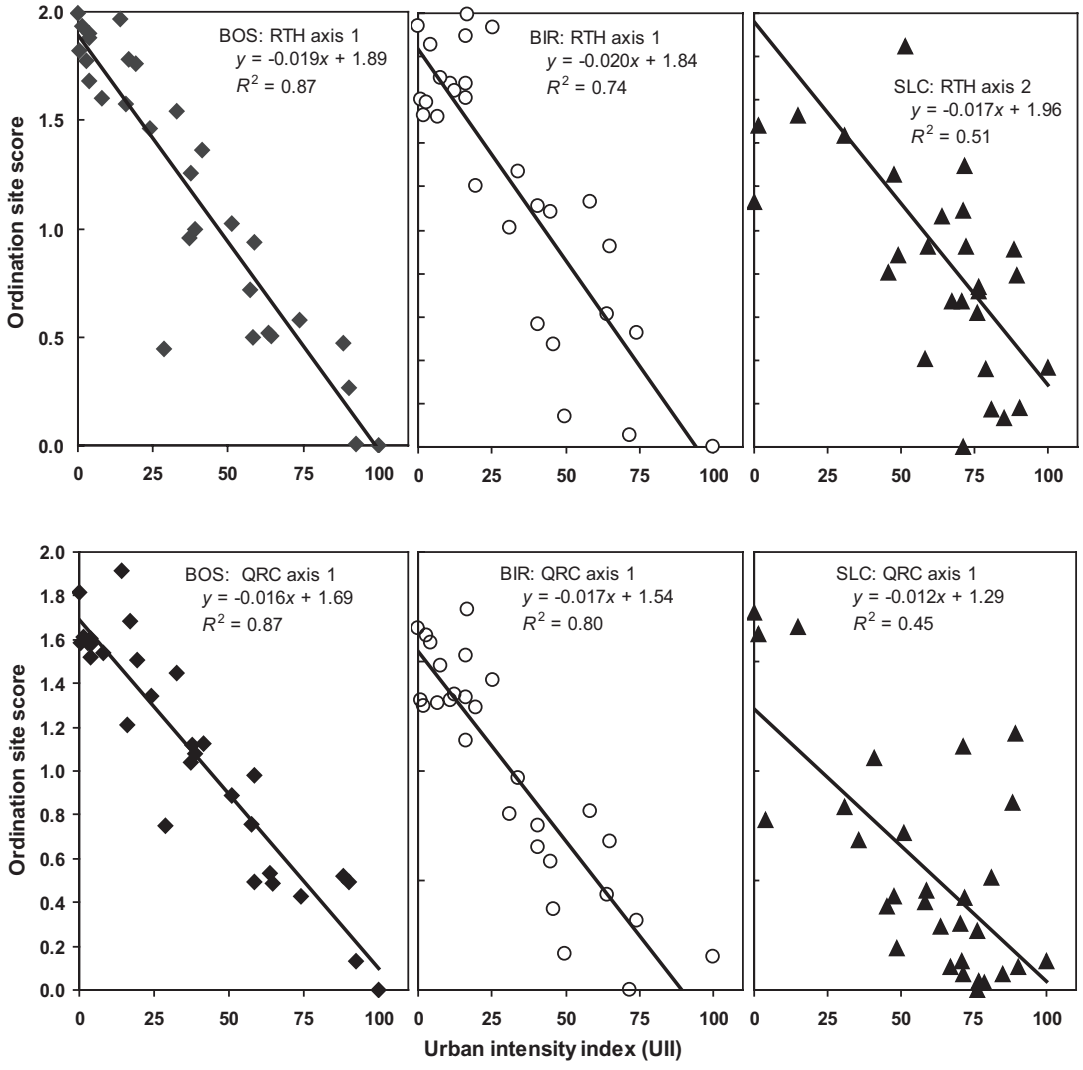

FIGURE 2. Relations between ordination (CA) axis site scores and urban intensity (UII) for assemblages based on RTH and QRC data for the Boston (BOS), Birmingham (BIR), and Salt Lake City (SLC) urban study areas. All slopes are statistically significant ($b \neq 0$, $P < 0.05$).

study areas regardless of whether richness was characterized by QRC or RTH data. Noninsect taxa richness increased with UII in all studies areas. Generally, BOS lost the most taxa over the urban gradient and SLC lost the least. Differences expressed as density were not consistent across the three study areas. Total density at high-intensity urban sites increased in BOS, primarily because of increases in densities of Trichoptera (hydropsychids) and noninsects. Total density decreased in BIR and SLC with SLC exhibiting very large decreases in the density of noninsects, primarily the gastropod family Hydrobiidae.

The B-IBI-f exhibited a very strong response to urban intensity for all study areas and for both RTH and QRC data (Figure 4), and in most cases, the correspondence (R^2) between UII and B-IBI-f was equal to or greater than the maximum observed for any of the component metrics (Figure 5). The advantages attributed to multimetric indices are clearly evident in the B-IBI-f results. The "common" and "reduced" versions of the B-IBI included far fewer metrics (3 quantitative, 14 qualitative, Table 5) than the "full" model (23–68 metrics, Figure 5; Appendices 3 and 4). However, the correlations between the B-IBI and urban

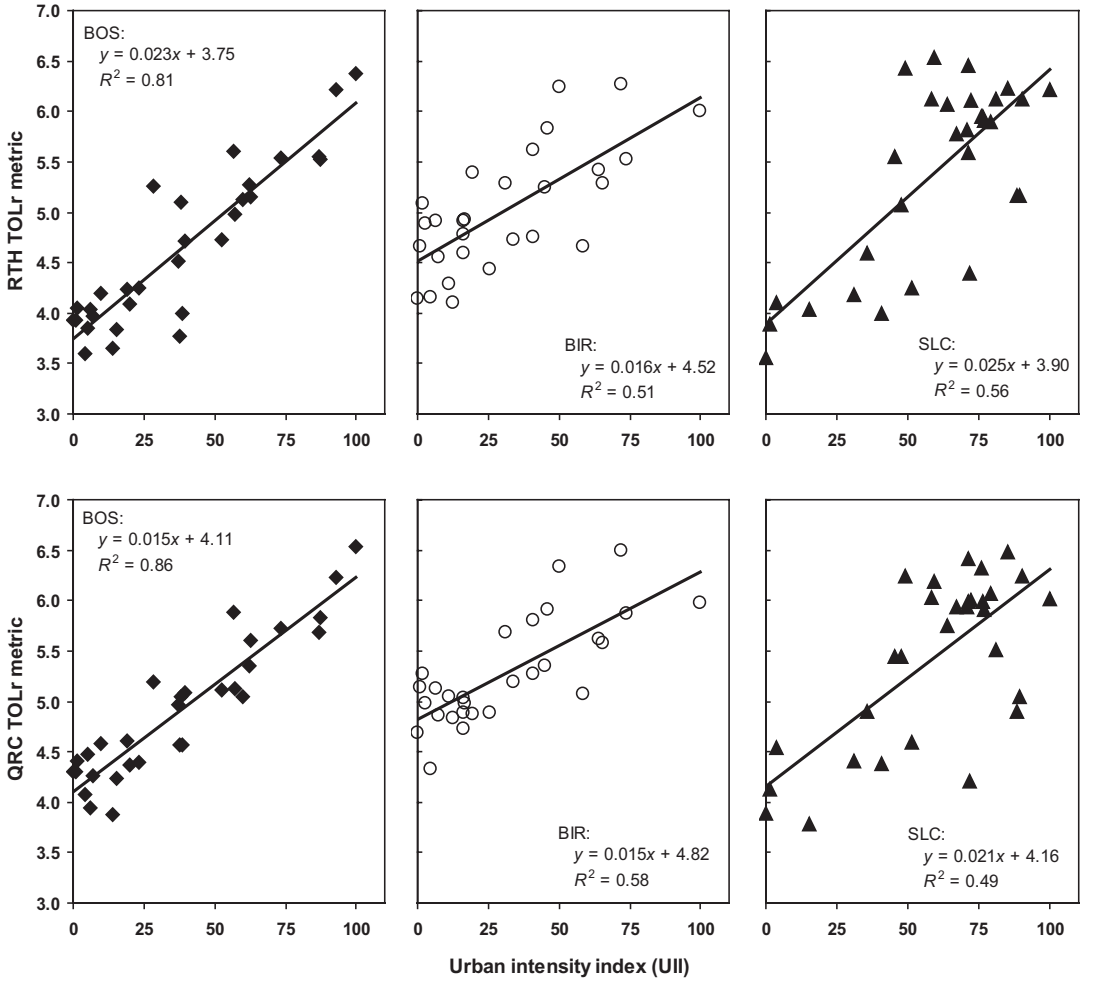

FIGURE 3. Relations between the richness-based tolerance metric (TOLr) and urban intensity (UII) for assemblages based on RTH and QRC samples for the Boston (BOS), Birmingham (BIR), and Salt Lake City (SLC) urban study areas. All slopes are statistically significant ($b \neq 0$, $P < 0.05$).

intensity for the "common" and "reduced" models were, generally, only slightly less than that of the "full" model (Table 6) and were nearly as strong, if not stronger, than the component metric with the best linear fit to urban intensity (Figure 6).

As with ordinations, invertebrate assemblage metrics and B-IBI indices could be predicted using UII. However, the individual metrics that were the best indicators of urbanization varied among study areas. Measures of taxa richness, whether from QRC or RTH data, provided the best indications of response. Quantitative metrics were highly variable and did not give results that were as comparable across study areas.

Response Thresholds

Response thresholds were only observed in the BOS urban study area and only for a few invertebrate metrics (CGr RTH, CGrp RTH, DOM5 RTH) that exhibited exhaustion thresholds at UII levels of about 35–40 (e.g., Figure 7). No assemblage metrics, B-IBIs, or ordinations displayed any initial resistance to urbanization (i.e., no threshold at the low end of the urban gradient) in any of the urban study areas. There was no level of urbanization that did not adversely affect the invertebrate assemblage in any of the three study areas. Invertebrate assemblages began to degrade as soon as the native vegetation, typically forest (BOS, BIR) or shrub land (SLC), began to be replaced with

TABLE 4. Average number of taxa or density lost (negative) or gained (positive values) between background (UII ≤ 10: BOS n = 8, BIR n = 7, SLC n = 3) and high intensity urban (UII ≥ 70: BOS n = 5, BIR n = 3, SLC n = 11) sites for selected taxonomic groups in the Boston (BOS), Birmingham (BIR), and Salt Lake City (SLC) urban study areas. Values in parentheses are percentages of the background values lost (negative) or gained (positive). No odonates were found at SLC background sites, so the percentage departure from background conditions could not be calculated (NA).

Metric	BOS		BIR		SLC	
Richness: Qualitative (QRC) samples						
Total	–34	(–41)	–25	(–42)	–10	(–21)
EPT	–20	(–75)	–14	(–81)	–11	(–66)
Ephemeroptera	–8	(–86)	–9	(–93)	–4	(–62)
Plecoptera	–2	(–63)	–1	(–100)	–3	(–87)
Trichoptera	–10	(–70)	–4	(–59)	–4	(–57)
Odonata	–5	(–79)	–2	(–48)	1	(NA)
Diptera	–12	(–45)	–5	(–24)	–2	(–10)
Chironomidae	–10	(–45)	–2	(–15)	1	(4)
Noninsects	9	(100)	3	(45)	4	(54)
Richness: Quantitative (RTH) samples						
Total	–25	(–52)	–7	(–34)	–8	(–24)
EPT	–13	(–78)	–6	(–67)	–9	(–69)
Ephemeroptera	–4	(–92)	–4	(–100)	–3	(–69)
Plecoptera	–2	(–100)	–1	(–100)	–3	(–90)
Trichoptera	–6	(–66)	–2	(–44)	–3	(–54)
Odonata	–1	(–80)	–1	(–7)	1	(NA)
Diptera	–11	(–58)	–1	(–18)	–1	(–5)
Chironomidae	–9	(–62)	2	(19)	1	(6)
Noninsects	5	(123)	3	(56)	3	(69)
Density: Quantitative (RTH) samples						
Total	8,426	(51)	–3,107	(–39)	–9,853	(–32)
EPT	2,074	(63)	–2,730	(–67)	–93	(–2)
Ephemeroptera	–1,153	(–90)	–1,888	(–88)	–67	(–6)
Plecoptera	–286	(–100)	–59	(–100)	–344	(–75)
Trichoptera	3,513	(202)	–783	(–42)	318	(8)
Odonata	–57	(–91)	14	(144)	44	(NA)
Diptera	1,595	(50)	768	(54)	4,604	(154)
Chironomidae	580	(21)	1,072	(116)	3,780	(141)
Noninsects	1,875	(314)	347	(37)	–13,045	(-64)

roads and buildings. Degradation of the invertebrate assemblage across the urban gradient followed the pattern expected for urban streams; that is, a decrease in insect taxa (particularly EPT taxa) and an increase in the numbers of noninsect taxa and oligochaetes as urban intensity increased.

Interpreting the Urban-Response Gradients

Many explanatory variables (BOS 156, BIR 153, SLC 178) were available to interpret invertebrate responses to increasing urban intensity. When combined with multiple studies, sample types, and response indicators, these explanatory variables constituted an unwieldy array of information. A more manageable subset of variables was derived by selecting the 12 variables that were most strongly correlated with each of three indicators of invertebrate assemblage responses: ordination site scores, richness-based tolerance (TOLr), and B-IBI-f. This reduced the number of explanatory variables to 30 for RTH data (Table 7) and 37 for QRC data (Table 8). Eight of these variables (bold type) were relatively consistent among urban study areas and sample types; the rest were restricted to a specific study area or sample type.

The explanatory variables most commonly associated with changes in invertebrate assemblages were related to land use, land cover, infrastructure, and popu-

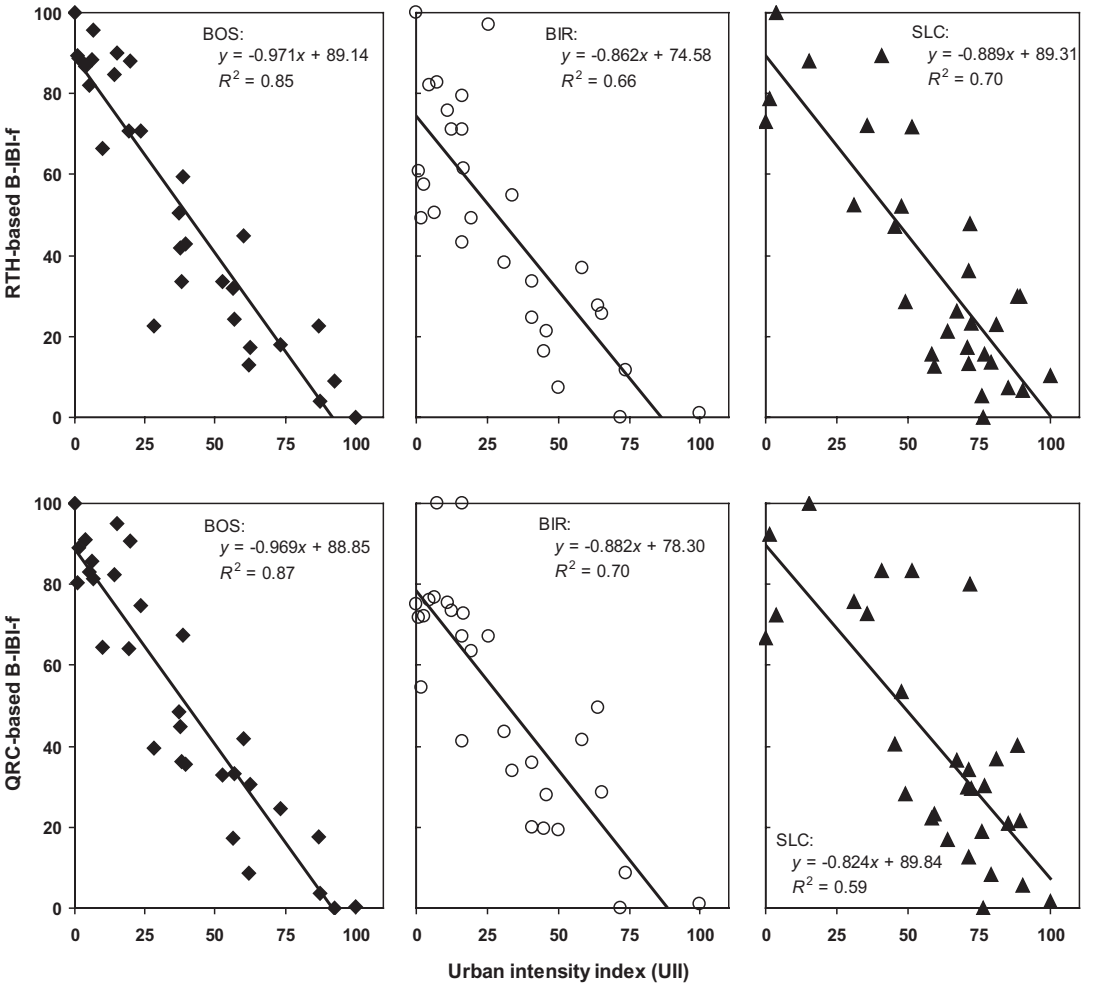

FIGURE 4. Relations between the B-IBI-f and urban intensity (UII) for assemblages based on RTH and QRC samples for the Boston (BOS), Birmingham (BIR), and Salt Lake City (SLC) urban study areas. All slopes are statistically significant ($b \neq 0$, $P < 0.05$).

lation. Urban intensity index, the percentage of developed (urban, low-intensity residential, commercial/industrial/transportation) land in the basin, the percentage of stream buffers in urban lands, road density, and population density (1990, 1999) were the most consistent explanatory variables across all study areas. Elevation, lithology and soils were important only for the SLC study area and then primarily for the qualitative samples. The importance of these factors in SLC is related to the pattern of urban development, which began on the valley floor and has been moving up the mountain benches.

Stream physical habitat variables (Short et al. 2005) were not important explanatory variables relative to land use, and only one variable (number of riffles in the sampling reach) made the top 12 associations and then only for one study area (SLC) and data type (QRC). The lack of correspondence between invertebrate responses and physical habitat structure variables is, in part, a result of our study design, which kept habitat structure consistent within each study area to maximize our ability to detect basin-scale effects versus local-scale effects.

A dozen chemical variables made the list of top explanatory variables. However, there was little consistency among study areas, and the chemical variables that were important tended to be specific to a particular study area. No common set of chemical variables

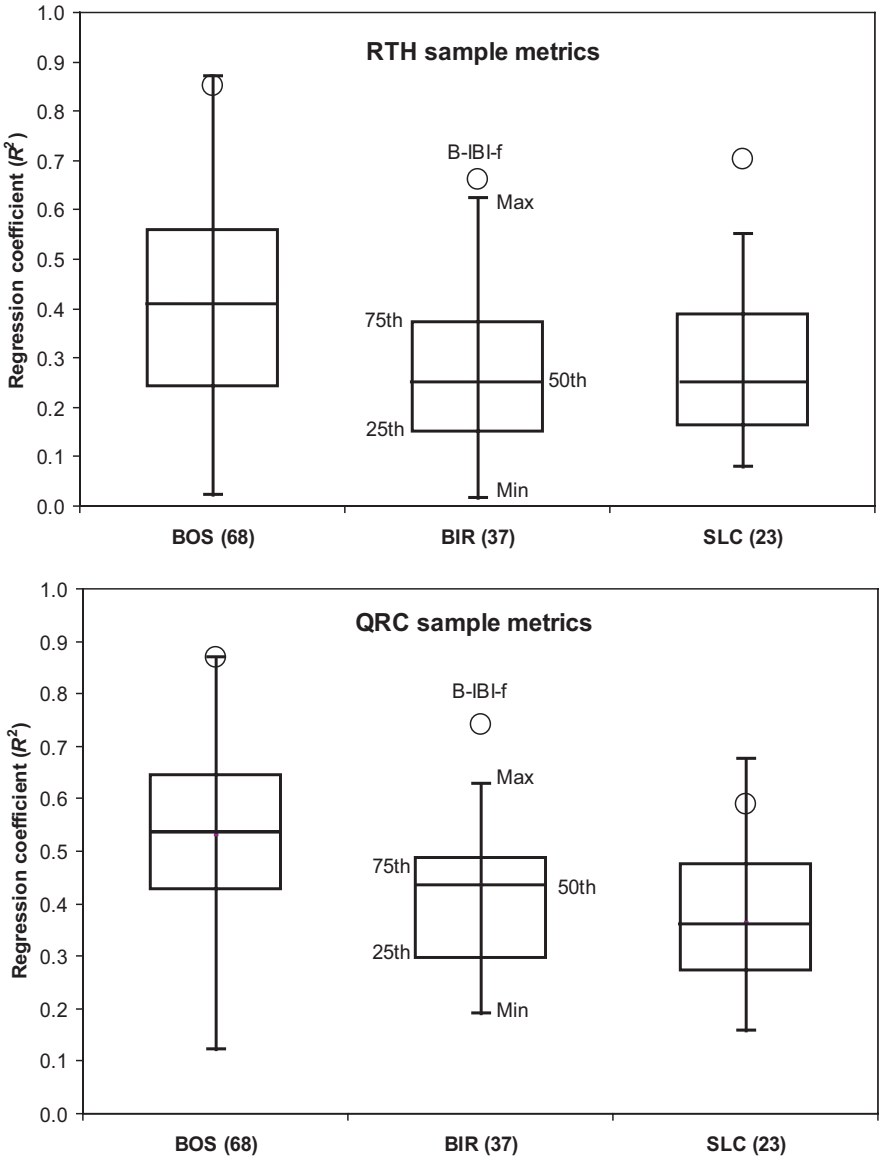

FIGURE 5. Comparison of the strengths (R^2) of the relations between the urban intensity index (UII), B-IBI-f (o), and the invertebrate assemblage metrics comprising the B-IBI-f (box plot) based on RTH and QRC samples. Minimum (min), maximum (max), 25th, 50th, and 75th percentiles are represented in the box plots. The values in parentheses indicate the number of assemblage metrics comprising the B-IBI-f for the Boston (BOS), Birmingham (BIR), and Salt Lake City (SLC) urban studies.

were associated with invertebrate responses across all three urban study areas.

Comparing Responses Among Urban Areas

Invertebrate assemblage responses were compared among study areas using both the UII and the CUII. Relations between response variables (ordination axis site scores and assemblage metrics) and urban intensity (UII and CUII) were evaluated by comparing the slopes of linear regressions. The response variables investigated were limited to those that were at least moderately correlated ($|\rho| \geq 0.5$) with the UII, had statistically significant slopes ($b \neq 0$, $P \leq 0.01$), rela-

TABLE 5. Assemblage metrics that were used in constructing the "common" and "reduced" variations of the B-IBI.

	RTH B-IBI		QRC B-IBI	
Metric	Common	Reduced	Common	Reduced
Richness				
RICH			X	X
PLECOr	X	X		
TRICHr			X	X
EPTr	X	X	X	X
EPT_CHr			X	X
COLEOPr	X	X	X	X
% richness				
PLECOrp	X			
TRICHrp			X	
EPTrp			X	
EPT_CHrp			X	
NONINSrp	X	X	X	X
ODIPNIrp			X	X
OLIGOrp			X	X
Functional group richness				
FCr			X	X
Density				
PLECO	X			
COLEOP	X			
% density				
COLEOPp	X			

TABLE 6. Relations between UII and B-IBIs based on all metrics that were correlated with the UII ($|\rho| \geq 0.5$) for each study (B-IBI-f), on a common set of metrics shared by all three studies (B-IBI-c), and on common metrics with no redundancy (B-IBI-r). Relations are derived from linear regressions: $Y = a + bX$ where Y is the B-IBI and X is the UII. N is the number of metrics used to calculate the B-IBI for the Boston (BOS), Birmhingham (BIR), and Salt Lake City (SLC) urban study areas.

	RTH metrics					QRC metrics			
Model	N	a	b	R^2	Model	N	a	b	R^2
BOS					BOS				
B-IBI-f	68	89.14	–0.97	0.85	B-IBI-f	37	88.85	–0.97	0.87
B-IBI-c	8	74.70	–0.84	0.75	B-IBI-c	12	94.43	–0.93	0.80
B-IBI-r	4	91.40	–1.03	0.85	B-IBI-r	9	94.44	–0.95	0.81
BIR					BIR				
B-IBI-f	37	74.58	–0.86	0.66	B-IBI-f	25	76.10	–0.85	0.74
B-IBI-c	8	49.73	–0.64	0.48	B-IBI-c	12	71.73	–0.79	0.65
B-IBI-r	4	60.31	–0.70	0.56	B-IBI-r	9	77.44	–0.79	0.66
SLC					SLC				
B-IBI-f	23	89.31	–0.89	0.70	B-IBI-f	23	89.84	–0.82	0.59
B-IBI-c	8	78.27	–0.85	0.67	B-IBI-c	12	88.03	–0.81	0.52
B-IBI-r	4	84.01	–0.91	0.63	B-IBI-r	9	86.26	–0.79	0.51

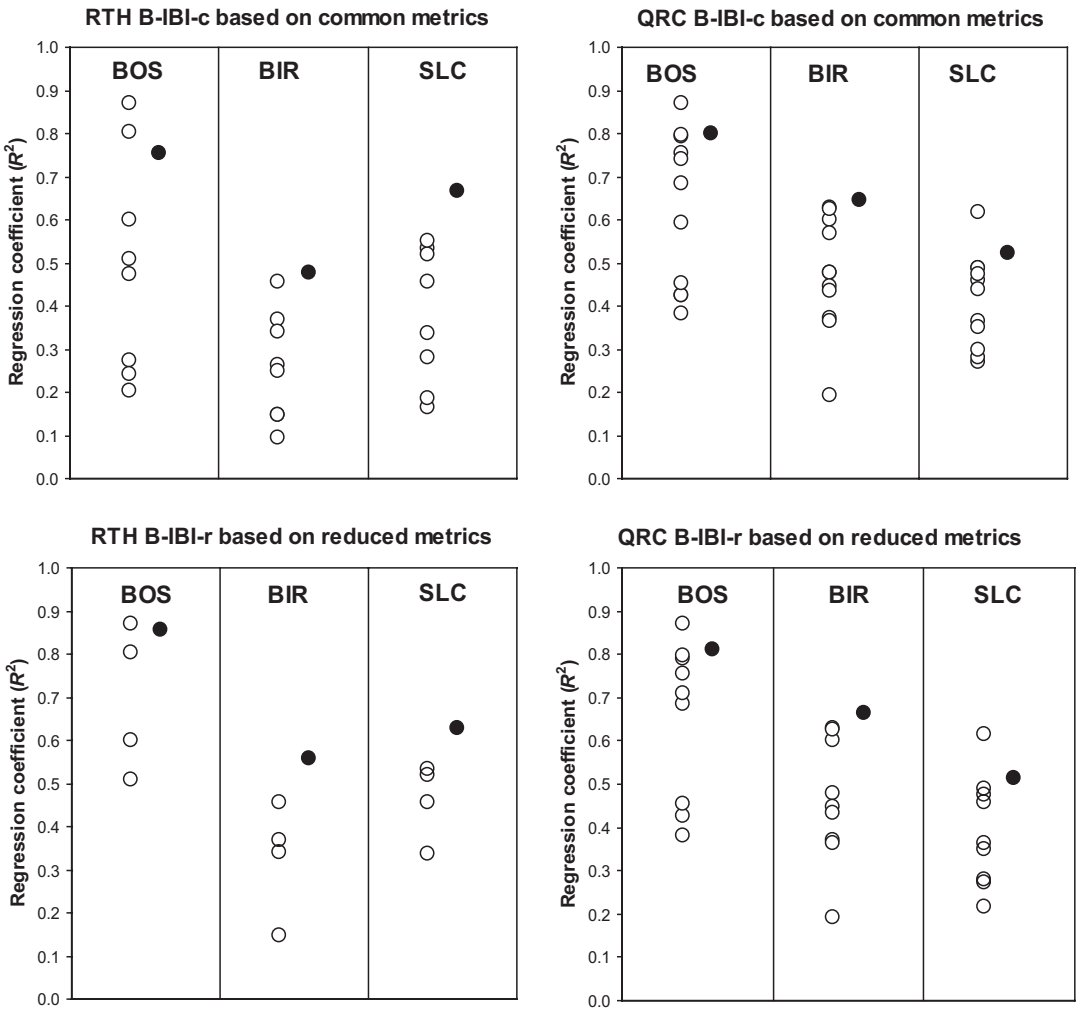

FIGURE 6. Comparison of the strengths (R^2) of the relations between the urban intensity index (UII) and the B-IBI-c, the B-IBI-r (•), and the invertebrate assemblage metrics comprising these indices (o) based on RTH and QRC samples for the Boston (BOS), Birmingham (BIR), and Salt Lake City (SLC) urban study areas.

tively even distributions across the gradient defined by the UII, included no outliers, and were common to all three study areas. Only 12 response variables (3 based on RTH data, 9 based on QRC data) possessed these characteristics.

Response rates (slopes) for most metrics (8 of 12) did not differ significantly ($P \leq 0.05$) among study areas when urban intensity was expressed as UII (Table 9). Four metrics that did differ significantly were all QRC richness metrics with higher response rates in BOS than in BIR or SLC. The significantly higher rates in BOS are a consequence of the higher taxa richness associated with background sites in BOS compared to BIR and SLC (Figure 1). Areas with higher background taxa richness have more taxa that can be lost along the urban gradient. Consequently, the numbers of taxa lost per unit of urban intensity (slope) are higher in these areas (BOS) than in areas with lower background richness (BIR, SLC).

When response rates were based on the CUII most of the metrics (8 of 12) were statistically different among study areas (Table 9). All of these differences were associated with higher response rates in BOS. Response rates in BIR and SLC were not statistically different from one another. The higher response rates in BOS are a result of differences in the maxi-

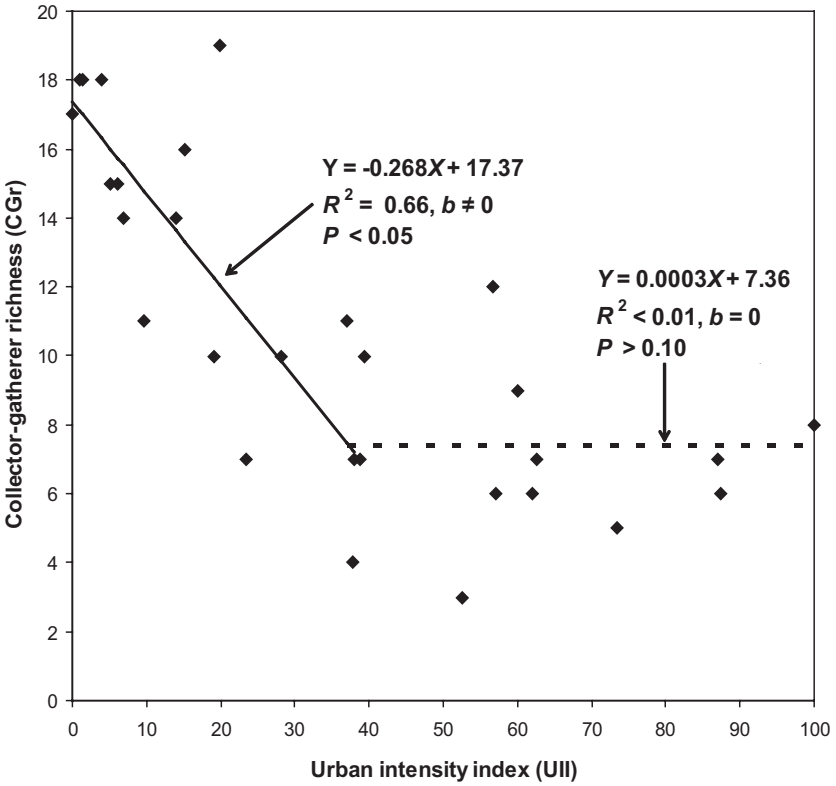

FIGURE 7. Response threshold for collector-gatherer richness in the Boston urban study area derived using a two-step regression analysis.

mum levels of urban intensity measured by CUII and UII, which were substantially different in BOS (CUII = 74, UII = 100) but not in BIR (CUII = 100, UII = 100) or SLC (CUII = 95, UII = 100). These differences are reflected in the relation between CUII and UII (Tate et al. 2005). There is almost a 1:1 relation in BIR (UII = 1.06 × CUII – 1.93, R^2 = 0.97) and SLC (UII = 1.19 × CUII – 5.69, R^2 = 0.86), but in BOS a unit of CUII corresponds to 1.52 units of UII (UII = 1.52 × CUII – 1.90, R^2 = 0.98). Consequently, the rate of change in BOS is substantially higher when expressed as CUII than as UII because the same magnitude of invertebrate response occurs over a smaller range of urban intensity. The average ratio (CUII:UII) of slopes for the response variables in Table 9 (BOS: 1.50, BIR: 1.04, SLC: 1.18) match the slopes of the regressions relating CUII and UII for each study area.

The strong relation between CUII and UII in each urban study area dictates that the correlations between urban intensity and assemblage metrics are virtually the same regardless of whether urban intensity is characterized by CUII or UII. Linear regressions relating correlations between metrics based on CUII and UII show a nearly 1:1 correspondence (BOS: b = 0.99, R^2 = 0.99; BIR: b = 0.97, R^2 = 0.99; SLC: b = 0.94, R^2 = 0.99). Consequently, expressing urban intensity using the CUII had no effect on determining which metrics were most strongly correlated with urbanization. The value of CUII is that it places the response rates of the three study areas on a common basis by using a common set of urban indicator variables.

Urban Optima and Tolerance Values

The inference models used to estimate taxa optima performed well with a close correspondence (R^2) between observed and modeled urban intensity (ca. 0.89 for BOS and BIR and 0.75 for SLC) and root mean-square errors for optima of ca. 7 units of intensity for BOS and BIR and 12 for SLC. There was a high degree of correspondence between the optima derived from UII and CUII (BOS: CUII = 0.60 × UII + 3.21,

TABLE 7. Spearman rank correlations between indicators of invertebrate response and explanatory variables based on RTH samples for the Boston (BOS), Birmingham (BIR), and Salt Lake City (SLC) urban study areas. Only the 12 explanatory variables having the highest correlations (|ρ|) with the ordination axis representing urbanization, richness-based tolerance (TOLr), or B-IBI-f are shown. Explanatory variables in bold type were relatively consistent among study areas and sample types. Stream buffer area is based on a 180 m wide (90 m per side) buffer thoughout the basin.

	BOS			BIR			SLC		
Explanatory variable	Axis 1	TOLr	B-IBI-f	Axis 1	TOLr	B-IBI-f	Axis 2	TOLr	B-IBI-f
Urban intensity (UII) index	0.91	0.85	–0.91	0.78	0.70	–0.81	–0.74	0.60	–0.75
Lithology and soils (% basin area)									
Lake sediment and playa							–0.64	0.58	–0.70
Land use/land cover (% basin area)									
Developed	0.87	0.79	–0.91	0.78	0.70	–0.80	–0.73	0.62	–0.71
Low intensity residential	0.86	0.78	–0.91	0.72	0.63	–0.76		0.62	
High intensity residential				0.75	0.66	–0.77			
Commercial/industrial/ transportation	0.84		–0.88	0.78	0.75	–0.79		0.60	
Forested	–0.90	–0.83	0.89			–0.64			
Mixed				–0.72	–0.59	0.71			
Shrub lands							0.65		
Deciduous							0.67		
Herbaceous planted/cultivated									
Pasture/hay				–0.71		0.72			
Urban/recreational grasses					0.60	–0.72		0.70	
Stream buffers (% buffer area)									
Urban	0.85	0.76	–0.90	0.78	0.70	–0.81	–0.72	0.61	–0.73
Forested	–0.86	–0.79	0.86						
Shrub land							0.72		0.69
Infrastructure									
Roads (km/km^2)	0.89	0.82	–0.92	0.74	0.64	–0.75			
Toxic release inverntory (no./ 100 km^2)		0.80	–0.87						
Sewers (% of households)				0.78		–0.76	–0.67	0.60	
Population and socioeconomic factors									
Population, 1990 (no./km^2)	0.86	0.82	–0.88	0.78	0.65	–0.77	–0.71		–0.67
Population, 1999 (no./km^2)	0.86	0.81	–0.88	0.79	0.65	–0.78	–0.72		–0.69
Population change: 1990–1999 (no./km^2)							–0.65		
Housing units built before 1980 (%)		0.78							
Socioeconomic factor 2 (PCA factor 2)					0.57		–0.74		–0.73
Chemistry									
Alkalinity (mg/L $CaCO_3$)	0.91		–0.89						
Ammonia (mg/L)		0.83							
Ammonia+organic nitrogen (mg/L)		0.76					0.65	0.60	
Chloride (mg/L)				0.70				0.75	–0.70
Conductance (microsiemens/ cm at 25°C)	0.85		–0.84					0.69	–0.75
Magnesium (mg/L)	0.89								–0.69
Sodium (mg/L)								0.70	–0.71
Periphytion									
biomass (g ash free dry mass/m^2)								0.67	–0.78

R^2 = 0.98; BIR: CUII = 0.90 × UII + 3.28, R^2 = 0.99; SLC: CUII = 0.63 × UII + 16.81, R^2 = 0.88). The CUII optima based on QRC data were used to derive invertebrate tolerances (Appendix 5) because the CUII is the most consistent representation of urban intensity across the three study areas and QRC data include more invertebrate taxa than do either QMH or RTH. Tolerances derived from QMH and RTH samples were

TABLE 8. Spearman rank correlations between indicators of invertebrate response and explanatory variables based on QRC data for the Boston (BOS), Birmingham (BIR), and Salt Lake City (SLC) urban study areas. Only the 12 explanatory variables having the highest correlations (|ρ|) with the ordination axis representing urbanization, richness-based tolerance (TOLr), or B-IBI-f are shown. Explanatory variables in bold type were relatively consistent among study areas and sample types. Stream buffer area is based on a 180-m-wide (90 m per side) buffer thoughout the basin.

	BOS			BIR			SLC		
Explanatory variable	Axis 1	TOLr	B-IBI-f	Axis 1	TOLr	B-IBI-f	Axis 2	TOLr	B-IBI-f
Urban intensity (UII) index	0.92	0.88	–0.91	0.84	0.74	–0.83	–0.53	0.58	–0.70
Elevation (m)									
Site							0.77	–0.62	
Mean in basin							0.62		
Range in basin							–0.72	0.62	
Lithology and soils (% basin area)									
Lake sediment and playa							–0.63	0.63	–0.65
Soil: proportion of sand							–0.52		
Land use/land cover (% basin area)									
Developed	0.88	0.86	–0.91	0.83	0.75	–0.82		0.60	–0.68
Low intensity residential	0.87	0.85	–0.90	0.78	0.72	–0.77			
High intensity residential				0.82	0.75	–0.81			
Commercial/industrial/ transportation		0.83	–0.87	0.83	0.74	–0.84			
Forested	–0.90	–0.86	0.91	–0.75		0.68			
Mixed				–0.76	–0.75				
Herbaceous planted/cultivated									
Pasture/hay					–0.71	0.72			
Urban/recreational grasses				0.78	0.69	–0.75			
Stream buffers (% buffer area)									
Urban	0.85	0.83	–0.88	0.84	0.74	–0.84			–0.68
Forested	–0.87	–0.81	0.86				0.52		
Infrastructure									
Roads (km/km²)	0.89	0.87	–0.91	0.79	0.68	–0.80			
Toxic release inverntory (no./ 100 km²)	0.90	0.88	–0.90						
Sewers (% households)				0.82	0.75	–0.75			
Population and socioeconomic factors									
Population, 1990 (no./km²)	0.87	0.86	–0.89	0.84	0.68	–0.82			–0.63
Population, 1999 (no./km²)	0.87	0.86	–0.90	0.84	0.67	–0.81			–0.64
Population change: 1990–1999 (no./km²)			–0.84						
Housing units built before 1980 (%)	0.83								
Socioeconomic factor 2 (PCA factor 2)									–0.68
Physical habitat structure and chemistry									
Number of riffles							0.71	–0.57	
Alkalinity (mg/L $CaCO_3$)	0.85	0.85	–0.85						
Ammonia (mg/L)	0.83								
Ammonia+organic nitrogen (mg/L)		0.83					–0.68	0.63	
Chloride (mg/L)							–0.53	0.74	–0.69
Conductance (micro-siemens/cm at 25°C)								0.68	–0.77
Magnesium (mg/L)									–0.72
Phosphorus (mg/L)							–0.67		

TABLE 8. Continued.

	BOS			BIR			SLC		
Explanatory variable	Axis 1	TOLr	B-IBI-f	Axis 1	TOLr	B-IBI-f	Axis 2	TOLr	B-IBI-f
Sodium (mg/L)							–0.59	0.76	–0.69
Sulphate (mg/L)									–0.67
Turbidity (NTU)								0.66	
Temperature (°C)							–0.81	0.57	
Periphyton biomass (g ash free dry mass/m^2)								0.65	–0.77

TABLE 9. Comparison of slopes (b) and regression coefficients (R^2) for linear regressions relating the two urban intensity indices (UII and CUII) to selected indicators of invertebrate response for the Boston (BOS), Birmingham (BIR), and Salt Lake City (SLC) urban study areas. All slopes are statistically significant ($b \neq 0$, $P \leq 0.01$). Slopes of study areas connected by a line are not statistically different from one another ($P \leq 0.05$). Metric abbreviations are explained in Table 1.

Metrics	Regressions with UII			Regressions with CUII		
Quantitative (RTH) sample metrics						
Ordination axis	SLC	BOS	BIR	SLC	BIR	BOS
b, R^2	–0.017, 0.51	–0.019, 0.87	–0.020, 0.74	–0.019, 0.42	–0.021, 0.71	–0.029, 0.77
Tolerance (TOLr)	BIR	BOS	SLC	BIR	SLC	BOS
b, R^2	0.016, 0.51	0.023, 0.81	0.025, 0.56	0.017, 0.50	0.029, 0.44	0.034, 0.72
B-IBI-f	BIR	SLC	BOS	BIR	SLC	BOS
b, R^2	–0.862, 0.66	–0.889, 0.70	–0.971, 0.85	–0.895, 0.61	–1.038, 0.58	–1.443, 0.79
Qualitative (QRC) data metrics						
Ordination axis	SLC	BOS	BIR	SLC	BIR	BOS
b, R^2	–0.012, 0.45	–0.016, 0.87	–0.017, 0.80	–0.013, 0.30	–0.018, 0.76	–0.024, 0.80
Richness metrics						
RICH	SLC	BIR	BOS	SLC	BIR	BOS
b, R^2	–0.183, 0.27	–0.266,0.60	–0.459, 0.68	–0.256, 0.32	–0.285, 0.59	–0.693, 0.66
EPTr	BIR	SLC	BOS	BIR	SLC	BOS
b, R^2	–0.168, 0.63	–0.169, 0.46	–0.268, 0.75	–0.174, 0.58	–0.204, 0.40	–0.400, 0.71
TRICHr	BIR	SLC	BOS	BIR	SLC	BOS
b, R^2	–0.058, 0.37	–0.077, 0.36	–0.131, 0.79	–0.059, 0.33	–0.098, 0.36	–0.191, 0.72
% richness metrics						
EPTrp	BIR	BOS	SLC	BIR	SLC	BOS
b, R^2	–0.242, 0.57	–0.245, 0.74	–0.268, 0.44	–0.249, 0.52	–0.310, 0.36	–0.369, 0.71
EPT_CHrp	BOS	SLC	BIR	BIR	SLC	BIR
b, R^2	–0.009, 0.42	–0.011, 0.49	–0.013, 0.48	–0.012, 0.40	–0.013, 0.43	–0.014, 0.44
NONINSrp	BIR	SLC	BOS	BIR	SLC	BOS
b, R^2	0.162, 0.45	0.194, 0.48	0.309, 0.87	0.174, 0.45	0.219, 0.37	0.455, 0.79
Tolerance (TOLr)	BIR	SLC	BOS	BIR	SLC	BOS
b, R^2	0.015, 0.58	0.021, 0.49	0.021, 0.86	0.015, 0.53	0.023, 0.35	0.032, 0.80
B-IBI-f	SLC	BIR	BOS	BIR	SLC	BOS
b, R^2	–0.824, 0.59	–0.882, 0.70	–0.969, 0.87	–0.910, 0.65	–0.988, 0.52	–1.427, 0.79

strongly related (R^2 = 0.80–0.93) to QRC tolerances except for tolerances derived from RTH samples in SLC (R^2 = 0.52). Tolerance values derived from CUII (Appendix 5) were not closely associated (R^2: BOS 0.43, BIR 0.18, SLC 0.42) with tolerances reported by Barbour et al. (1999). However, despite the lack of correspondence at the taxa level, there was a strong relation between the mean of the tolerances for taxa at a site based on CUII optima and the tolerance metric (TOLr) derived from tolerances reported in Barbour et al. (1999; Figure 8A). These relations differed by study area with SLC assemblages showing much higher mean tolerances than BOS and BIR. There was little overlap between sites in SLC and those in BIR and BOS regardless of whether tolerances were based on CUII or literature values (TOLr). The similarity in site characterization based on CUII and literature (Barbour et al. 1999) tolerances (TOLr) indicate that the differences between SLC and BOS and BIR are real and not artifacts associated with the derivation of CUII, species optima, and tolerances. The cumulative distribution of CUII tolerance values within each study area (Figure 8B) also indicates that distributions of tolerances in BOS and BIR are similar to one another, but in SLC, the tolerances are skewed toward the high urban tolerances. Approximately 90% of the taxa in BOS and BIR had tolerance values of 5 or less, whereas only 31% of the taxa in SLC had optima below 5. These data imply that the invertebrate assemblages from SLC are more tolerant to urbanization than are assemblages from BOS or BIR.

Discussion

The responses of invertebrate assemblages observed in BOS, BIR, and SLC were generally consistent with those observed in other studies of urbanization. Taxa richness and many of its components (e.g., EPTr, EPEMr, PLECOr) decreased as urbanization increased (Garie and McIntosh 1986; Kennen 1999; Paul and Meyer 2001; Huryn et al. 2002; Kennen and Ayers 2002; Roy et al. 2003). B-IBI decreased and TOLr increased as urban intensity increased, consistent with other urban studies (May et al. 1997; Paul and Meyer 2001; Morley and Karr 2002; Roy et al. 2003). However, many studies have shown that diversity index values decreased with increasing urbanization (Klein 1979; Pratt et al. 1981; Duda et al. 1982; Whiting and Clifford 1983; Pedersen and Perkins 1986; Jones and Clark 1987; Schueler and Galli 1992; Shaver et al. 1995; Paul and Meyer 2001). This finding was not consistent with our results. Only BOS had a statistically significant ($P \leq 0.05$) decrease in diversity with increasing urbanization. Neither the BIR nor SLC urban study areas had any statistically ($P > 0.05$) meaningful relations between diversity and urban intensity.

Assemblage characteristics based on qualitative measures (i.e., taxa richness) had better and more consistent relations with urbanization than did quantitative measures (i.e., density) in all three urban study areas regardless of whether responses were assessed based on assemblage metrics or ordinations. Results of several other studies (Garie and McIntosh 1986; Huryn et al. 2002) also indicate that quantitative measures (e.g., density and biomass) are not as closely associated with changes in urbanization as are measures of taxa richness. This is likely a result of adding the errors associated with estimating density and biomass to that associated with detecting the taxon, which increases total variability and makes it more difficult to discern associations with urbanization. Our data (Figure 1) show that quantitative metrics are much more variable than are qualitative metrics. This may account for the generally poor performance of quantitative metrics. Quantitative samples also present numerous compromises when attempting to resolve taxonomic ambiguities (Cuffney 2003); the method used to do this can strongly affect both richness and abundance metrics and associations with environmental variables. Given the generally poor results observed with quantitative metrics, additional costs associated with generating these metrics may not be justified because they do not appear to increase our ability to detect responses along the urban gradient.

Effect thresholds for invertebrate assemblages at 5–18% total impervious surface area have been reported previously (Klein 1979; Jones and Clark 1987; Schueler 1994; Shaver et al. 1995; Booth and Jackson 1997; Maxted and Shaver 1997; May et al. 1997; Kennen and Ayers 2002; Morse et al. 2003). However, the results of our urban studies did not indicate that an effect threshold exists at low levels of urbanization. That is, the assemblages did not show any evidence of being able to resist or compensate for changes brought about during the initial phases of urbanization. Instead, responses can best be described as linear, with degradation of the invertebrate assemblage beginning as soon as the native vegetation begins to be replaced with roads and buildings. Our data provided no evidence to suggest that there is a level of urban intensity that has no effect on invertebrate assemblages. Thresholds at higher levels of urban intensity (Figure 7) also are rare for invertebrates. Only three high-level (exhaustion) thresholds were evident in more than

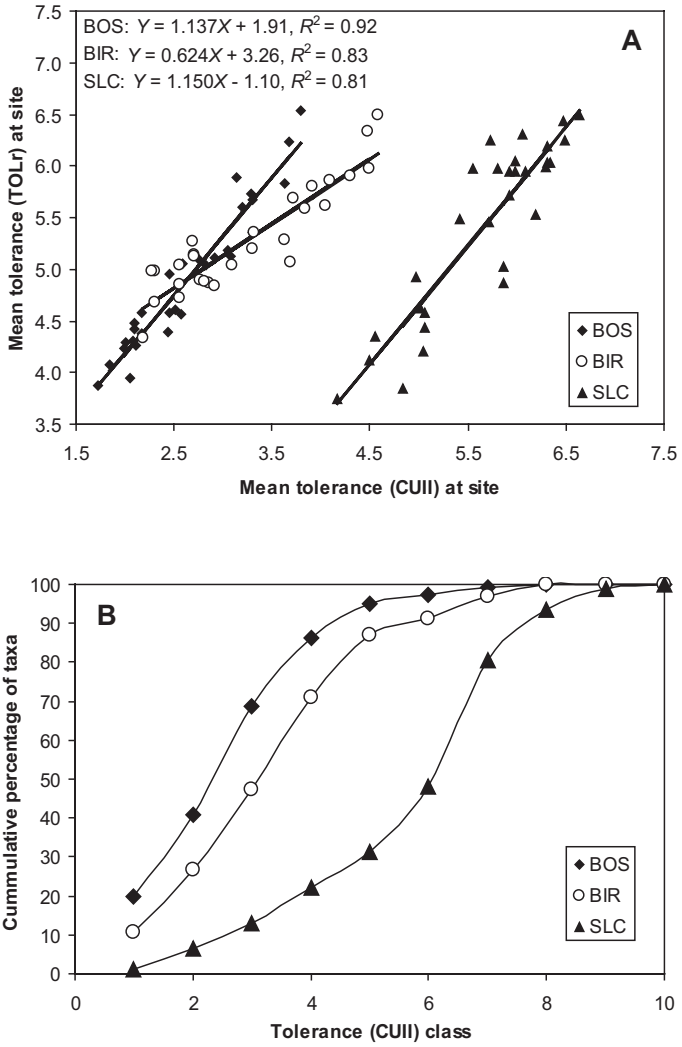

FIGURE 8. Relation between mean tolerance of invertebrates at a site based on literature values (TOLr) and tolerances derived from weighted-average calibration using CUII (A). All slopes are statistically significant ($b \neq 0$, $P < 0.05$). The cumulative distribution of CUII tolerances across 10 tolerance classes are compared for the Boston (BOS), Birmingham (BIR), and Salt Lake City (SLC) urban study areas (B).

400 responses (metrics, indices, ordinations) examined, and all of these occurred in BOS. Consequently, response thresholds cannot be described as a common feature of invertebrate responses to urbanization.

Four impairment categories for invertebrate assemblages have been identified (Schueler 1994) based on percent total impervious surface area (PTIA): unaffected at ≤ 5%, stressed at levels of 5–10%, impacted at levels of 11–25%, and degraded at levels above 26%. The corresponding impairment categories based on UII are unaffected—BOS≤ 11, BIR ≤ 8, and SLC ≤ 5; stressed—BOS: 11–22, BIR: 8–17, and SLC: 5–17; impacted—BOS 22–56, BIR 17–45, and SLC 17–52; and degraded—BOS > 56, BIR > 45, and SLC > 52. These impairment categories indicate that assemblages in BOS, BIR, and SLC are degraded over about half of the urban gradient, impacted in the first quarter of the gradient, and stressed in the first 5–10% of the urban gradient. Only sites at the very low end of the urban gradient are unimpaired. If we accept Schueler's categories, about 67% of our study basins are in the

impaired or degraded classifications and only 15% can be classified as unimpaired.

There was a strong commonality among study areas in terms of the environmental factors that are important in driving changes in invertebrate assemblages. In each area, the amount of developed land, land devoted to commercial/industrial/transportation activities, stream buffers in urban land use, road density, and population density are the paramount factors driving changes in the invertebrate assemblages. At a gross level, the responses of invertebrate assemblages to urbanization also are the same with a continual loss of taxa richness as urban-intensity increases. However, the components of the invertebrate assemblages that are most strongly affected by urbanization and that best serve as indicators of effect differed substantially among the three urban areas in our study. These differences arise because, while the driving factors associated with urbanization are consistent among regions, these factors are acting on an ecological template (i.e., climate, topography, geology, chemistry, habitat, and biology) that differs substantially among regions. It is this variation in the underlying ecological template that results in the variation in assemblage responses and lack of commonality in the biological indicators of urbanization.

Despite the lack of commonality in the response of assemblage metrics, we were able to extract indices (B-IBI-c and B-IBI-r) that were representative of the response of the invertebrate assemblages in each study area, based on a common set of metrics, and comparable among study areas. These indices suggest that it may be possible to derive nationally consistent IBIs that can be used to compare urban responses across the United States. The B-IBI-r was particularly interesting because this index was based on as few as four metrics (RTH), yet it was almost as strongly related to UII as were IBIs based on a much larger number of metrics (e.g., 20–68). Tolerance metrics (TOLr and the CUII-based tolerances derived from WA calibration) and ordinations provided another set of assemblage characteristics that were good indicators of urbanization and that were consistent and comparable among studies. The tolerances values that were derived by WA calibration varied by study area (Appendix 5), and there was relatively low correspondence among tolerance values derived for the same taxon in different study areas. This is consistent with tolerance values reported in Barbour et al. (1999), which also varied by region. Even though there was little correspondence between the tolerances derived from WA calibration and those reported in Barbour et al. (1999), site characterizations based on average taxa tolerances at a site using these two methods were strongly related. This indicates that, while these tolerances differ at the level of the taxon, they both provide a meaningful representation of the assemblage response.

Ordinations provided valuable insights into assemblage responses to urbanization and were critical in the development of urban tolerances. However, the strengths (eigenvalues) of the ordinations were less than anticipated given that the study design was based on an urban intensity gradient. A strong eigenvalue occurs when there is a continual replacement of species along the environmental gradient (McCune et al. 2002) as conditions become more favorable for some species and less favorable for others. Data from our urban studies indicate that this is not the pattern associated with urban gradients. Rather than a continual replacement of taxa along the gradient, there is a continual loss of taxa richness with little replacement by new taxa. Data from BOS (Figure 9) illustrate the steady loss of taxa richness across the urban intensity gradient and very low recruitment of new taxa.

Further insight into the pattern of taxa distribution across the urban gradient in BOS was obtained by modeling the distribution of taxa along the primary (i.e., urban) ordination axis extracted from qualitative (QRC) data using CA. These models were developed by calculating Gaussian response curves for each taxon based on the optima and tolerances (i.e., the variance of the optima; ter Braak 1996) obtained from CA:

$$\text{Relative occurrence} \quad c_k \quad \frac{x_i \quad {}_k^{\,2}}{2t_k^2} \quad \text{(ter Braak 1996)}$$

where $_k$ optima for taxon k

x_i value of urbanization at site i

t_k tolerance for taxon k

c_k maximum of curve for taxon k (set to 100)

Displaying individual response curves for all taxa was not feasible, so the 140 response curves were reduced to 12 by ordering taxa from lowest to highest optima and averaging response curves for groups of 11 taxa, except for the last curve, which was based on 8 taxa. Response curves were converted to units of UII using the relation between taxa optima (taxa ordination scores) and taxa tolerances (i.e., WA-derived tolerances) derived from UII ($Y = 10.983x + 30.481$, $R^2 = 0.95$).

The modeled responses do not show a consistent replacement of taxa over the gradient. Instead, most of

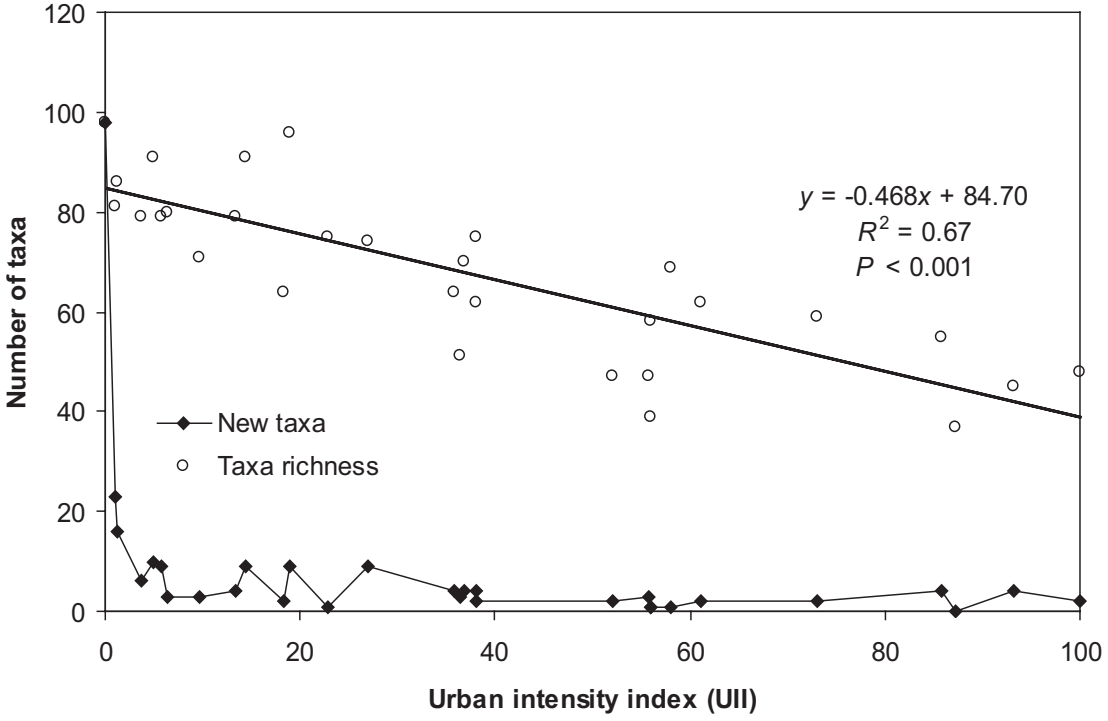

FIGURE 9. Taxa richness and number of new taxa encountered at sites along the urban gradient for the Boston urban study area.

the taxa drop out completely by UII = 55, and there are relatively few taxa that have optima above this level of urban intensity (Figure 10). This type of response is more typical of exposure to a toxicant where there is a loss of taxa over the gradient with no recruitment of new taxa, and all taxa that occur at the high end of the gradient are very tolerant and occur over a very large range of the gradient (i.e., the t_k is very large relative to m_k). Unfortunately, the chemical data collected as part of these studies are not sufficiently rigorous to determine if chemical toxicity was responsible for this distribution pattern. However, toxicants (e.g., pesticides, metals, polyaromatic hydrocarbons, and other organic compounds) are known to be common occurrences in urban runoff (Seaburn 1969; Andrea et al. 1997; Sansalone and Buchberger 1997a, 1997b; Sansalone et al. 1998; Hoffman et al. 2000; Beasley and Kneale 2002). The taxa that comprise the response curves with the highest optima are generally noninsect taxa (Turbellaria, Megadrile, Erpobdellidae, *Glossiphonia complanata*, *Physella* sp., *Laevapex* sp., *Musculium, Caecidotea* sp., and *Gammarus* sp.), with the exception of one species of elimid beetle *Ancyronyx variegate*. In contrast, the taxa that comprise the response curves with the lowest optima are all insect taxa that typically are considered to be forms that are intolerant of pollutants (*Epeorus* sp., *Stenonema vicarium*, *Paraleptophlebia* sp., *Eurylophella* sp., *Psilotreta labida*, *Helicopsyche borealis*, *Rhyacophila fuscula*, *Ectopria* sp., *Hexatoma* sp., *Atherix lantha*, *Stempellina* sp., *Stempellinella* sp., *Parachaetocladius* sp., and *Hagenius brevistylus*).

The very strong linear relations (R^2 = 0.6–0.9) that we detected between assemblage responses and urban intensity are one of the most important results of our studies. Other studies (May et al. 1997; Morley and Karr 2002; Morse et al. 2003; Roy et al. 2003) have identified significant relations between invertebrate responses and urban intensities, but generally these relations were not as strong, or were not linear, or required many explanatory variables (i.e., multiple regression) to form a strong relation. The success of our studies lies in several aspects of our study and program design. Perhaps most important was our effort to control for natural sources of variability by dividing candidate sites into relatively homogeneous environmental settings (Appendix 2). The criteria used to select sampling reaches also helped by reducing local-scale sources

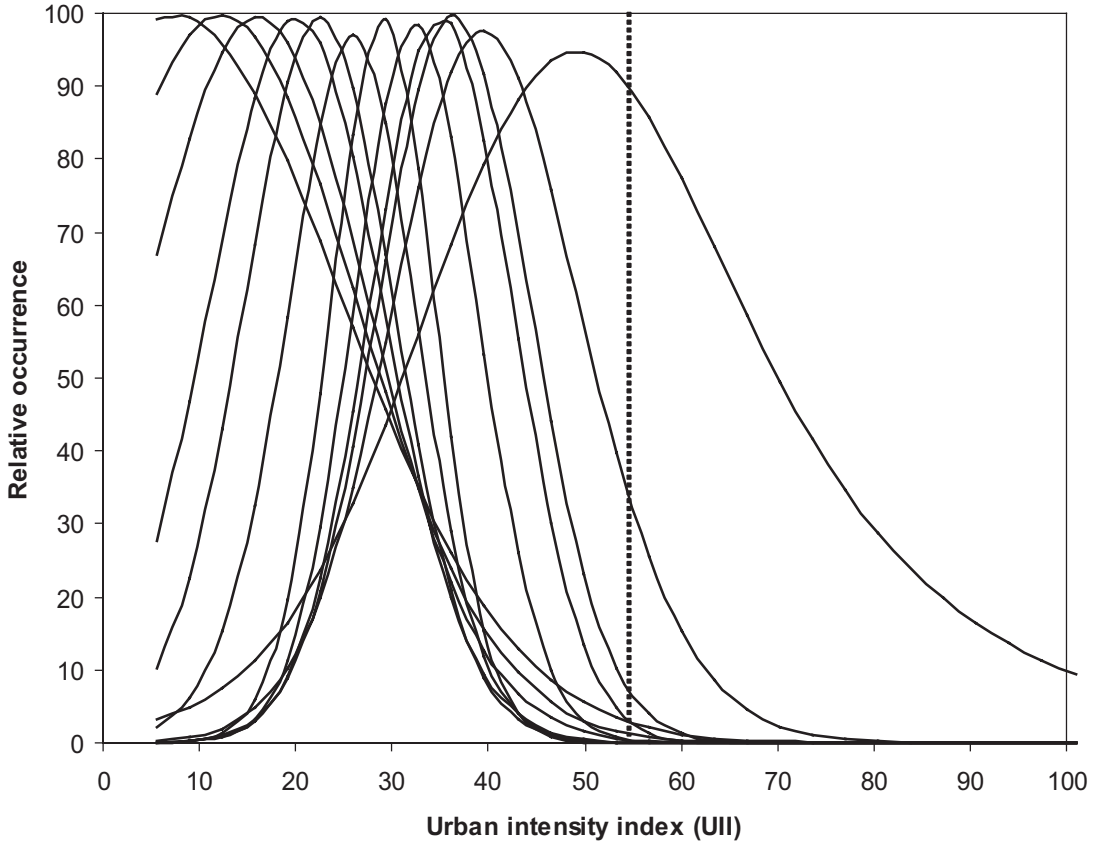

FIGURE 10. The distribution of taxa over the urban gradient modeled from Gaussian response curves based on QRC data for the Boston urban study area.

of disturbance, which allowed us to focus on basin-scale disturbances. While this enhanced our ability to detect the effects of urbanization, it may have reduced the importance of instream physical habitat variables in favor of land-use variables. The use of a simple multimetric index of urban intensity (UII) also was important because it allowed us to achieve a relatively even distribution of sites along the urban intensity gradient. Also important was our ability to use standard sample collection and processing protocols that allowed direct comparisons of invertebrate assemblages among urban areas. The NAWQA Program invertebrate collection (Cuffney et al. 1993) and processing (Moulton et al. 2000) methods are designed to provide a comprehensive characterization of invertebrates in the sampling reach. These methods are somewhat unusual in that samples are collected from a fairly large area of the sampling reach, which results in large numbers of taxa. This enhanced our ability to measure change (Cao et al. 2002a, 2002b).

We can illustrate how sampling area affects the ability to detect change along a gradient using three hypothetical sampling scenarios and the EPT taxa richness data from BOS (Figure 11). Sampling scenario 1 corresponds to the actual EPT taxa richness collected from BOS. Scenarios 2 and 3 represent sampling methods (e.g., smaller sampling areas) that produce one-half and one-quarter, respectively, of the taxa encountered in scenario 1. The response along the urban gradient is simulated using two methods. The first method (Figure 11A) sets the number of EPT taxa collected at each site to a constant proportion of the richness originally collected (i.e., 1, 0.5, and 0.25). The second method (Figure 11B) models the rate of EPT loss across the gradient as a constant regardless of how many taxa are collected by the sampling method. In the first method, the slopes and intercepts of the regression lines change as the initial number of taxa collected decreases from scenario 1–3 and our ability to detect change decreases as the magnitude of change

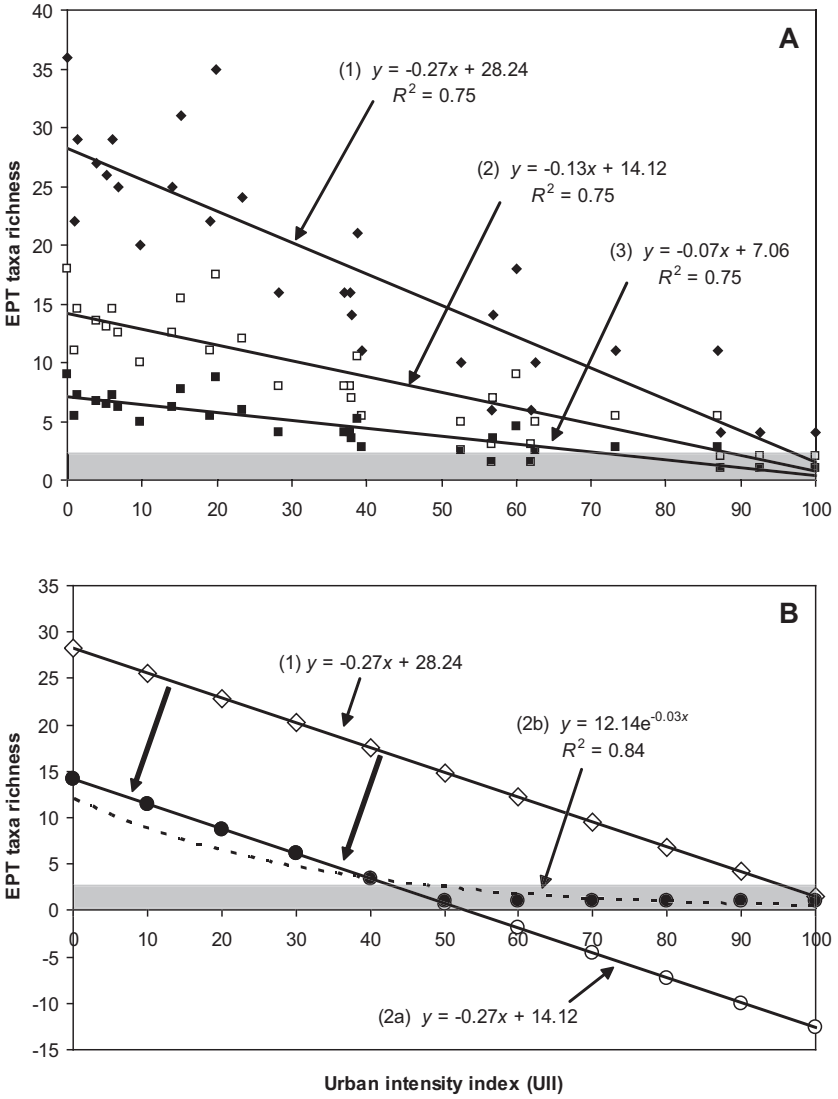

FIGURE 11. Three hypothetical sampling methods that sample progressively smaller proportions of the initial EPT taxa richness: (1) 1.0×, (2) 0.5×, and (3) 0.25×. The first scenario (A) simulates the recovery of a constant proportion of EPT taxa richness over the gradient, and the second scenario (B) simulates a constant rate of EPT taxa loss over the gradient.

in EPT taxa over the gradient (i.e., slope of the response line) decreases. In other words, it is much easier to detect the loss of 26 taxa over the gradient (scenario 1) than it is to detect the loss of 6 taxa (scenario 3), particularly as the number of taxa recovered approaches the zone (gray area) in which the method can no longer reliably detect changes in taxa richness.

The intercept changes, but the slope is constant in simulations using the second method. Only the first two scenarios (1× and 0.5) are shown in Figure 11B. In this example, the number of EPT taxa collected using the second method (equation (2a) falls below the detection level (gray zone) about half way along the gradient. This has two consequences for understanding and detecting responses. First, the response can be detected only over a small portion of the urban gradient. Second, the form of the apparent response (equation (2b) is not linear, but corresponds to a negative exponential curve created when the response is lost in the "noise" associated with the method (gray zone), and the response slope is essentially zero.

The scenarios presented in Figure 11 are illustrative of some of the problems associated with comparing urban responses derived from different studies. Not only can the method used to estimate urban intensity make comparisons difficult, but the method used to collect and process invertebrates also may affect the ability to detect change, quantify the rate of response, and determine the form of the response. Caution is advised when comparing results among studies that differ markedly not only in terms of the area sampled, but also in the number of different habitat types sampled. These problems may account for some of the nonlinear relationships reported for responses to urbanization that have been reported in the literature.

Effective management, protection, and restoration of urban streams is dependent on a comprehensive understanding of physical, chemical, and biological responses and a clear understanding of the similarities and differences in responses among urban areas. Such information is essential if effective monitoring procedures are to be developed, and effective rules and regulations are implemented for urban areas in vastly different natural settings. While the factors (e.g., land-use changes) driving urbanization are similar among regions, our results indicate that elements of the invertebrate assemblages that respond are strongly affected by the local environmental setting in which urbanization is acting. Despite differences in responses among urban areas, there are a few indicators of invertebrate responses (e.g., B-IBIs, tolerance metrics, and ordinations) that can be used to compare responses among disparate environmental settings. These indicators may form the basis for nationally consistent biological indicators of urbanization.

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References

Andrea, R., T. Striebel, and R. Herrmann. 1997. Modeling of dissolved and particle-bound pollutants in urban street runoff. Water Science Technology 36:77–82.

Arnold, C. L., and C. J. Gibbons. 1996. Impervious surface coverage: the emergence of a key environmental indicator. American Planners Association Journal 62:243–258.

Barbour, M. T., J. Gerritsen, B. D. Snyder, and J. B. Stribling. 1999. Rapid bioassessment protocols for use in streams and wadeable rivers: periphyton, benthic macroinvertebrates, and fish. 2nd edition. U.S. Environmental Protection Agency, Office of Water, EPA Report 841-B-99–002, Washington, D.C.

Beasley, G., and P. Kneale. 2002. Reviewing the impact of metals and PAHs on macroinvertebrates in urban watercourses. Progress in Physical Geography 26:236–270.

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.

Cao, Y., D. P. Larsen, R. M. Hughes, P. L. Angermeier, and T. M. Patton. 2002a. Sampling effort affects multivariate comparisons of stream communities. Journal of the North American Benthological Society 21:701–714.

Cao, Y., D. D. Williams, and D. P. Larsen. 2002b. Comparison of ecological communities: the problem of sample representativeness. Ecological Monographs 72:41–56.

Coles, J. F., T. F. Cuffney, G. McMahon, and K. Beaulieu. 2004. The effects of urbanization on the biological, physical, and chemical characteristics of coastal New England streams. U.S. Geological Survey Professional Paper 1695, Denver.

Cuffney, T. F. 2003. User's manual for the National Water-Quality Assessment Program Invertebrate Data Analysis System (IDAS) software: Version 3. U.S. Geological Survey Open-File Report 03–172, Raleigh, North Carolina. Available: ftp://ftpext.usgs.gov/pub/er/nc/raleigh/tfc/IDAS/. (March 2004)

Cuffney, T. F., M. E. Gurtz, and M. R. Meador. 1993. Methods for collecting benthic invertebrate samples as part of the National Water-Quality Assessment Program. U.S. Geological Survey Open-File Report 93–406, Raleigh, North Carolina.

Davis, W. S., and T. P. Simon, editors. 1995. Biological assessment and criteria: tools for water resource planning and decision making. Lewis Publishers, Boca Raton, Florida.

Duda, A. M., D. R. Lenat, and D. L. Penrose. 1982. Water quality in urban streams; what can we expect? Journal of the Water Pollution Federation 54:1139–1147.

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

Fore, L. S., J. R. Karr, and R. W. Wisseman. 1996. Assessing invertebrate responses to human activities: evaluating alternative approaches. Journal of the North American Benthological Society 15:212–231.

Garie, H. L., and A. McIntosh. 1986. Distribution of benthic macroinvertebrates in a stream exposed to urban runoff. Water Resources Bulletin 22:447–455.

Hoffman, R. S., P. D. Capel, and S. J. Larson. 2000. Comparison of pesticides in eight U.S. urban streams. Environmental Toxicology and Chemistry 19:2249–2258.

Horner, R. R., B. D. Booth, A. Azous, and C. W. May. 1997. Watershed determinants of ecosystem functioning. Pages 251–274 *in* C. Roessner, editor. Effects of watershed development and management on aquatic ecosystems. American Society of Civil Engineers, New York.

Huryn, A. D., V. M. B. Huryn, C. J. Arbuckle, and L. Tsomides. 2002. Catchment land-use, macroinvertebrates and detritus processing in headwater streams: taxonomic richness versus function. Freshwater Biology 47:401–415.

Jones, R. C., and C. C. Clark. 1987. Impact of watershed urbanization on stream insect communities. Water Resources Bulletin 23:1047–1055.

Juggins, S. 2003. C^2 user guide. Software for ecological and palaeoecological data analysis and visualisation. University of Newcastle, Newcastle-upon-Tyne, UK.

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

Karr, J. R., and E. W. Chu. 1999. Restoring life in running waters: better biological monitoring. Island Press, Washington, D.C.

Karr, J. R., and E. W. Chu. 2000. Sustaining living rivers. Hydrobiologia 422:1–14.

Kerans, B. L., and J. R. Karr. 1994. Development and testing of a benthic index of biotic integrity (B-IBI) for rivers of the Tennessee Valley. Ecological Applications 4:768–785.

Kemp, S. J., and J. R. Spotila. 1997. Effects of urbanization on brown trout *Salmo trutta*, other fishes and macroinvertebrates in Valley Creek, Valley Forge, Pennsylvania. American Midland Naturalist 138:55–68.

Kennen, J. G. 1999. Relation of macroinvertebrate community impairment to catchment characteristics in New Jersey streams. Journal of the American Water Resources Association 36:939–955.

Kennen, J. G., and M. A. Ayers. 2002. Relation of environmental characteristics to the composition of aquatic assemblages along a gradient of urban land use in New Jersey, 1996–98. U.S. Geological Survey Water-Resources Investigations Report 02–4069, West Trenton, New Jersey.

Klein, R. D. 1979. Urbanization and stream quality impairment. Water Resources Bulletin 15:948–963.

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.

May, C. W., R. Horner, J. R. Karr, B. Mar, and E. Welch. 1997. Effects of urbanization on small streams in the Puget Sound Lowland ecoregion. Watershed Protection Techniques 2:483–494.

Maxted, J.R. and E. Shaver. 1997. The use of retention basins to mitigate stormwater on aquatic life. Pages 494–512 *in* L.A. Roesner, editor. Effects of watershed development and management on aquatic ecosystems. American Society of Civil Engineers, New York.

McCune, B., J. B. Grace, and D. L. Urban. 2002. Analysis of ecological communities. MjM Software Design, Gleneden Beach, Oregon.

McMahon, G., and T. F. Cuffney. 2000. Quantifying urban intensity in drainage basins for assessing stream ecological conditions. Journal of American Water Resources Association 36:1247–1261.

Meador, M. R., J. F. Coles, and H. Zappia. 2005. Fish assemblage responses to urban intensity gradients in contrasting metropolitan areas: Birmingham, Alabama, and Boston, Massachusetts. Pages 409–423 *in* L. R. Brown, R. H. Gray, R. M. Hughes, and M. R. Meador, editors. Effects of urbanization on stream ecosystems. American Fisheries Society, Symposium 47, Bethesda, Maryland.

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.

Moulton, S. R., J. L. Carter, S. A. Grotheer, T. F. Cuffney, and T. M. Short. 2000. Methods for analysis by the U.S. Geological Survey National Water Quality Laboratory – processing, taxonomy, and quality control of benthic macroinvertebrate samples. U.S.

Geological Survey Open-File Report 00–212, Denver.

Natural Resources Conservation Service. 2000. Summary report, 1997 National resources inventory, revised December 2000. U.S. Department of Agriculture, Washington, D.C.

NCDENR (North Carolina Department of Environment and Natural Resources). 2003. Standard operating procedures for benthic macroinvertbrates, Raleigh, North Carolina. Available: http://www.esb.enr.state.nc.us/BAUwww/benthossop.pdf. (August, 2001)

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.

Paul, M. J., and J. L. Meyer. 2001. Streams in the urban landscape. Annual Review of Ecology and Systematics 32:333–365.

Pedersen, E. R., and M. A. Perkins. 1986. The use of benthic macroinvertebrate data for evaluating impacts of urban runoff. Hydrobiologia 139:13–22.

Potapova, M., J. C. Coles, E. M. P. Giddings, and H. Zappia. 2005. A comparison of the influences of urbanization in contrasting environmental settings on stream benthic algal assemblages. Pages 333–359 *in* L. R. Brown, R. H. Gray, R. M. Hughes, and M. R. Meador, editors. Effects of urbanization on stream ecosystems. American Fisheries Society, Symposium 47, Bethesda, Maryland.

Pratt, J. M., R. A. Coler, and P. J. Godfrey. 1981. Ecological effects of urban stormwater runoff on benthic macroinvertebrates inhabiting the Green River, Massachusetts. Hydrobiologia 83:29–42.

Rosenberg, D. M., and V. H. Resh, editors. 1993. Freshwater biomonitoring and benthic macroinvertebrates. Routledge, Chapman and Hall, Inc., New York.

Roy, A. H., A. D. Rosemona, M. J. Paul, D. S. Leigh, and J. B. Wallace. 2003. Stream macroinvertebrate response to catchment urbanization (Georgia, U.S.A.). Freshwater Biology 48:329–346.

Sansalone, J. J., and S. G. Buchberger. 1997a. Characterization of solid and metal element distributions in urban highway stormwater. Water Science and Technology 36(8–9):155–160.

Sansalone, J. J., and S. G. Buchberger. 1997b. Partitioning and first flush of metals in urban roadway stormwater. Journal of Environmental Engineering 123:134–143.

Sansalone, J. J., J. M. Koran, J. A., Smithson, and S. G. Buchberger. 1998. Physical characteristics of urban roadway solids transported during rain events. Journal of Environmental Engineering 124:427–440.

Schueler, T. 1994. The importance of imperviousness. Watershed Protection 1:100–111.

Schueler, T. R., and J. Galli. 1992. Environmental impacts of stormwater ponds. Pages 159–180 *in* P. Kumble and T. Schueler, editors. Watershed restoration source book. Metropolitan Washington Council of Governments, Publication 92705, Washington, D.C.

Seaburn, G. E. 1969. Effects of urban development on direct runoff to East Meadow Brook, Nassau County, New York. U.S. Geological Survey Professional Paper 627-B, Washington, D.C.

Shaver, E. J., G. C. Maxted, and D. Carter. 1995 Watershed protection using an integrated approach. Pages 435–459 *in* H. C. Torno, editor. Stormwater NPDES related monitoring needs. American Society of Civil Engineers, Crested Butte, Colorado.

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

Short, T. M., E. M. P. Giddings, H. Zappia, and J. F. Coles. 2005. Urbanization effects on stream habitat characteristics in Boston, Massachusetts; Birmingham, Alabama; and Salt Lake City, Utah. Pages 317–332 *in* L. R. Brown, R. H. Gray, R. M. Hughes, M. R. Meador, editors. Effects of urbanization on stream ecosystems. American Fisheries Society, Symposium 47, Bethesda, Maryland.

SPSS. 1999. SYSTAT 9 statistics I & II. SPSS Inc., Chicago.

Tate, C. M., T. F. Cuffney, G. McMahon, E. M. P. Giddings, J. F. Coles, and H. Zappia. 2005. Use of an urban intensity index to assess urban effects on streams in three contrasting environmental settings. Pages 291–315 *in* L. R. Brown, R. H.Gray, R. M. Hughes, and M. R. Meador, editors. Effects of urbanization on stream ecosystems. American Fisheries Society, Symposium 47, Bethesda, Maryland.

ter Braak, C. J. F. 1995. Chapter 5, Ordination. Pages 91–173 *in* R. H. G. Jongman, C. J. F. ter Braak, and O. F. R. Van Tongeren, editors. Data analysis in community and landscape ecology. Cambridge University Press, New York.

ter Braak, C. J. F. 1996. Unimodal models to relate species to environment. DLO-Agricultural Mathematics Group, Wageningen, Netherlands.

ter Braak, C. J. F. and P. Šmilauer. 1998. CANOCO manual and user's guide to CANOCO for windows: software for CANOCO community ordination (version 4). Microcomputer Power, Ithaca, New York.

U.S. Environmental Protection Agency. 2000. The quality of our nation's waters. EPA Report 841-S-00–001, Washington, D.C.

U.S. Geological Survey. 2003. National elevation dataset. U.S. Geological Survey EROS Data Center, Sioux Falls, South Dakota. Available: http://gisdata.usgs.gov/ned/. (May 2003)

Vølstad, J. H., N. E. Roth, G. Mercurio, M. T. Southerland, and D. E. Strebel. 2003. Using environmental stressor information to predict the ecological status of Maryland non-tidal streams as measured by biological indicators. Environmental Monitoring and Assessment 84:219–242.

Whiting, E. R., and H. J. Clifford. 1983. Invertebrates and urban runoff in a small northern stream, Edmonton, Alberta, Canada. Hydrobiologia 102:73–80.

Yoder, C. O., R. J. Miltner, and D. White. 1999. Assessing the status of aquatic life designated uses in urban and suburban watersheds. Pages 16–28 *in* A. Everson, S. Minamyer, J. Dye, P. Heimbrock, and S.Wilson, editors. National Conference on Retrofit Opportunities for Water Resource Protection in Urban Environments. U.S. Environmental Protection Agency, EPA/625/R-99/002, Washington, DC.

Yoder, C. O., and E. T. Rankin. 1996. Assessing the condition and status of aquatic life designated uses in urban and suburban watersheds. Pages 201–227 *in* L. A. Roesner, editor. Effect of watershed development and management on aquatic ecosystems. American Society of Civil Engineers, New York.

Zar, J. H. 1974. Biostatistical analysis. Prentice-Hall, Inc., Englewood Cliffs, New Jersey.

APPENDIX 1. Occurrence (% of sites) of taxa across the urban gradient based on QRC samples for the Boston (BOS) Birmingham (BIR) and Salt Lake City (SLC) urban study areas; all sites (ALL: BOS n = 30, BIR n = 28, and SLC n = 30), background sites with UII ≤ 10 (BOS n = 8, BIR n = 7, SLC n = 3) and highly urbanized sites with UII ≥ 70 (BOS n = 5, BIR n = 3, SLC n = 11).

	BOS			BIR			SLC		
Taxon	All	UII ≤ 10	UII ≥ 70	All	UII ≤ 10	UII ≥ 70	All	UII ≤ 10	UII ≥ 70
Hydra sp.	3.3	0.0	20.0				6.7	0.0	0.0
Turbellaria	46.7	12.5	100.0	25.0	14.3	100.0	60.0	100.0	63.6
Prostoma sp.	16.7	0.0	40.0				6.7	0.0	18.2
Nematoda	83.3	75.0	80.0	53.6	42.9	66.7	70.0	33.3	54.5
Campeloma sp.	16.7	12.5	20.0	10.7	14.3	0.0			
Hydrobiidae	60.0	37.5	100.0						
Fluminicola sp.							60.0	66.7	54.5
Elimia sp.				67.9	100.0	0.0			
Leptoxis sp.				7.1	0.0	0.0			
Pleurocera sp.				7.1	0.0	33.3			
Ferrissia sp.	6.7	0.0	0.0	32.1	0.0	66.7			
Hebetancylus excentricus				35.7	0.0	100.0			
Laevapex sp.	30.0	12.5	80.0						
Lymnaeidae	6.7	0.0	20.0						
Pseudosuccinea columella				3.6	14.3	0.0	3.3	0.0	9.1
Stagnicola sp.							23.3	0.0	27.3
Physella sp.	46.7	37.5	100.0	32.1	28.6	66.7	60.0	33.3	63.6
Gyraulus sp.	6.7	0.0	0.0				20.0	0.0	45.5
Helisoma sp.	40.0	37.5	40.0						
Planorbella sp.	16.7	0.0	20.0		25.0	0.0	33.3		
Planorbula armigera	13.3	0.0	20.0						
Elliptio complanata	23.3	12.5	0.0						
Corbicula sp.				53.6	42.9	33.3			
Pisidium sp.	93.3	100.0	100.0	3.6	0.0	0.0	63.3	66.7	45.5
Musculium sp.	50.0	12.5	100.0						
Sphaerium sp.				32.1	28.6	0.0			
Megadrile	30.0	0.0	60.0	78.6	57.1	100.0	86.7	66.7	90.9
Lumbriculidae	90.0	87.5	80.0	46.4	42.9	66.7	40.0	0.0	63.6
Naididae				35.7	0.0	100.0	83.3	0.0	100.0

APPENDIX 1. Continued.

Taxon	BOS All	BOS UII ≤ 10	BOS UII ≥ 70	BIR All	BIR UII ≤ 10	BIR UII ≥ 70	SLC All	SLC UII ≤ 10	SLC UII ≥ 70
Dero sp.	73.3	87.5	80.0						
Tubificidae	40.0	25.0	60.0				93.3	100.0	100.0
Branchiura sowerbyi				67.9	57.1	66.7			
Enchytraeidae	26.7	37.5	40.0	7.1	14.3	33.3	73.3	66.7	90.9
Glossiphonia complanata	20.0	0.0	80.0						
Helobdella stagnalis	16.7	0.0	60.0				43.3	33.3	54.5
Placobdella ornata	3.3	12.5	0.0						
P. parasitica	6.7	0.0	0.0	7.1	0.0	33.3			
Erpobdellidae	50.0	37.5	80.0	21.4	14.3	66.7	76.7	33.3	90.9
Acari	86.7	100.0	60.0	100.0	100.0	100.0	96.7	100.0	100.0
Cambaridae							3.3	0.0	9.1
Cambarus sp.				57.1	100.0	33.3			
Orconectes sp.	63.3	50.0	80.0						
O. rusticus				35.7	14.3	33.3			
Procambarus sp.	3.3	0.0	0.0						
Caecidotea sp.	60.0	37.5	100.0	17.9	28.6	0.0	46.7	33.3	54.5
Lirceus sp.				32.1	42.9	33.3			
Crangonyx sp.	16.7	0.0	40.0	21.4	14.3	0.0	13.3	0.0	18.2
Synurella sp.	36.7	62.5	20.0						
Gammaridae				3.6	0.0	0.0			
Gammarus sp.	30.0	0.0	80.0				23.3	0.0	0.0
Hyalella azteca	53.3	37.5	20.0	14.3	28.6	0.0	23.3	0.0	27.3
Collembola	3.3	0.0	0.0	17.9	0.0	66.7	20.0	33.3	9.1
Choroterpes sp.				10.7	42.9	0.0			
Paraleptophlebia sp.	36.7	87.5	0.0	10.7	42.9	0.0	10.0	0.0	0.0
Hexagenia atrocaudata				3.6	0.0	0.0			
H. bilineata				17.9	28.6	0.0			
H. limbata				3.6	14.3	0.0			
Caenis sp.	20.0	37.5	40.0						
C. diminuta group				3.6	14.3	0.0			
C. hilaris group				3.6	14.3	0.0			
C. anceps				17.9	42.9	0.0			
Drunella coloradensis							3.3	0.0	9.1
D. doddsi							6.7	66.7	0.0
D. flavilinea							10.0	66.7	9.1
Ephemerella inermis							6.7	0.0	0.0
Eurylophella sp.	26.7	62.5	0.0						
E. aestiva				21.4	57.1	0.0			
Serratella deficiens	33.3	75.0	0.0	17.9	42.9	0.0			
S. serrata	33.3	75.0	0.0						
S. tibialis							13.3	66.7	18.2
Timpanoga lita				3.6	14.3	0.0			
Tricorythodes sp.	26.7	25.0	0.0	46.4	14.3	0.0	30.0	0.0	36.4
Ameletus sp.							6.7	66.7	0.0
Centroptilum/Procloeon sp.	16.7	37.5	0.0	10.7	14.3	0.0			
Acentrella turbida	6.7	0.0	0.0	14.3	42.9	0.0			
Acerpenna pygmaea	23.3	25.0	0.0	10.7	28.6	0.0			

APPENDIX 1. Continued.

	BOS			BIR			SLC		
Taxon	All	UII ≤ 10	UII ≥ 70	All	UII ≤ 10	UII ≥ 70	All	UII ≤ 10	UII ≥ 70
Baetis bicaudatus							3.3	33.3	0.0
B. flavistriga	76.7	87.5	40.0	82.1	71.4	66.7			
B. intercalaris	6.7	12.5	0.0	39.3	57.1	0.0			
B. tricaudatus	13.3	12.5	0.0				93.3	100.0	100.0
Callibaetis sp.				7.1	0.0	0.0	6.7	0.0	9.1
Diphetor hageni							3.3	33.3	0.0
Heterocloeon curiosum				7.1	14.3	0.0			
Plauditus sp.	36.7	37.5	0.0						
Pseudocloeon sp.	30.0	12.5	0.0						
P. propinquum				3.6	14.3	0.0			
Siphlonurus sp.	13.3	37.5	0.0				10.0	0.0	18.2
Cinygmula sp.							13.3	66.7	0.0
Epeorus sp.	20.0	50.0	0.0						
E. longimanus							26.7	66.7	9.1
Leucrocuta sp.	3.3	0.0	0.0	14.3	57.1	0.0			
Nixe sp.							3.3	0.0	0.0
Rhithrogena sp.							3.3	0.0	0.0
Stenacron sp.	10.0	0.0	0.0						
S. interpunctatum				35.7	100.0	0.0			
Stenonema modestum/ smithae	60.0	37.5	40.0						
S. exiguum				3.6	14.3	0.0			
S. femoratum				10.7	14.3	0.0			
S. mediopunctatum				3.6	14.3	0.0			
S. pulchellum				46.4	71.4	0.0			
S. terminatum				10.7	28.6	0.0			
S. vicarium	26.7	75.0	0.0						
Isonychia sp.	63.3	100.0	0.0	60.7	100.0	0.0			
Calopteryx maculata	66.7	87.5	0.0	14.3	42.9	0.0			
Hetaerina americana	10.0	0.0	0.0	3.6	0.0	0.0			
Coenagrion/Enallagma sp.	6.7	0.0	20.0						
Argia fumipennis	43.3	62.5	0.0	10.7	0.0	33.3			
A. moesta				3.6	0.0	0.0			
A. sedula				71.4	71.4	66.7			
A. vivida							20.0	0.0	18.2
Enallagma sp.	6.7	0.0	40.0						
Enallagma weewa				3.6	14.3	0.0			
Ischnura sp.	3.3	0.0	20.0						
Aeshna sp.	16.7	12.5	20.0				10.0	0.0	9.1
Basiaeschna janata	13.3	25.0	0.0						
Boyeria grafiana	23.3	37.5	0.0						
Boyeria vinosa	83.3	100.0	40.0	78.6	100.0	33.3			
Nasiaeschna pentacantha	3.3	0.0	0.0						
Epitheca princeps				7.1	14.3	0.0			
Helocordulia uhleri	20.0	50.0	0.0						
Neurocordulia obsoleta	3.3	12.5	0.0						

APPENDIX 1. Continued.

Taxon	BOS All	BOS UII ≤ 10	BOS UII ≥ 70	BIR All	BIR UII ≤ 10	BIR UII ≥ 70	SLC All	SLC UII ≤ 10	SLC UII ≥ 70
Dromogomphus sp.				3.6	0.0	0.0			
Erpetogomphus sp.				3.6	0.0	0.0			
Gomphus sp.	36.7	87.5	0.0	42.9	28.6	33.3			
Hagenius brevistylus	33.3	75.0	0.0	39.3	28.6	33.3			
Lanthus sp.				7.1	28.6	0.0			
Ophiogomphus sp.	16.7	25.0	0.0						
O. severus							13.3	0.0	18.2
Progomphus sp.				3.6	0.0	0.0			
Stylogomphus albistylus	43.3	87.5	0.0	7.1	14.3	0.0			
Libellulidae				3.6	0.0	0.0			
Libellula sp.							3.3	0.0	0.0
Macromia illinoiensis	6.7	0.0	0.0	28.6	42.9	0.0			
Capniidae							10.0	33.3	0.0
Paracapnia sp.	10.0	37.5	0.0						
Leuctra sp.	33.3	50.0	40.0	3.6	14.3	0.0			
Paraleuctra sp.							3.3	33.3	0.0
Amphinemura sp.							6.7	0.0	0.0
Malenka sp.							6.7	0.0	0.0
Zapada sp.							13.3	66.7	9.1
Suwallia sp.							6.7	33.3	0.0
Sweltsa sp.	13.3	50.0	0.0				16.7	66.7	18.2
Peltoperlidae				3.6	14.3	0.0			
Tallaperla sp.	10.0	0.0	0.0						
Acroneuria sp.				14.3	42.9	0.0			
A. lycorias	66.7	100.0	40.0						
Hesperoperla pacifica							30.0	100.0	0.0
Perlesta sp.	10.0	0.0	0.0	21.4	28.6	0.0			
Perlinella drymo	3.3	0.0	0.0						
Paragnetina immarginata	3.3	12.5	0.0						
P. media	40.0	75.0	40.0						
Perlodidae	3.3	0.0	0.0						
Isoperla sp.							10.0	0.0	0.0
Skwala sp.							10.0	33.3	0.0
Pteronarcella badia							10.0	0.0	0.0
Pteronarcys californica							20.0	33.3	0.0
Belostoma sp.				3.6	14.3	0.0			
B. flumineum	13.3	0.0	20.0						
Corisella decolor							6.7	0.0	9.1
Hesperocorixa laevigata							6.7	0.0	18.2
Sigara sp.	6.7	12.5	20.0				16.7	0.0	9.1
Trichocorixa sp.	6.7	0.0	40.0	3.6	0.0	0.0			
Aquarius conformis	20.0	25.0	0.0	17.9	0.0	0.0			
A.nyctalis							60.0	33.3	54.5
A. remigis	50.0	75.0	60.0						
Gerris marginatus				3.6	0.0	0.0			
Limnoporus canaliculatus				3.6	14.3	0.0			
Rheumatobates sp.	6.7	0.0	20.0	7.1	0.0	0.0			

APPENDIX 1. Continued.

	BOS			BIR			SLC		
Taxon	All	UII ≤ 10	UII ≥ 70	All	UII ≤ 10	UII ≥ 70	All	UII ≤ 10	UII ≥ 70
Metrobates sp.				14.3	0.0	0.0			
M. hesperius	16.7	25.0	0.0						
Trepobates pictus				32.1	28.6	0.0			
Ranatra fusca	3.3	12.5	0.0						
R. kirkaldyi	3.3	0.0	20.0						
Notonecta sp.				3.6	0.0	0.0	3.3	0.0	0.0
N. irrorata	13.3	12.5	40.0						
Microvelia sp.	3.3	12.5	0.0						
Rhagovelia distincta							16.7	0.0	9.1
R. obesa	66.7	75.0	60.0	39.3	71.4	0.0			
Chauliodes rastricornis	13.3	0.0	60.0						
Nigronia fasciatus				3.6	14.3	0.0			
N. serricornis	73.3	100.0	0.0	32.1	57.1	0.0			
Corydalus cornutus	30.0	25.0	0.0	71.4	71.4	0.0			
Sialis sp.	56.7	87.5	60.0	25.0	57.1	0.0			
Glossosoma sp.	56.7	87.5	20.0	10.7	28.6	0.0	3.3	0.0	0.0
Protoptila sp.				17.9	14.3	0.0			
Hydroptila sp.	36.7	37.5	60.0	57.1	28.6	100.0			
H. arctia							60.0	33.3	81.8
Leucotrichia sp.							3.3	0.0	0.0
L. pictipes				7.1	0.0	0.0			
Mayatrichia ayama	3.3	0.0	0.0						
Ochrotrichia sp.				3.6	0.0	0.0	13.3	33.3	0.0
Oxyethira sp.	3.3	0.0	0.0						
Rhyacophila coloradensis group							30.0	66.7	18.2
R. hyalinata group							3.3	33.3	0.0
R. rotunda group							3.3	33.3	0.0
R. brunnea							30.0	33.3	18.2
R. coloradensis							3.3	0.0	0.0
R. fuscula	43.3	100.0	0.0	3.6	14.3	0.0			
Chimarra sp.	93.3	100.0	60.0	57.1	85.7	33.3			
Dolophilodes sp.				3.6	0.0	0.0	10.0	33.3	0.0
D. distinctus	30.0	50.0	0.0						
Wormaldia sp.							6.7	33.3	0.0
Phylocentropus sp.	3.3	12.5	0.0						
Arctopsyche grandis							20.0	33.3	18.2
Parapsyche elsis							3.3	33.3	0.0
Diplectrona modesta	3.3	12.5	0.0						
Ceratopsyche cf. *alhedra*	3.3	12.5	0.0						
C. alhedra	3.3	12.5	0.0						
C. bronta	20.0	50.0	0.0						
C. cheilonis				14.3	28.6	0.0			
C. cockerelli							36.7	66.7	9.1
C. morosa	10.0	12.5	0.0						
C. oslari							36.7	66.7	9.1
C. sparna	63.3	75.0	0.0	25.0	57.1	0.0			

APPENDIX 1. Continued.

Taxon	BOS			BIR			SLC		
	All	UII ≤ 10	UII ≥ 70	All	UII ≤ 10	UII ≥ 70	All	UII ≤ 10	UII ≥ 70
Cheumatopsyche sp.	90.0	75.0	100.0	96.4	85.7	100.0	10.0	0.0	9.1
Hydropsyche rossi/simulans				3.6	0.0	0.0			
H. depravata group	100.0	100.0	100.0	64.3	28.6	33.3			
H. betteni	3.3	0.0	20.0						
H. californica							3.3	0.0	0.0
H. occidentalis							63.3	33.3	72.7
Macrostemum sp.	23.3	25.0	0.0						
Neureclipsis sp.	3.3	0.0	0.0						
Paranyctiophylax sp.	3.3	0.0	0.0						
Polycentropus sp.	40.0	50.0	20.0	35.7	85.7	0.0			
Lype diversa	40.0	75.0	0.0	17.9	28.6	0.0			
Psychomyia flavida	6.7	0.0	0.0	7.1	14.3	0.0			
Tinodes sp.							10.0	0.0	9.1
Limnephiloidea	13.3	25.0	0.0						
Apatania sp.							3.3	0.0	0.0
Brachycentrus americanus							16.7	0.0	9.1
B. appalachia	13.3	25.0	0.0						
B. numerosus	26.7	62.5	0.0						
B. occidentalis							3.3	0.0	0.0
Micrasema sp.	56.7	87.5	0.0				10.0	33.3	0.0
M. wataga				35.7	57.1	0.0			
Goera sp.	3.3	0.0	0.0	3.6	14.3	0.0			
Lepidostoma sp.	16.7	37.5	0.0				43.3	100.0	27.3
Onocosmoecus unicolor							3.3	0.0	0.0
Frenesia sp.	3.3	0.0	0.0						
Hesperophylax sp.							13.3	0.0	0.0
Pycnopsyche sp.	63.3	75.0	0.0						
Neophylax sp.	33.3	25.0	0.0				3.3	0.0	0.0
Neothremma alicia							3.3	33.3	0.0
Anisocentropus pyraloides				3.6	14.3	0.0			
Ceraclea sp.	10.0	12.5	0.0						
Mystacides sepulchralis	13.3	25.0	0.0	7.1	14.3	0.0			
Oecetis sp.	30.0	62.5	40.0						
O. avara group							10.0	0.0	18.2
O. disjuncta							13.3	33.3	9.1
O. persimilis				10.7	14.3	0.0			
Triaenodes sp.	3.3	0.0	20.0						
T. cumberlandensis/ melaca				7.1	14.3	0.0			
T. ignitus				3.6	0.0	0.0			
Molanna sp.	23.3	25.0	0.0	3.6	0.0	0.0			
Psilotreta labida	20.0	62.5	0.0						
Helicopsyche borealis	20.0	50.0	0.0	7.1	14.3	0.0			
Lepidoptera	33.3	37.5	20.0						
Petrophila sp.				25.0	0.0	0.0			
Agabus sp.				3.6	14.3	0.0	30.0	0.0	36.4
Liodessus affinis						3.3	0.0	0.0	

APPENDIX 1. Continued.

	BOS			BIR			SLC		
Taxon	All	UII ≤ 10	UII ≥ 70	All	UII ≤ 10	UII ≥ 70	All	UII ≤ 10	UII ≥ 70
Hydroporini	13.3	0.0	0.0						
Oreodytes sp.							6.7	0.0	9.1
Laccophilus sp.	3.3	0.0	20.0						
Dineutus ciliatus	3.3	0.0	0.0						
D. discolor	46.7	37.5	0.0	17.9	28.6	0.0			
Gyrinus bifarius							3.3	0.0	0.0
G. lecontei	16.7	37.5	0.0						
G. marginellus	16.7	25.0	0.0						
Brychius sp.							13.3	0.0	0.0
Haliplus sp.	6.7	0.0	0.0						
Peltodytes sp.	13.3	0.0	60.0	21.4	14.3	0.0			
P. callosus							6.7	33.3	0.0
Staphylinidae				14.3	42.9	0.0	6.7	33.3	0.0
Helophorus sp.							3.3	0.0	0.0
Ametor scabrosus							20.0	33.3	9.1
Berosus sp.				25.0	0.0	66.7			
Cymbiodyta sp.	6.7	12.5	0.0	3.6	14.3	0.0			
Hydrobius sp.	6.7	12.5	0.0						
Laccobius sp.							3.3	0.0	0.0
Paracymus sp.	3.3	0.0	20.0	3.6	0.0	0.0			
Sperchopsis tessellata	16.7	12.5	0.0	10.7	14.3	0.0			
Tropisternus sp.	3.3	0.0	20.0	7.1	0.0	0.0			
Scirtidae				10.7	0.0	0.0			
Helichus basalis				7.1	28.6	0.0			
H. fastigiatus	6.7	0.0	0.0	7.1	0.0	0.0			
H. lithophilus				3.6	0.0	0.0			
H. striatus							16.7	33.3	9.1
Ancyronyx variegata	53.3	25.0	80.0	21.4	14.3	0.0			
Cleptelmis addenda							10.0	33.3	0.0
Dubiraphia sp.	63.3	87.5	40.0	75.0	100.0	0.0			
Heterlimnius corpulentus							16.7	66.7	18.2
Lara avara							3.3	33.3	0.0
Macronychus glabratus	80.0	87.5	40.0	35.7	42.9	0.0			
Microcylloepus pusillus	26.7	12.5	0.0	42.9	42.9	0.0			
Narpus concolor							20.0	66.7	27.3
Optioservus castanipennis							16.7	0.0	9.1
O. divergens							6.7	0.0	0.0
O. fastiditus	3.3	0.0	0.0						
O. ovalis	43.3	62.5	0.0	10.7	14.3	0.0			
O. quadrimaculatus							43.3	66.7	9.1
O. trivittatus				57.1	85.7	0.0			
Oulimnius latiusculus	56.7	87.5	0.0	25.0	42.9	0.0			
Promoresia tardella	40.0	62.5	0.0	3.6	14.3	0.0			
Stenelmis concinna	6.7	12.5	0.0						
S. crenata	80.0	87.5	60.0	89.3	100.0	100.0			
S, mera	3.3	0.0	0.0						

APPENDIX 1. Continued.

Taxon	BOS All	BOS UII ≤ 10	BOS UII ≥ 70	BIR All	BIR UII ≤ 10	BIR UII ≥ 70	SLC All	SLC UII ≤ 10	SLC UII ≥ 70
Zaitzevia parvula							26.7	33.3	0.0
Ectopria sp.	23.3	62.5	0.0	3.6	0.0	0.0			
Psephenus herricki	60.0	75.0	0.0	64.3	100.0	0.0			
Anchytarsus bicolor	3.3	0.0	0.0						
Lampyridae							3.3	0.0	9.1
Curculionidae				3.6	0.0	0.0	10.0	0.0	0.0
Blephariceridae				3.6	14.3	0.0			
Ceratopogonidae	6.7	0.0	0.0	14.3	14.3	0.0			
Bezzia/Palpomyia sp.							6.7	0.0	0.0
Probezzia sp.							20.0	0.0	27.3
Phaenopsectra/Tribelos sp.	23.3	25.0	20.0	28.6	28.6	0.0			
Chironomus sp.	10.0	0.0	0.0	57.1	57.1	66.7	53.3	0.0	72.7
Cryptochironomus sp.	20.0	12.5	40.0	67.9	71.4	66.7	66.7	0.0	81.8
Demicryptochironomus sp.							3.3	0.0	0.0
Dicrotendipes sp.	13.3	12.5	40.0	53.6	42.9	100.0	33.3	0.0	45.5
Glyptotendipes sp.	3.3	0.0	20.0	3.6	0.0	0.0			
Lauterborniella agrayloides	6.7	12.5	0.0						
Microtendipes sp.	63.3	100.0	0.0	32.1	85.7	0.0	6.7	0.0	0.0
Nilothauma sp.	3.3	12.5	0.0						
Parachironomus sp.	3.3	0.0	0.0				6.7	0.0	18.2
Paracladopelma sp.							10.0	33.3	0.0
Paratendipes sp.	40.0	50.0	20.0	35.7	42.9	0.0	16.7	0.0	27.3
Phaenopsectra sp.	20.0	25.0	20.0	67.9	71.4	33.3	93.3	66.7	100.0
Polypedilum sp.	100.0	100.0	100.0	96.4	100.0	100.0	73.3	100.0	63.6
Stelechomyia perpulchra	3.3	12.5	0.0						
Stenochironomus sp.	46.7	37.5	40.0	25.0	28.6	0.0	6.7	0.0	18.2
Stictochironomus sp.	3.3	0.0	0.0	14.3	28.6	0.0			
Tribelos sp.	23.3	50.0	0.0	32.1	28.6	0.0			
Xenochironomus xenolabis	13.3	25.0	0.0	3.6	14.3	0.0	3.3	0.0	9.1
Xestochironomus sp.				3.6	0.0	0.0			
Pseudochironomus sp.	3.3	0.0	0.0	17.9	28.6	0.0			
Micropsectra/Tanytarsus sp.	46.7	75.0	40.0	14.3	14.3	0.0	50.0	66.7	36.4
Cladotanytarsus sp.				7.1	0.0	0.0			
Micropsectra sp.	30.0	75.0	20.0				63.3	66.7	54.5
Paratanytarsus sp.	3.3	0.0	0.0	28.6	42.9	33.3	6.7	0.0	9.1
Rheotanytarsus sp.	93.3	100.0	80.0	92.9	100.0	100.0	40.0	33.3	36.4
Stempellina sp.	23.3	62.5	0.0						
Stempellinella sp.	30.0	62.5	0.0	17.9	14.3	0.0	6.7	33.3	0.0
Sublettea coffmani	3.3	0.0	0.0	7.1	0.0	0.0			
Tanytarsus sp.	56.7	87.5	40.0	64.3	85.7	100.0	10.0	0.0	18.2
Diamesa sp.	6.7	0.0	20.0						
Pagastia sp.	30.0	50.0	20.0				63.3	100.0	45.5
Pseudodiamesa sp.							6.7	33.3	9.1
Cricotopus/ Orthocladius sp.	63.3	75.0	60.0	53.6	42.9	66.7	70.0	66.7	90.9
Eukiefferiella/Tvetenia sp.	30.0	62.5	20.0						

APPENDIX 1. Continued.

	BOS			BIR			SLC		
Taxon	All	UII ≤ 10	UII ≥ 70	All	UII ≤ 10	UII ≥ 70	All	UII ≤ 10	UII ≥ 70
Brillia sp.	66.7	87.5	40.0	14.3	28.6	0.0	60.0	66.7	63.6
Cardiocladius sp.	16.7	0.0	20.0	28.6	14.3	33.3	10.0	33.3	0.0
Corynoneura sp.	16.7	12.5	20.0				10.0	0.0	18.2
Cricotopus bicinctus group	53.3	37.5	40.0	57.1	28.6	100.0	80.0	66.7	90.9
C. trifascia group				7.1	0.0	33.3	53.3	33.3	45.5
Eukiefferiella sp.	56.7	75.0	40.0	10.7	14.3	0.0	90.0	100.0	90.9
Heleniella sp.							3.3	33.3	0.0
Heterotrissocladius sp.				3.6	14.3	0.0			
Hydrobaenus sp.							6.7	33.3	0.0
Limnophyes sp.							6.7	0.0	9.1
Lopescladius sp.	6.7	25.0	0.0	7.1	14.3	0.0			
Nanocladius sp.	40.0	75.0	60.0	14.3	0.0	100.0	16.7	0.0	36.4
Orthocladius lignicola	33.3	75.0	0.0	3.6	0.0	0.0	6.7	33.3	0.0
Parachaetocladius sp.	30.0	87.5	0.0						
Parakiefferiella sp.	6.7	12.5	0.0				13.3	0.0	9.1
Parametriocnemus sp.	83.3	100.0	40.0	67.9	100.0	33.3	76.7	33.3	90.9
Paraphaenocladius sp.	13.3	0.0	0.0	3.6	0.0	0.0			
Psectrocladius sp.	3.3	12.5	0.0						
Rheocricotopus sp.	56.7	62.5	40.0	71.4	57.1	100.0	66.7	100.0	90.9
Synorthocladius sp.	6.7	12.5	0.0	25.0	14.3	0.0	10.0	0.0	0.0
Thienemanniella sp.	53.3	75.0	40.0	28.6	28.6	0.0	40.0	33.3	45.5
Tvetenia sp.	80.0	100.0	40.0	35.7	57.1	0.0	53.3	100.0	27.3
Xylotopus par	33.3	62.5	0.0	10.7	14.3	0.0			
Zalutschia sp.							3.3	0.0	0.0
Odontomesa sp.							16.7	33.3	0.0
Prodiamesa sp.	3.3	0.0	0.0				60.0	33.3	54.5
Clinotanypus sp.	10.0	0.0	20.0	3.6	14.3	0.0			
Alotanypus sp.							6.7	0.0	9.1
Apsectrotanypus sp.				3.6	14.3	0.0			
Psectrotanypus sp.	3.3	0.0	20.0						
Radotanypus submarginella							46.7	33.3	45.5
Natarsia sp.				14.3	14.3	0.0			
Thienemannimyia group	93.3	100.0	100.0	92.9	100.0	100.0	50.0	66.7	45.5
Ablabesmyia sp.	73.3	62.5	80.0	96.4	85.7	100.0			
Labrundinia sp.	6.7	0.0	0.0	3.6	0.0	0.0			
Larsia sp.	3.3	12.5	0.0				6.7	33.3	0.0
Nilotanypus sp.	40.0	37.5	0.0	10.7	14.3	0.0			
Paramerina sp.				14.3	14.3	33.3	3.3	0.0	0.0
Pentaneura sp.	20.0	0.0	20.0	14.3	14.3	0.0			
Zavrelimyia sp.	3.3	12.5	0.0	3.6	14.3	0.0			
Procladius sp.	16.7	25.0	40.0	46.4	14.3	66.7	10.0	0.0	27.3
Culex sp.	3.3	0.0	0.0				10.0	0.0	18.2
Dixidae							3.3	0.0	0.0
Dixa sp.				10.7	14.3	0.0			
Dixella sp.				7.1	0.0	0.0			
Pericoma/ Telmatoscopus sp.							13.3	0.0	9.1

APPENDIX 1. Continued.

Taxon	BOS			BIR			SLC		
	All	UII ≤ 10	UII ≥ 70	All	UII ≤ 10	UII ≥ 70	All	UII ≤ 10	UII ≥ 70
Maruina sp.							6.7	33.3	0.0
Psychoda sp.							3.3	0.0	0.0
Simulium sp.	100.0	100.0	100.0	85.7	85.7	66.7	80.0	66.7	90.9
Tanyderidae				3.6	14.3	0.0			
Thaumalea sp.							6.7	33.3	0.0
Prionocera sp.							6.7	33.3	9.1
Tipula sp.	56.7	37.5	60.0	50.0	57.1	0.0	53.3	66.7	36.4
Antocha sp.	46.7	50.0	20.0	71.4	71.4	33.3	26.7	66.7	0.0
Dicranota sp.	23.3	37.5	0.0				13.3	33.3	9.1
Hexatoma sp.	33.3	87.5	0.0	7.1	28.6	0.0	20.0	33.3	0.0
Limonia sp.	3.3	0.0	0.0				26.7	0.0	18.2
Pedicia sp.							10.0	66.7	0.0
Atherix lantha	23.3	62.5	0.0	10.7	28.6	0.0			
A. pachypus							13.3	33.3	0.0
Clinocera sp.							3.3	33.3	0.0
Wiedemannia sp.							6.7	33.3	0.0
Chelifera/Metachela sp.				3.6	14.3	0.0			
Hemerodromia sp.	83.3	87.5	80.0	75.0	71.4	100.0	36.7	0.0	54.5
Neoplasta sp.							33.3	66.7	0.0
Ephydridae							3.3	33.3	0.0
Muscidae							6.7	0.0	9.1
Sciomyzidae	3.3	0.0	0.0						
Myxosargus sp.				3.6	14.3	0.0			
Eristalis sp.							6.7	0.0	0.0
Chrysops sp.	3.3	0.0	0.0						
Tabanus sp.				7.1	28.6	0.0			

APPENDIX 2. Minimum and maximum values for selected site characteristics and assemblage metrics (see Table 1 for explanation of abbreviations).

	BOS		BIR		SLC	
Basin characteristics	Minimum	Maximum	Minimum	Maximum	Minimum	Maximum
Basin area (km^2)	46	125	5	66	4	1,764
Stream order	2	5	2	4	2	6
Population density (no./km^2)	25	1,261	10	1,543	13	2,251
Urban intensity (UII)	0	100	0	100	0	100
Sampling reach characteristics						
Area (m^2)	750	1,879	628	4,051	99	3,511
Riffle (%)	18	64	5	77	3	92
Gradient (m/m)	0.001	0.016	<0.001	0.007	0.004	0.166
Dominant substrate (size-class)	5.1	8.6	2.3	8.8	0.5	8.0
Fines (%)	0	45	0	79	2	21
Silt cover (%)	0	15	0	45	0	100
Canopy closure (%)	77	100	51	100	40	80
Richness metrics						
Total	37	98	32	65	31	62
EPTr	4	36	2	21	1	24
PLECOr	0	6	0	3	0	9
TRICHr	3	17	2	10	1	13
COLEOPr	0	14	1	10	0	7
DIPr	10	32	13	26	12	27
CHr	7	28	9	22	10	23
ORTHO_CHr	0.20	0.74	0.14	0.67	0.18	0.59
NONINSr	6	19	4	13	3	14
ODIPNIr	10	22	7	17	10	19
MOLCRUr	2	12	2	9	0	7
OLIGOr	0	4	1	5	1	5
Density metrics						
DENSITY	2,177	29,699	1,939	12,390	1,673	88,832
EPT	544	14,302	146	9,839	2	26,725
PLECO	0	1,254	0	243	0	1,358
TRICH	524	13,870	67	4,315	2	22,021
COLEOP	0	2,965	0	5,001	0	4,516
DIP	822	14,947	92	5,774	806	40,417
CH	296	12,368	0	5,678	538	33,642
ORTHO_CH	0	1	0	1	0.12	0.93
NONINS	97	9,261	181	8,852	195	60,368
ODIPNI	446	11,840	371	8,982	422	60,686
MOLCRU	0	3,773	9	8,723	0	59,741
OLIGO	0	4,944	0	908	2	12,042
Tolerance metrics						
TOL	3.2	6.4	3.4	6.6	3.1	6.9
TOLr	3.9	6.5	4.3	6.5	3.8	6.5
Diversity index						
SHANND	0.68	1.49	0.46	1.23	0.36	1.31

APPENDIX 3. Spearman rank correlations between the UII and assemblage metrics based on richness and percent richness. Metrics in bold type are strongly correlated ($|\rho| \geq 0.7$) with the UII. Metrics marked with an ampersand (@) are among the 12 metrics most strongly correlated with the UII in each study area. Metrics marked with an asterisk (*) were used to construct the B-IBI-f for each study area. NA indicates that the metric could not be calculated in that study. Abbreviations are explained in Table 1.

	Quantitative (RTH) samples			Qualitative (QRC) data		
Metric	BOS	BIR	SLC	BOS	BIR	SLC
Richness						
RICH	**@*–0.85**	–0.41	–0.39	**@*–0.86**	**@*–0.78**	*–0.51
EPTr	**@*–0.89**	**@*–0.71**	*–0.47	**@*–0.88**	**@*–0.83**	*–0.53
EPT_CHr	–0.24	**@*–0.69**	*–0.48	*–0.63	**@*–0.77**	@*–0.57
EPEMr	**@*–0.82**	**@*–0.77**	–0.25	**@*–0.83**	**@*–0.80**	–0.19
PLECOr	**@*–0.88**	*–0.49	@*–0.63	***–0.68**	–0.39	@*–0.61
PTERYr	NA	NA	*–0.46	NA	NA	*–0.46
TRICHr	**@*–0.83**	–0.39	*–0.49	**@*–0.86**	***–0.65**	@*–0.60
ODONOr	*–0.53	–0.11	0.06	***–0.70**	–0.32	0.07
COLEOPr	***–0.66**	*–0.61	**@*–0.65**	*–0.56	**@*–0.72**	@*–0.57
DIPr	***–0.82**	0.24	–0.13	***–0.78**	–0.11	–0.25
CHr	***–0.81**	*0.46	<–0.01	***–0.73**	0.03	0.03
ORTHOr	***–0.79**	0.32	0.04	***–0.73**	0.28	0.02
ORTHO_CHr	0.13	0.01	0.06	–0.15	0.33	0.01
TANYr	***–0.65**	0.20	–0.31	*–0.58	–0.02	–0.23
TANY_CHr	–0.27	–0.23	–0.35	–0.35	–0.03	–0.24
NCHDIPr	*–0.45	–0.31	*–0.47	***–0.69**	–0.41	*–0.54
NONINSr	*0.62	*0.45	0.33	***0.78**	0.34	0.40
ODIPNIr	*0.52	0.13	0.14	*0.65	–0.09	0.05
MOLCRUr	***0.71**	–0.06	–0.03	***0.78**	–0.21	0.13
GASTROr	*0.52	0.32	–0.04	***0.71**	0.41	0.24
BIVALr	0.27	–0.15	–0.04	*0.49	–0.15	–0.25
CORBICr	NA	–0.09	NA	NA	–0.09	NA
AMPHIr	*0.59	0.13	–0.25	0.23	–0.40	0.03
ISOPr	*0.54	–0.06	0.18	*0.45	–0.18	0.11
OLIGOr	–0.01	*0.61	0.42	0.23	*0.46	**@*0.78**
% richness						
EPTrp	*–0.62	**@*–0.66**	–0.40	**@*–0.84**	**@*–0.77**	*–0.51
EPT_CHrp	–0.24	**@*–0.70**	*–0.49	*–0.62	**@*–0.77**	@*–0.57
EPEMrp	*–0.64	**@*–0.69**	–0.15	***–0.75**	**@*–0.75**	–0.04
PLECOrp	***–0.78**	*–0.49	@*–0.64	*–0.58	–0.38	@*–0.60
PTERYrp	NA	NA	*–0.47	NA	NA	*–0.46
TRICHrp	–0.32	–0.25	–0.38	***–0.74**	*–0.48	*–0.51
ODONOrp	–0.35	–0.10	0.07	–0.40	0.12	0.07
COLEOPrp	–0.35	*–0.54	@*–0.57	–0.24	*–0.57	*–0.48
DIPrp	–0.29	*0.52	0.37	–0.09	*0.56	0.42
CHrp	–0.43	*0.58	*0.47	–0.08	**@*0.65**	**@*0.66**
ORTHOrp	–0.27	*0.47	0.44	–0.22	*0.61	*0.49
ORTHO_CHrp	0.15	<0.01	0.04	–0.15	0.34	0.02
TANYrp	–0.42	0.40	–0.16	–0.33	0.32	–0.04
TANY_CHrp	–0.26	–0.23	–0.35	–0.34	–0.04	–0.24
NCHDIPrp	0.31	–0.18	–0.29	–0.14	–0.15	–0.39
NONINSrp	***0.81**	**@*0.68**	@*0.50	**@*0.91**	*0.64	@*0.63
ODIPNIrp	***0.81**	*0.51	0.37	**@*0.85**	*0.56	*0.53
MOLCRUrp	**@*0.86**	0.10	0.02	**@*0.90**	0.29	0.26
GASTROrp	*0.63	*0.47	–0.04	**@*0.82**	**@*0.65**	0.37

APPENDIX 3. Continued.

Metric	Quantitative (RTH) samples			Qualitative (QRS) data		
	BOS	BIR	SLC	BOS	BIR	SLC
BIVALrp	*0.53	–0.11	–0.04	***0.77**	<–0.01	–0.09
CORBICrp	NA	0.06	NA	NA	0.10	NA
AMPHIrp	***0.65**	0.13	–0.25	*0.61	–0.37	0.07
ISOPrp	*0.58	–0.01	0.20	***0.69**	–0.04	0.22
OLIGOrp	0.37	**@*0.71**	@*0.52	*0.61	*0.63	**@*0.75**
Functional groups						
Richness						
PAr	–0.02	0.21	–0.10	0.04	0.31	–0.14
PRr	***–0.80**	–0.22	–0.29	***–0.69**	*–0.53	–0.44
OMr	***0.67**	–0.15	–0.16	0.30	*–0.63	–0.14
GCr	***–0.75**	0.15	–0.09	**@*–0.79**	*–0.52	–0.04
FCr	***–0.71**	*–0.46	–0.30	**@*–0.80**	*–0.50	*–0.54
SCr	***–0.80**	*–0.56	–0.43	***–0.71**	**@*–0.69**	–0.44
SHr	–0.33	–0.11	@*–0.56	–0.40	–0.37	*–0.49
PIr	NA	NA	NA	NA	NA	NA
% richness						
PArp	*0.48	0.33	0.15	*0.53	*0.57	0.18
PRrp	–0.33	0.05	0.15	–0.14	0.09	–0.04
OMrp	***0.67**	–0.15	–0.16	***0.78**	–0.32	0.13
GCrp	–0.14	*0.50	0.35	–0.17	0.28	@*0.60
FCrp	0.37	–0.27	–0.04	–0.03	–0.02	–0.42
SCrp	–0.22	*–0.46	–0.32	–0.37	–0.40	–0.26
SHrp	0.13	0.05	@*–0.51	0.21	–0.16	–0.39
PIrp	NA	NA	NA	NA	NA	NA
Tolerance						
TOLr	**@*0.85**	**@*0.70**	@*0.60	**@*0.88**	**@*0.74**	@*0.58
Number of correlations $\|\rho\| \geq 0.7$						
Positive	8	3	0	11	3	3
Negative	15	6	1	17	10	0
Total	23	9	1	28	13	3

Appendix 4. Spearman rank correlations between UII and assemblage metrics based on density and percent density. Metrics in bold type are strongly correlated ($|\rho| \geq 0.7$) with the UII. Metrics marked with an ampersand (@) are among the 12 metrics most strongly correlated with the UII in each study area. Metrics marked with an asterisk (*) were used to construct the B-IBI-f for each study area. NA indicates that the metric could not be calculated in that study. Abbreviations are explained in Table 1.

Metric	Density			Metric	% density		
	BOS	BIR	SLC		BOS	BIR	SLC
Taxonomic groupings							
DEN	0.13	–0.44	–0.13				
EPT	0.38	–0.36	–0.07	EPTp	0.34	–0.27	0.02
EPT_CH	0.35	*–0.56	–0.02	EPT_CHp	0.35	*–0.56	–0.02

Appendix 4. Continued.

Metric	Density			Metric	% density		
	BOS	BIR	SLC		BOS	BIR	SLC
EPEM	***–0.76**	–0.40	–0.31	EPEMp	***–0.74**	–0.28	–0.21
PLECO	**@*–0.89**	*–0.49	@*–0.55	PLECOp	**@*–0.90**	*–0.50	*–0.49
PTERY	NA	NA	*–0.47	PTERYp	NA	NA	*–0.47
TRICH	*0.57	–0.31	0.05	TRICHp	*0.61	–0.18	0.17
ODONO	*–0.48	–0.06	0.07	ODONOp	*–0.48	–0.08	0.07
COLEOP	*–0.61	**@*–0.72**	**@*–0.68**	COLEOPp	***–0.70**	–0.62	@*–0.64
DIP	–0.05	0.28	0.02	DIPp	–0.10	*0.61	0.11
CH	–0.18	0.28	0.07	CHp	–0.28	*0.61	0.16
ORTHO	*–0.59	0.38	0.08	ORTHOp	***–0.73**	*0.55	0.20
ORTHO_CH	*–0.54	0.03	0.26	ORTHO_CHp	*–0.54	0.05	0.26
TANY	–0.22	0.12	–0.44	TANYp	–0.22	0.36	–0.35
TANY_CH	–0.03	–0.12	–0.32	TANY_CHp	–0.03	–0.11	–0.32
NCHDIP	0.27	–0.22	–0.13	NCHDIPp	0.19	0.02	–0.14
NONINS	0.23	0.01	0.02	NONINSp	0.27	0.21	0.22
ODIPNI	0.34	–0.10	–0.07	ODIPNIp	0.38	0.19	0.15
MOLCRU	0.43	–0.16	–0.06	MOLCRUp	*0.46	–0.04	–0.02
GASTRO	*0.54	0.01	–0.28	GASTROp	*0.53	0.10	–0.27
BIVAL	0.14	–0.03	–0.11	BIVALp	0.14	–0.03	–0.03
CORBIC	NA	–0.02	NA	CORBICp	NA	0.01	NA
AMPHI	*0.60	0.13	–0.25	AMPHIp	*0.59	0.13	–0.25
ISOP	*0.57	–0.14	0.19	ISOPp	*0.56	–0.03	0.21
OLIGO	0.04	*0.58	0.33	OLIGOp	0.06	*0.61	*0.48
Functional group metrics							
PA	0.10	0.13	–0.09	PAp	0.08	0.20	–0.07
PR	–0.37	0.16	0.09	PRp	*–0.59	**@*0.66**	0.24
OM	***0.67**	–0.16	–0.17	OMp	***0.67**	–0.15	–0.17
GC	*–0.62	–0.04	–0.12	GCp	***–0.66**	0.19	0.05
FC	*0.53	–0.20	–0.02	FCp	*0.64	0.20	0.14
SC	*–0.61	@*–0.63	–0.39	SCp	***–0.68**	*–0.58	*–0.45
SH	–0.09	0.11	0.01	SHp	–0.15	0.33	0.18
PI	NA	NA	NA	PIp	NA	NA	NA
Dominance metrics							
				DOM1	***0.67**	–0.34	0.06
				DOM2	***0.75**	–0.29	–0.01
				DOM3	***0.79**	–0.23	0.01
				DOM4	**@*0.82**	–0.15	0.08
				DOM5	**@*0.84**	–0.12	0.12
Tolerance metrics							
TOL	**@*0.86**	*0.45	0.34				
Number of correlations $\|\rho\| \geq 0.7$							
Positive	8	1	0				
Negative	8	1	1				
Total	16	2	1				

APPENDIX 5. CUII derived optima and tolerance values for invertebrates collected at five or more sites within the Boston (BOS), Birmingham (BIR), and Salt Lake City (SLC) urban study areas. National tolerances are averages of regional tolerance values reported by Barbour et al. (1999) and NCDENR (2003).

	Optima (CUII)			Tolerances (CUII)			National
Taxon	BOS	BIR	SLC	BOS	BIR	SLC	tolerance
Minor taxa							
Turbellaria	36.7	55.7	55.1	5.0	7.8	6.3	4.0
Prostoma sp.	41.2			5.7			
Nematoda	24.1	37.1	52.4	2.9	4.7	5.5	5.5
Mollusca							
Campeloma sp.	27.3			3.5			
Hydrobiidae	33.3			4.4			7.0
Fluminicola sp.			52.6			5.6	5.0
Elimia sp.		18.2			1.5		2.7
Ferrissia sp.		44.9			6.0		6.2
Hebetancylus excentricus		50.4			6.9		
Laevapex sp.	41.4			5.8			
*Stagnicol*a sp.			54.7			6.2	8.4
Physella sp.	35.2	46.7	55.7	4.7	6.3	6.5	8.1
Gyraulus sp.			67.4			9.7	6.1
Helisoma sp.	28.5			3.7			
Planorbella sp.	33.3	47.9		4.4	6.5		6.4
Elliptio complanata	19.9			2.3			5.1
Corbicula sp.		28.5			3.2		
Pisidium sp.	25.1		52.6	3.1		5.6	6.8
Musculium sp.	38.7			5.3			5.0
Sphaerium sp.		25.0			2.6		6.1
Oligochaeta							
Megadrile	34.0	35.9	54.7	4.6	4.5	6.2	
Lumbriculidae	24.8	31.5	63.2	3.1	3.8	8.6	7.5
Naididae		54.8	60.4		7.7	7.8	
Dero sp.	26.1			3.3			9.5
Tubificidae	31.5		55.9	4.1		6.5	9.0
Branchiura sowerbyi		30.7			3.6		8.3
Enchytraeidae	26.4		59.7	3.3		7.6	9.9
Glossiphonia complanata	50.7			7.3			
Helobdella stagnalis	48.4		53.1	6.9		5.7	8.6
Erpobdellidae	32.8	45.9	56.9	4.4	6.2	6.8	8.0
Mites							
Acari	22.5	31.5	54.2	2.7	3.7	6.1	
Crustacea							
Cambarus sp.		22.4			2.2		7.6
Orconectes sp.	30.4			4.0			2.6
Orconectes rusticus		34.5			4.3		
Caecidotea sp.	32.8	21.3	56.1	4.4	2.0	6.6	7.7
Lirceus sp.		29.4			3.4		7.9
Crangonyx sp.	44.2	20.1		6.2	1.8		5.9
Synurella sp.	16.6			1.7			
Gammarus sp.	46.1		48.0	6.5		4.3	6.6
Hyalella azteca	19.9		48.4	2.3		4.4	7.9
Ephemeroptera							
Paraleptophlebia sp.	8.4			0.4			1.3
Hexagenia bilineata		17.0			1.3		

APPENDIX 5. Continued.

Taxon	Optima (CUII)			Tolerances (CUII)			National tolerance
	BOS	BIR	SLC	BOS	BIR	SLC	
Caenis sp.	20.9			2.4			6.3
Caenis anceps		14.7			0.9		
Eurylophella sp.	8.6			0.4			3.5
Eurylophella aestiva		11.0			0.3		
Serratella deficiens	11.6	11.9		0.9	0.4		2.2
Serratella serrata	10.0			0.7			1.4
Tricorythodes sp.	18.0	28.7	55.2	1.9	3.3	6.3	4.2
Centroptilum/Procloeon sp.	12.2			1.0			
Acerpenna pygmaea	14.1			1.3			3.4
Baetis flavistriga	20.7	31.9		2.4	3.8		4.4
Baetis intercalaris		24.8			2.6		4.9
Baetis tricaudatus			54.7			6.2	1.6
Plauditus sp.	21.9			2.6			
Pseudocloeon sp.	18.4			2.0			3.4
Epeorus sp.	8.0			0.3			0.4
Epeorus longimanus			38.6			1.7	0.0
Stenacron interpunctatum		11.5			0.4		6.9
Stenonema modestum/smithae	26.1			3.3			
Stenonema pulchellum		16.6			1.2		2.3
Stenonema vicarium	7.1			0.2			1.9
Isonychia sp.	14.4	17.7		1.4	1.4		2.3
Odonata							
Calopteryx maculata	18.1			2.0			
Argia fumipennis	19.8			2.2			
Argia sedula		32.5			3.9		8.5
Argia vivida			57.8			7.1	
Aeshna sp.	24.3			3.0			5.0
Boyeria grafiana	16.5			1.7			6.1
Boyeria vinosa	20.8	23.7		2.4	2.4		3.8
Helocordulia uhleri	8.7			0.4			4.9
Gomphus sp.	14.7	32.8		1.4	4.0		5.4
Hagenius brevistylus	11.7	29.6		0.9	3.4		2.5
Ophiogomphus sp.	13.7			1.2			2.5
Stylogomphus albistylus	12.1			1.0			4.7
Macromia illinoiensis		25.1			2.7		
Plecoptera							
Leuctra sp.	17.2			1.8			0.3
Sweltsa sp.			40.9			2.4	
Acroneuria lycorias	17.5			1.9			2.3
Hesperoperla pacifica			33.5			0.3	1.0
Perlesta sp.		17.3			1.3		4.7
Paragnetina media	16.1			1.6			2.1
Pteronarcys californica			40.7			2.3	0.0
Hemiptera							
Sigara sp.			55.9			6.5	
Aquarius conformis	20.0	33.1		2.3	4.0		
Aquarius nyctalis			51.6			5.3	
Aquarius remigis	24.9			3.1			

APPENDIX 5. Continued.

Taxon	Optima (CUII)			Tolerances (CUII)			National tolerance
	BOS	BIR	SLC	BOS	BIR	SLC	
Metrobates hesperius	13.9			1.3			
Trepobates pictus		27.6			3.1		
Rhagovelia distincta			53.8			5.9	
Rhagovelia obesa	23.6	12.7		2.9	0.6		
Megaloptera	16.9	17.8					
Nigronia serricornis				1.8	1.4		2.9
Corydalus cornutus	19.5	26.6		2.2	2.9		4.5
Sialis sp.	18.9	9.5		2.1	0.0		4.8
Trichoptera							
Glossosoma sp.	17.6			1.9			0.8
Protoptila sp.		19.7			1.8		1.5
Hydroptila sp.	28.6	38.3		3.7	4.9		5.5
Hydroptila arctia			55.8			6.5	6.0
Rhyacophila coloradensis group			47.6			4.2	
Rhyacophila brunnea			46.2			3.8	
Rhyacophila fuscula	10.5			0.7			0.9
Chimarra sp.	23.1	24.2		2.8	2.5		3.6
Dolophilodes distinctus	10.0			0.7			
Arctopsyche grandis			52.7			5.6	2.0
Ceratopsyche bronta	10.6			0.7			3.7
Ceratopsyche cockerelli			39.5			2.0	
Ceratopsyche oslari			38.2			1.6	
Ceratopsyche sparna	16.9	22.0		1.8	2.1		2.5
Cheumatopsyche sp.	28.1	32.4		3.6	3.9		4.8
Hydropsyche depravata group	25.9	31.3		3.2	3.7		
Hydropsyche occidentalis			58.9			7.4	4.0
Macrostemum sp.	19.5			2.2			3.2
Polycentropus sp.	17.3	12.1		1.8	0.5		4.8
Lype diversa	15.3	15.2		1.5	1.0		3.0
Brachycentrus americanus			44.3			3.3	
Brachycentrus numerosus	9.9			0.6			1.4
Micrasema sp.	14.7			1.4			1.5
Micrasema wataga		14.5			0.9		2.3
Lepidostoma sp.	12.6		41.8	1.1		2.6	1.0
Pycnopsyche sp.	18.5			2.0			3.5
Neophylax sp.	21.5			2.5			2.7
Oecetis sp.	23.8			2.9			6.3
Molanna sp.	16.1			1.6			6.0
Psilotreta labida	7.7			0.3			0.0
Helicopsyche borealis	7.0			0.2			2.0
Lepidoptera	21.2			2.5			
Petrophila sp.		38.8			5.0		3.3
Coleoptera							
Agabus sp.			62.1			8.2	7.3
Dineutus discolor	18.2	15.9		2.0	1.1		
Gyrinus lecontei	8.4			0.4			
Gyrinus marginellus	15.7			1.6			
Peltodytes sp.		33.8			4.1		6.9

APPENDIX 5. Continued.

Taxon	Optima (CUII)			Tolerances (CUII)			National tolerance
	BOS	BIR	SLC	BOS	BIR	SLC	
Ametor scabrosus			47.9			4.3	
Berosus sp.		53.0			7.4		6.7
Sperchopsis tessellata	16.4			1.7			
Helichus striatus			47.8			4.3	
Ancyronyx variegata	33.8	22.8		4.5	2.3		
Dubiraphia sp.	22.1	21.2		2.6	2.0		5.3
Heterlimnius corpulentus			52.1			5.5	
Macronychus glabratus	22.7	17.3		2.7	1.3		3.8
Microcylloepus pusillus	24.2	24.7		3.0	2.6		2.4
Narpus concolor			46.6			3.9	4.0
Optioservus castanipennis		37.3				1.4	
Optioservus ovalis	14.2			1.3			2.4
Optioservus quadrimaculatus			45.6			3.7	4.0
Optioservus trivittatus		20.9			1.9		
Oulimnius latiusculus	16.9	21.2		1.8	2.0		1.8
Promoresia tardella	11.3			0.9			1.0
Stenelmis crenata	23.5	30.9		2.8	3.6		
Zaitzevia parvula			44.3			3.3	4.0
Ectopria sp.	7.7			0.3			4.5
Psephenus herricki	16.4	17.4		1.7	1.4		3.3
Diptera							
Probezzia sp.			57.8			7.1	6.0
Phaenopsectra/Tribelos sp.	27.0	31.0		3.4	3.7		
Chironomus sp.		36.3	63.4		4.6	8.6	9.5
Cryptochironomus sp.	32.9	33.1	58.6	4.4	4.0	7.3	6.8
Dicrotendipes sp.		41.1	58.4		5.4	7.2	7.4
Microtendipes sp.	15.2	11.5		1.5	0.3		6.2
Paratendipes sp.	20.2	17.0	57.4	2.3	1.3	6.9	7.0
Phaenopsectra sp.	19.2	30.3	56.3	2.1	3.5	6.6	6.9
Polypedilum sp.	25.9	32.1	53.0	3.2	3.8	5.7	6.0
Stenochironomus sp.	25.6	21.9		3.2	2.1		5.0
Tribelos sp.	11.5	22.3		0.9	2.2		5.4
Pseudochironomus sp.		25.0			2.6		5.0
Micropsectra/Tanytarsus sp.	18.8		50.7	2.1		5.1	
Micropsectra sp.	11.8		50.4	0.9		5.0	5.2
Paratanytarsus sp.		30.0			3.5		6.1
Rheotanytarsus sp.	25.1	31.0	50.5	3.1	3.7	5.0	5.4
Stempellina sp.	10.9			0.8			1.3
Stempellinella sp.	9.2	16.5		0.5	1.2		3.8
Tanytarsus sp.	22.1	35.4		2.6	4.4		5.7
Pagastia sp.	16.7		52.1	1.7		5.5	1.3
Cricotopus/Orthocladius sp.	25.8	38.5	59.4	3.2	4.9	7.5	
Eukiefferiella/Tvetenia sp.	21.9			2.6			
Brillia sp.	20.0		55.6	2.3		6.4	5.0
Cardiocladius sp.	33.7	41.0		4.5	5.4		5.3
Corynoneura sp.	30.0			3.9			6.1
Cricotopus bicinctus group	29.2	43.6	57.1	3.8	5.8	6.9	7.4
Cricotopus trifascia group		58.4				7.2	4.9

APPENDIX 5. Continued.

Taxon	Optima (CUII)			Tolerances (CUII)			National tolerance
	BOS	BIR	SLC	BOS	BIR	SLC	
Eukiefferiella sp.	22.6		55.5	2.7		6.4	8.0
Nanocladius sp.	24.2		56.6	3.0		6.7	4.3
Orthocladius lignicola	10.9			0.8			
Parachaetocladius sp.	8.2			0.4			2.7
Parametriocnemus sp.	21.7	20.7	55.8	2.5	1.9	6.5	4.3
Rheocricotopus sp.	24.7	36.9	55.6	3.0	4.7	6.4	6.1
Synorthocladius sp.		44.4			5.9		2.8
Thienemanniella sp.	22.8	22.5	54.4	2.7	2.2	6.1	5.5
Tvetenia sp.	20.5	26.9	43.7	2.3	3.0	3.1	5.0
Xylotopus par	10.7			0.8			4.0
Prodiamesa sp.			42.5			2.8	3.0
Odontomesa sp.			58.7			7.3	
Radotanypus submarginella			52.5			5.6	
Thienemannimyia group	25.7	33.1	54.3	3.2	4.0	6.1	6.0
Ablabesmyia sp.	26.7	32.5		3.4	3.9		6.8
Nilotanypus sp.	20.6			2.4			5.3
Pentaneura sp.	30.2			3.9			5.6
Procladius sp.	35.3	38.0		4.8	4.8		8.5
Simulium sp.	25.9	31.3	55.0	3.2	3.7	6.3	5.2
Tipula sp.	24.3	22.7	55.6	3.0	2.3	6.4	5.3
Antocha sp.	22.6	27.9	40.1	2.7	3.1	2.1	3.1
Dicranota sp.	8.5			0.4			2.0
Hexatoma sp.	8.1		48.1	0.3		4.4	2.5
Limonia sp.			56.1			6.6	7.2
Atherix lantha	8.1			0.3			2.4
Hemerodromia sp.	25.4	35.5	61.9	3.2	4.4	8.2	6.0
Neoplasta sp.			43.8			3.2	