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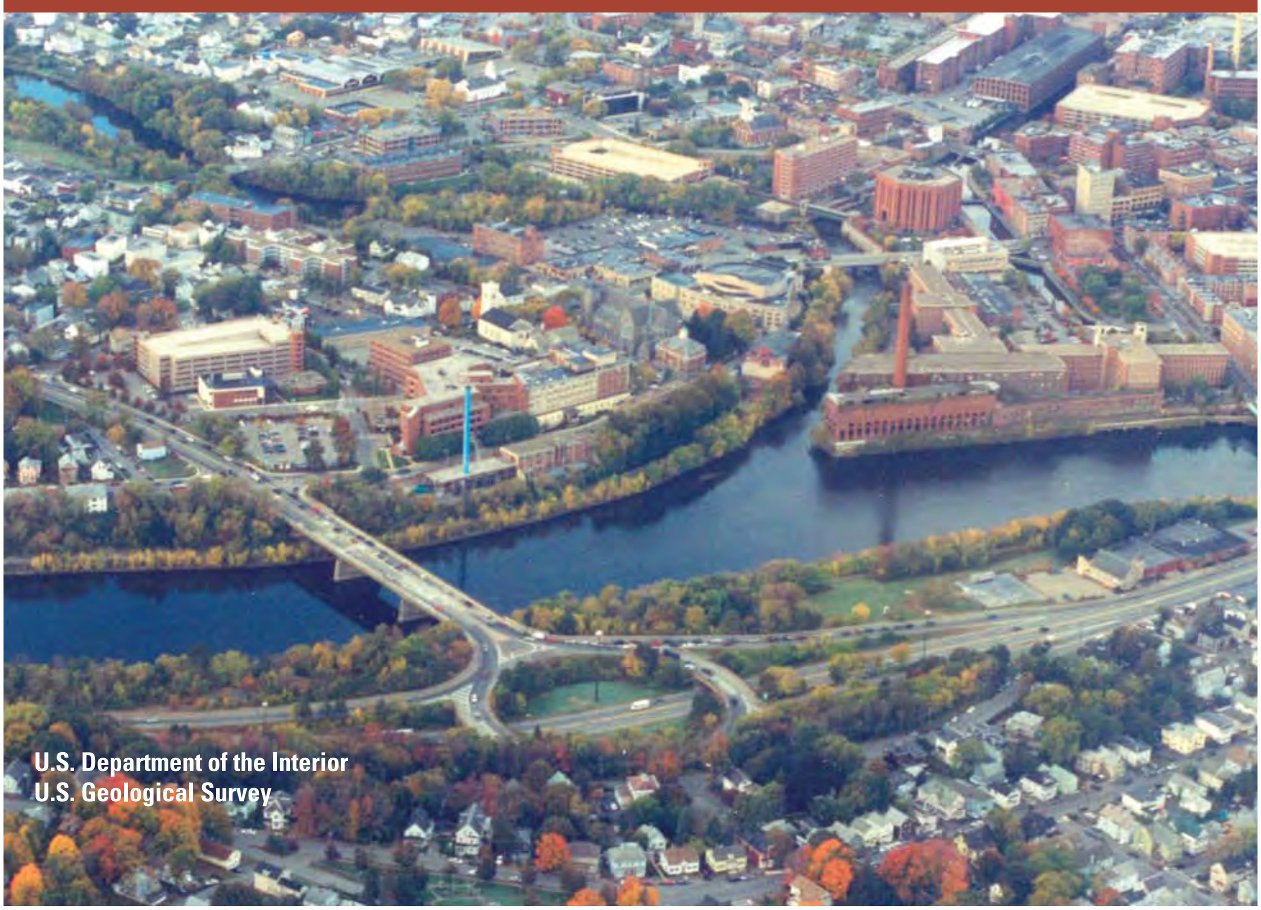
# The Effects of Urbanization on the Biological, Physical, and Chemical Characteristics of Coastal New England Streams

Professional Paper 1695



U.S. Department of the Interior
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**Cover.** The confluence of the Merrimack and Concord Rivers, at Lowell, Massachusetts.
Photo by Keith W. Robinson, U.S. Geological Survey.

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By James F. Coles, Thomas F. Cuffney,
Gerard McMahon, and Karen M. Beaulieu

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# FOREWORD

The U.S. Geological Survey (USGS) is committed to serve the Nation with accurate and timely scientific information that helps enhance and protect the overall quality of life, and facilitates effective management of water, biological, energy, and mineral resources. (http://www.usgs.gov/). Information on the quality of the Nation's water resources is of critical interest to the USGS because it is so integrally linked to the long-term availability of water that is clean and safe for drinking and recreation and that is suitable for industry, irrigation, and habitat for fish and wildlife. Escalating population growth and increasing demands for the multiple water uses make water availability, now measured in terms of quantity *and* quality, even more critical to the long-term sustainability of our communities and ecosystems.

The USGS implemented the National Water-Quality Assessment (NAWQA) Program to support national, regional, and local information needs and decisions related to water-quality management and policy. (http://water.usgs.gov/nawqa/). Shaped by and coordinated with ongoing efforts of other Federal, State, and local agencies, the NAWQA Program is designed to answer: What is the condition of our Nation's streams and ground water? How are the conditions changing over time? How do natural features and human activities affect the quality of streams and ground water, and where are those effects most pronounced? By combining information on water chemistry, physical characteristics, stream habitat, and aquatic life, the NAWQA Program aims to provide science-based insights for current and emerging water issues and priorities. NAWQA results can contribute to informed decisions that result in practical and effective water-resource management and strategies that protect and restore water quality.

Since 1991, the NAWQA Program has implemented interdisciplinary assessments in more than 50 of the Nation's most important river basins and aquifers, referred to as Study Units. (http://water.usgs.gov/nawqa/nawqamap.html). Collectively, these Study Units account for more than 60 percent of the overall water use and population served by public water supply, and are representative of the Nation's major hydrologic landscapes, priority ecological resources, and agricultural, urban, and natural sources of contamination.

Each assessment is guided by a nationally consistent study design and methods of sampling and analysis. The assessments thereby build local knowledge about water-quality issues and trends in a particular stream or aquifer while providing an understanding of how and why water quality varies regionally and nationally. The consistent, multi-scale approach helps to determine if certain types of water-quality issues are isolated or pervasive, and allows direct comparisons of how human activities and natural processes affect water quality and ecological health in the Nation's diverse geographic and environmental settings. Comprehensive assessments on pesticides, nutrients, volatile organic compounds, trace metals, and aquatic ecology are developed at the national scale through comparative analysis of the Study-Unit findings. (http://water.usgs.gov/nawqa/natsyn.html).

The USGS places high value on the communication and dissemination of credible, timely, and relevant science so that the most recent and available knowledge about water resources can be applied in management and policy decisions. We hope this NAWQA publication will provide you the needed insights and information to meet your needs, and thereby foster increased awareness and involvement in the protection and restoration of our Nation's waters.

The NAWQA Program recognizes that a national assessment by a single program cannot address all water-resource issues of interest. External coordination at all levels is critical for a fully integrated understanding of watersheds and for cost-effective management, regulation, and conservation of our Nation's water resources. The Program, therefore, depends extensively on the advice, cooperation, and information from other Federal, State, interstate, Tribal, and local agencies, non-government organizations, industry, academia, and other stakeholder groups. The assistance and suggestions of all are greatly appreciated.

**Robert M. Hirsch**

**Associate Director for Water**

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# Figures



# Tables

# CONVERSION FACTORS, DATUMS, ACRONYMS AND ABBREVIATIONS

| Multiply | By | To obtain |
|---|---|---|
| hectare (ha) | 2.471 | acre |
| hectare (ha) | 0.003861 | square mile ($mi^2$) |
| inch (in.) | 2.54 | centimeter (cm) |
| inch (in.) | 25.4 | millimeter (mm) |
| kilometer (km) | 0.6214 | mile (mi) |
| kilometer per square kilometer ($km/km^2$) | | mile per square mile ($mi/mi^2$) |
| meter (m) | 1.094 | yard (yd) |
| millimeter (mm) | 0.03937 | inch (in.) |
| square kilometer ($km^2$) | 247.1 | acre |
| square kilometer ($km^2$) | 0.3861 | square mile ($mi^2$) |
| square meter ($m^2$) | 0.0002471 | acre |
| square meter ($m^2$) | 10.76 | square foot ($ft^2$) |

Vertical coordinate information is referenced to the National Geodetic Vertical Datum of 1929 (NGVD29).

Horizontal coordinate information is referenced to the North American Datum of 1927 (NAD27).

| | |
|---|---|
| AI | Autotrophic Index |
| AFDM | Ash Free Dry Mass |
| ANSP | Academy of Natural Sciences of Philadelphia |
| BPC | Biological, Physical, and Chemical |
| CA | Correspondence Analysis |
| CV | Coefficients of Variation |
| DC | Direct Current |
| DO | Dissolved Oxygen |
| DTH | Depositional Target Habitat |
| EPT | Ephemeroptera, Plecoptera, Trichoptera |
| GIS | Geographic Information System |
| GLM | General Linear Model |
| IDAS | Invertebrate Data Analysis System |
| MNN | Mean nearest neighbor |
| MRLC | Multi-Resolution Land Characteristics |
| NAWQA | National Water-Quality Assessment Program |
| NECB | New England Coastal Basins |
| NNCV | Variability in nearest neighbor distance |
| PCA | Principal Correspondence Analysis |
| PD | Forest-Patch Density |
| PSCV | Patch-size coefficient of variation |
| QMH | Qualitative Multihabitat |
| RTH | Richest Target Habitat |
| TIA | Total Impervious Area |
| TKN | Total Kjeldahl Nitrogen |
| ULUG | Urban Land-use Gradient |
| USEPA | U.S. Environmental Protection Agency |
| USGS | U.S. Geological Survey |
| USFS | U.S. Forest Service |
| WTW | Wissenschaftlich-Technische-Werkstätten |
| WQ Index | Water-Quality Index |
| $\leq$ | Less than or equal to |
| $\geq$ | Greater than or equal to |

# The Effects of Urbanization on the Biological, Physical, and Chemical Characteristics of Coastal New England Streams

By James F. Coles, Thomas F. Cuffney, Gerard McMahon, and Karen M. Beaulieu

## Abstract

During August 2000, responses of biological communities (invertebrates, fish, and algae), physical habitat, and water chemistry to urban intensity were compared among 30 streams within 80 miles of Boston, Massachusetts. Sites chosen for sampling represented a gradient of the intensity of urban development (urban intensity) among drainage basins that had minimal natural variability. In this study, spatial differences were used as surrogates for temporal changes to represent the effects of urbanization over time. The degree of urban intensity for each drainage basin was characterized with a standardized urban index (0–100, lowest to highest) derived from land cover, infrastructure, and socioeconomic variables. Multivariate and multimetric analyses were used to compare urban index values with biological, physical, and chemical data to determine how the data indicated responses to urbanization. Multivariate ordinations were derived for the invertebrate-, fish-, and algae-community data by use of correspondence analysis, and ordinations were derived for the chemical and physical data by use of principal-component analysis. Site scores from each of the ordinations were plotted in relation to the urban index to test for a response. In all cases, the primary axis scores showed the strongest response to the urban index, indicating that urbanization was a primary factor affecting the data ordination.

For the multimetric analyses, each of the biological data sets was used to calculate a series of community metrics. For the sets of chemical and physical data, the individual variables and various combinations of individual variables were used as measured and derived metrics, respectively. Metrics that were generally most responsive to the urban index for each data set included: EPT (Ephemeroptera, Plecoptera, Trichoptera) taxa for invertebrates; cyprinid taxa for fish; diatom taxa for algae; bicarbonate, conductivity, and nitrogen for chemistry; and water depth and temperature for physical habitat. The slopes of the responses generally were higher between the urban index values of 0 to 35, indicating that the greatest change in aquatic health may occur between low and moderate levels of urban intensity. Additionally, many of the responses showed that at urban index values greater than 35, there was a threshold effect where the response variable no longer changed with respect to urban intensity. Recognizing and understanding this type of response is important in management and monitoring programs that rely on decisive interpretations of variable responses. Any biological, physical, or chemical variable that is used to characterize stream health over a gradient of disturbance would not be a reliable indicator when a level of disturbance is reached where the variable does not respond in a predictable manner.

## Introduction

The premise that urbanization of river basins changes ecological characteristics of streams is generally accepted, but the intensity of development that brings about ecological changes and the rates of these changes are less clearly understood (Karr and Chu, 1999). In addition, many aspects of human intervention most strongly associated with these ecological changes have been difficult to ascertain and are not widely recognized. In this report, “urbanization” refers to the anthropogenic process of changing the land cover through urban development, and “urban intensity” specifies the extent to which this process has altered the land cover from its natural form. Single-variable surrogates for urban intensity, such as population density or measures of impervious surface, are often used to represent the urban intensity in a drainage basin, but a variety of factors can be responsible for the associated ecological disturbances. This fact makes it difficult to predict how ecological components of a stream will respond to specific aspects of urbanization, particularly in different geographical locations across the United States. To address these issues, a combination of univariate, multivariate, and multimetric approaches is necessary to analyze and model responses to urbanization.

Landscape changes caused by human alterations have broad implications for the health of aquatic ecosystems (Naiman and others, 1995). Drinking-water supplies, aquatic life, and fishing can all be affected by human alterations to the landscape. Although urban land use is only a small component of human-engendered landscape alteration, it is second only to agriculture as a source of stream impairment in the United States (U.S. Environmental Protection Agency, 2000). Also, its effects are disproportionate to that of agriculture (for each square kilometer of basin area, urban land use impairs 0.154 km of stream length compared to 0.046 km for agricultural land use) (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) at a time when the amount of agricultural land in the United States is decreasing (about 270,000 $km^2$ between 1987 and 1997). Consequently, urbanization is expected to become an increasingly significant source of impairment for streams.

Urbanization affects stream hydrology (Espey and others, 1966; Leopold, 1968; Arnold and Gibbons, 1996; Booth and Jackson, 1997; Otto and others, 2002), geomorphology (Hammer, 1972; Dunne and Leopold, 1978; Morisawa and LaFlure, 1979), water temperature (Galli, 1991), water chemistry (Wilber and Hunter, 1977; Horowitz and others, 1999; U.S. Geological Survey, 1999; U.S. Environmental Protection Agency, 2000), fish communities (Klein, 1979; Steedman, 1988; Limburg and Schmidt, 1990; Schueler and Galli, 1992; Wang and others, 1997; Yoder and others, 1999), and invertebrate communities (Klein, 1979; Jones and Clark, 1987; Schueler and Galli, 1992, Horner and others, 1997; Yoder and others, 1999). Although the number of urban-stream studies has increased substantially and the effects of urbanization are well documented ( Paul and Meyer, 2001) for selected urban areas, there has not been a coordinated effort that uses a common design to examine and compare the effects of urbanization in different environmental settings across the United States. Such an approach would provide a nationally consistent, science-based perspective that would produce information for water-resource managers to use in assessing the effects of urbanization across the diverse environmental regions in the United States.

To address the effects of urbanization on aquatic biological communities, physical habitat, and water chemistry (BPC), the National Water-Quality Assessment (NAWQA) Program of the U.S. Geological Survey (USGS) investigated the relations between the varying urban intensities of drainage basins and stream ecology in three environmental settings associated with major metropolitan areas (Couch and Hamilton, 2002). These three Urban Land-Use Gradient (ULUG) studies were conducted during 2000–01 in (1) the humid Northeast around Boston, Massachusetts; (2) the humid Southeast in and around Birmingham, Alabama; and (3) the semi-arid West around Salt Lake City, Utah. These studies used an *a priori* multimetric index of urban intensity to identify and sample representative gradients of urban intensity within relatively homogenous environmental settings (McMahon and Cuffney, 2000) associated with each urban area. The objectives of these studies were (1) to determine if BPC variables respond to urban intensity as defined by the *a priori* urban land-use gradient index (urban index); (2) to describe how BPC characteristics of streams respond to urban intensity; (3) to determine which BPC variables are useful indicators of urban intensity; and (4) to identify specific characteristics of urbanization that are most strongly associated with BPC responses so that they may be used in constructing better indices of urban intensity.

The purpose of this report is to review the findings of the ULUG study in the humid Northeast. The scope of the report includes discussions on how the landscape in the study area changes with urbanization, which of the land-use variables identified are the most consistent in characterizing urban intensity over a land-use gradient, which of the BPC characteristics of streams show responses to urban intensity, and how the responses vary in their form and function over the urban-intensity gradient.

Multivariate and multimetric analyses were used to compare urban index values with BPC data from 30 sites to determine how the data indicated responses to urbanization. Multivariate ordinations were derived for the invertebrate-, fish-, and algae-community data by use of correspondence analysis and multivariate ordinations were derived for the chemical and physical data by use of principal component analysis. Site scores from each of the ordinations were plotted in relation to the urban index to test for a response. For the multimetric analyses, each of the BPC data sets was used to calculate a series of metrics that characterized the variability in the data set, and these metrics were then plotted in relation to the urban index to test for a response. These ordination- and metric-based responses were then examined individually to determine how specific BPC characteristics were changing with urban intensity, and additionally, the responses were evaluated cumulatively to determine if the *a priori* urban index effectively characterized urban intensity in the humid Northeast. Ultimately, results from this paper can be compared to the results of other ongoing USGS urban-gradient studies (Couch and Hamilton, 2002). Comparisons among these regional studies can then be used to determine how the urban process differs among different regions of the country, and how urbanization affects the BPC characteristics of streams differently in the various regions.

# Study Design, Sample Collection, and Data Analysis

The ULUG design (McMahon and Cuffney, 2000) is based on a multimetric index of urban intensity, termed “urban land-use gradient index” (referred to as the urban index within this report) within the NAWQA program. The urban index defines an *a priori* gradient of urban intensity from low to high. The BPC variables are correlated with the urban index and interpreted in the context of how their response changes with urban intensity. Additionally, BPC responses are correlated with each other to determine if interactions exist. The responses are also correlated with individual variables associated with urbanization to determine which aspects of urbanization drive the responses. This analysis involves a combination of univariate, multivariate, and multimetric methods to develop an understanding of the form of BPC responses, as shown in the shape of response curves and rate of response, and the interrelationships of responses within the stream and within the basin. Derivation of the urban index and selection of the study sites was an iterative process of sequentially reducing the number of candidate index variables and study sites, because the urban index was specifically associated with the degree of urban intensity within the network of study sites.

## Study Design

The study design focused on the following four components that were considered important in characterizing urbanization in drainage basins so that its effects on the BPC conditions of streams could be accurately evaluated: basin characterization, land-cover classification, derivation of the urban index, and site selection. Data specific to each of these areas were used to provide comprehensive information that was essential in quantifying changes on the large- to small-scale landscape, such as the extent of development in a drainage basin, or the effects that the size of the developed area of the drainage basin had on stream health.

### Basin Characterization

A primary feature in the study design was that site basins would be relatively homogeneous with respect to natural characteristics; therefore, any ecological differences could be attributed to urbanization rather than to natural variability. To minimize effects of natural variability, only drainage basins within the U.S. Environmental Protection Agency (USEPA) Level III Ecoregion 59, the Northeastern Coastal Zone Ecoregion (Omernik, 1987, 1995), were considered. In the initial phase of the study design, 206 small drainage basins (about 50–250 $km^2$) that were coincidentally within the boundaries of Ecoregion 59 and the New England Coastal Basin (NECB) NAWQA Program Study Area (Flanagan and others, 1999) were identified as candidate basins. These basins were delineated by using 30-m digital elevation model (DEM) data, which was used in conjunction with geographic information systems (GIS) programs (U.S. Geological Survey, 2000) to determine the population density of the drainage basins. This information aided in the selection of a site network that had drainage basins representing a gradient of land use from low to high urban intensity, and ensured that basin areas were consistent with sites sampled by NAWQA in other land-use and ecological studies (Gilliom and others, 1995).

To ensure a further degree of homogeneity among natural features, candidate basins were additionally categorized by using U.S. Forest Service (USFS) Ecological Units (Keys and others, 1995). The USFS Ecological Unit 221A (Southern New England Coastal Hills and Plain) closely corresponds to the delineation of USEPA Ecoregion 59 and consists of 12 ecological subsections that are analogous to USEPA Level IV Ecoregions (Level IV Ecoregions are not yet delineated for all of Ecoregion 59). The 12 USFS Ecological Subsections, therefore, were used to group candidate basins in categories that represented a more homogeneous set of natural environmental variables than was possible with Ecoregion 59 alone. This provided an important mechanism for reducing the 206 candidate basins down to more manageable subsets of basins having relatively little variability in their natural features. Ecological Subsection 221Ai (Gulf of Maine Coastal Plain) was ultimately selected as the primary area for the site network because it included basins at the high end of the urban index, covered a significant portion of the Boston suburbs, and extended far enough away from Boston to include areas of low urban intensity.

### Land-Cover Classification

Land-cover data were essential for this study because they provided information necessary to quantify the degree of anthropogenic influence within basins, which is ultimately the basis of urbanization. Land-cover classifications were based on data from the Multi-Resolution Land Characteristics (MRLC) consortium, a Federal interagency project to develop mapped land-cover data for the conterminous United States (Loveland and Shaw, 1996; Vogelman and others, 2001; U.S. Geological Survey, 2002). These data were developed by use of a 30-m resolution Landsat Thematic Mapper data set that was collected during the early nineteen nineties. In this study, land-cover data are used at two different levels of characterizations—a more generalized level 1 classification (for example, MRLC_2, the percentage of the basin that is composed of developed land cover) and a level 2 classification (for example, MRLC_21, the percentage of the basin that is composed of low-density residential development).

The MRLC data were used to describe the composition and the configuration of several important land-cover classes in each basin (McGarigal and Marks, 1995). In addition to characterizing land use for derivation of the urban index, the MRLC data were used to determine how landscape composition and configuration changes with urban intensity. Although not a part of the BPC characterization of streams, the underlying processes of urbanization may be made clearer by interpreting how the landscape changes with increasing development within a basin. Landscape composition refers to the presence and number of patches of a particular cover type in a basin (a patch is a discrete area composed entirely of a single land-cover type, such as developed or forested land) in a basin. Landscape configuration refers to the arrangement of land-cover patches in a basin (the density of developed land cover patches in a basin). To determine if there were specific changes to the landscape that were consistent with the urbanization process, land-use data were interpreted by examining how the composition and distribution of land-cover patches changed with an increase in urban intensity.

## Derivation of the Urban Index

To construct the urban index, 53 basin variables based on infrastructure, land cover, socioeconomic factors, and population density were evaluated (table 1). Infrastructure variables were obtained from the USEPA (Toxic Release Inventory sites, U.S. Environmental Protection Agency, 1997; Price and Clawges, 1999; point-source discharges, U.S. Environmental Protection Agency, 1999) and the U.S. Army Corps of Engineers (number of dams, U.S. Army Corps of Engineers, 1996). As previously discussed, land-cover variables were extracted from the MRLC data set. Population, labor, income, and housing characteristics were obtained from United States census data (Geolytics, 1998). Socioeconomic indices (SEI) were derived for each basin by a principal components analysis (PCA) ordination of population, labor, income, and housing-census variables; therefore, each SEI represents site scores along an axis of the PCA.

Of the original 53 infrastructure, land cover, socioeconomic, and population variables that describe the basin landscape (table 1), 24 were used to calculate the urban index. Variables expressed as area, length, and counts were normalized for basin area (areas were normalized to percentages of basin area, lengths to kilometer per square kilometer, and counts to counts/100 $km^2$). A variable was included in the urban index if it was correlated ($|rho| \geq 0.6$) with 1999 population density and if it was uncorrelated ($|rho| \leq 0.5$) with basin area. The urban index was based on range-standardizing each of the 24 variables so that they ranged from 0 to 1. Variables that were negatively correlated with 1999 population density were adjusted by subtracting the range-standardized values from 1 so that all standardized values would increase as population density increased. A mean of the range of standardized variables was calculated for each site, range-standardized, and then multiplied by 100 to produce an index that ranged from 0 (low urban intensity) to 100 (high urban intensity). McMahon and Cuffney (2000) describe in greater detail the process of creating the urban index.

## Site Selection

To help with the site-selection process, an initial urban index and general cartographic information, such as roads, towns, and streams, were determined for each of the 206 candidate basins. This information was used during the summer and fall of 1999 to reconnoiter candidate-basin streams that were wadeable and third- to fifth-order magnitude. Sampling locations were chosen on the basis of similarity of natural features, regardless of the degree of urban intensity within the drainage basin of the site. The primary criteria for selecting a sampling reach for each basin were that the stream reach was free-flowing for 150 m, showed no sign of recent anthropogenic modification, contained riffles for sampling, and had well-defined banks with at least 50-percent mature-vegetation cover. Sampling reaches meeting these criteria were often difficult to locate for streams in more highly urbanized basins; however, the objective for selecting sampling reaches meeting these criteria was to help ensure that the ecological differences within the sampling reach resulted from the degree of urban intensity in the basin (macro-scale) rather than from differences within the reach (meso-scale).

In all, 32 sites were initially selected for the study, but 2 sites were not used in the data analysis because water-management issues that did not fit the study design were later discovered within these basins. Ultimately, 30 sites were chosen that had drainage-basin boundaries either within Ecological Subsection 221Ai or within a subsection adjacent to 221Ai. Of the entire 30-site network, 80 percent of the inclusive basin area was within Subsection 221Ai, and 20 percent was within a subsection adjacent to 221Ai. As a final step, the urban index was rescaled among the 30 sites so that the individual values ranged from 0 to 100. This resulted in a network of 30 sites that represented a broad gradient of urban intensity within approximately an 80-mi radius of the Boston, Massachusetts, metropolitan area (fig. 1, table 2).

# Sample Collection

Within the study design, the three ecological components of biology, chemistry, and physical conditions were expected to show a response in some way to increasing urban intensity. The sampling plan, therefore, necessitated collecting data or monitoring physical parameters in each of these areas to develop a comprehensive data set that could be used to determine the BPC characteristics that were affected most strongly by urbanization.

**Table 1.** Basin variables evaluated in the development of the urban index.

[Variables in bold type were selected for use in the urban index on the basis of their relation to 1999 population density. Explanations of the socioeconomic indices are a generalized characterization of the variables used in the indices. MRLC, Multi-Resolution Land Characteristic; ha, hectare; km, kilometer; $km^2$, square kilometer; m, meter]

| Abbreviation | Explanation | Spearman rank correlations with: Drainage area | Spearman rank correlations with: 1999 population density |
|---|---|---|---|
| | Infrastructure variables | | |
| **ROADDEN** | **Road density in watershed [road length(km)/watershed area($km^2$)]** | 0.162 | **0.964** |
| **PSCNTDEN** | **Number of point source dischargers/100 ha of basin area** | .469 | **.613** |
| **DAMDEN** | **Number of dams/100 ha of basin area** | .403 | **.621** |
| **TRIDEN** | **Number of Toxics Release Inventory sites/100 ha of basin area** | .271 | **.858** |
| | Land-cover variables | | |
| **pBUF_2** | **Percentage of stream buffer (240 m) area in urban land cover** | 0.317 | **0.942** |
| **pBUF_4** | **Percentage of stream buffer (240 m) area in forested land cover** | -.320 | **-.865** |
| pBUF_9 | Percentage of stream buffer (240 m) area in wetland land cover | .301 | .064 |
| pMRLC_1 | Percentage of drainage area in MRLC "level 1" category: water | .055 | .031 |
| **pMRLC_2** | **Percentage of drainage area in MRLC "level 1" category: developed** | .313 | **.965** |
| pMRLC_3 | Percentage of drainage area in MRLC "level 1" category: transitional | .443 | .256 |
| **pMRLC_4** | **Percentage of drainage area in MRLC "level 1" category: forest** | -.303 | **-.939** |
| pMRLC_5 | Percentage of drainage area in MRLC "level 1" category: shrub | .320 | .247 |
| pMRLC_6 | Percentage of drainage area in MRLC "level 1" category: orchard | .128 | .160 |
| pMRLC_8 | Percentage of drainage area in MRLC "level 1" category: agricultural/urban | -.146 | -.366 |
| pMRLC_9 | Percentage of drainage area in MRLC "level 1" category: wetlands | .298 | -.146 |
| pMRLC_11 | Percentage of drainage area in open water | .055 | .031 |
| **pMRLC_21** | **Percentage of drainage area in low-intensity residential** | .271 | **.963** |
| **pMRLC_22** | **Percentage of drainage area in high-intensity residential** | .334 | **.888** |
| **pMRLC_23** | **Percentage of drainage area in commercial/industrial/transportation** | .410 | **.872** |
| pMRLC_31 | Percentage of drainage area in bare rock/sand/clay | .301 | .194 |
| pMRLC_32 | Percentage of drainage area in quarries/strip mines | .396 | .557 |
| pMRLC_33 | Percentage of drainage area in transitional cover | .347 | .037 |
| pMRLC_41 | Percentage of drainage area in deciduous forest | .048 | -.229 |
| **pMRLC_42** | **Percentage of drainage area in evergreen forest** | -.062 | **-.717** |
| **pMRLC_43** | **Percentage of drainage area in mixed forest** | -.101 | **-.767** |
| pMRLC_51 | Percentage of drainage area in deciduous shrubland | .320 | .247 |
| pMRLC_61 | Percentage of drainage area in orchards/vineyards/other | .128 | .160 |
| pMRLC_81 | Percentage of drainage area in pasture/hay | -.034 | -.411 |
| **pMRLC_82** | **Percentage of drainage area in row crops** | -.178 | **-.680** |
| **pMRLC_85** | **Percentage of drainage area in urban/recreational grasses** | .167 | **.695** |
| pMRLC_91 | Percentage of drainage area in woody wetlands | .311 | -.378 |
| pMRLC_92 | Percentage of drainage area in emergent herbaceous wetlands | .146 | .414 |
| | Socioeconomic indices | | |
| SEI_1 | Population density (low), Homeowners (high), Moderately-high household income (high) | 0.066 | -0.100 |
| **SEI_2** | **Housing and population density (high), College educated (high), Number of children (low)** | .015 | **.707** |
| **SEI_3** | **Rural-residential population (high), adults in workforce (high)** | -.131 | **-.878** |
| SEI_4 | Older-age population (high), annual household income exceeding 100,000 dollars (high) | -.136 | .232 |
| **SEI_5** | **Rural population (high), residents above 65 (high), adults in workforce (low)** | -.027 | **-.712** |
| SEI_6 | Minority population (high), Poverty (high), Number of children (high) | .258 | .478 |

**Table 1.** Basin variables evaluated in the development of the urban index.—Continued

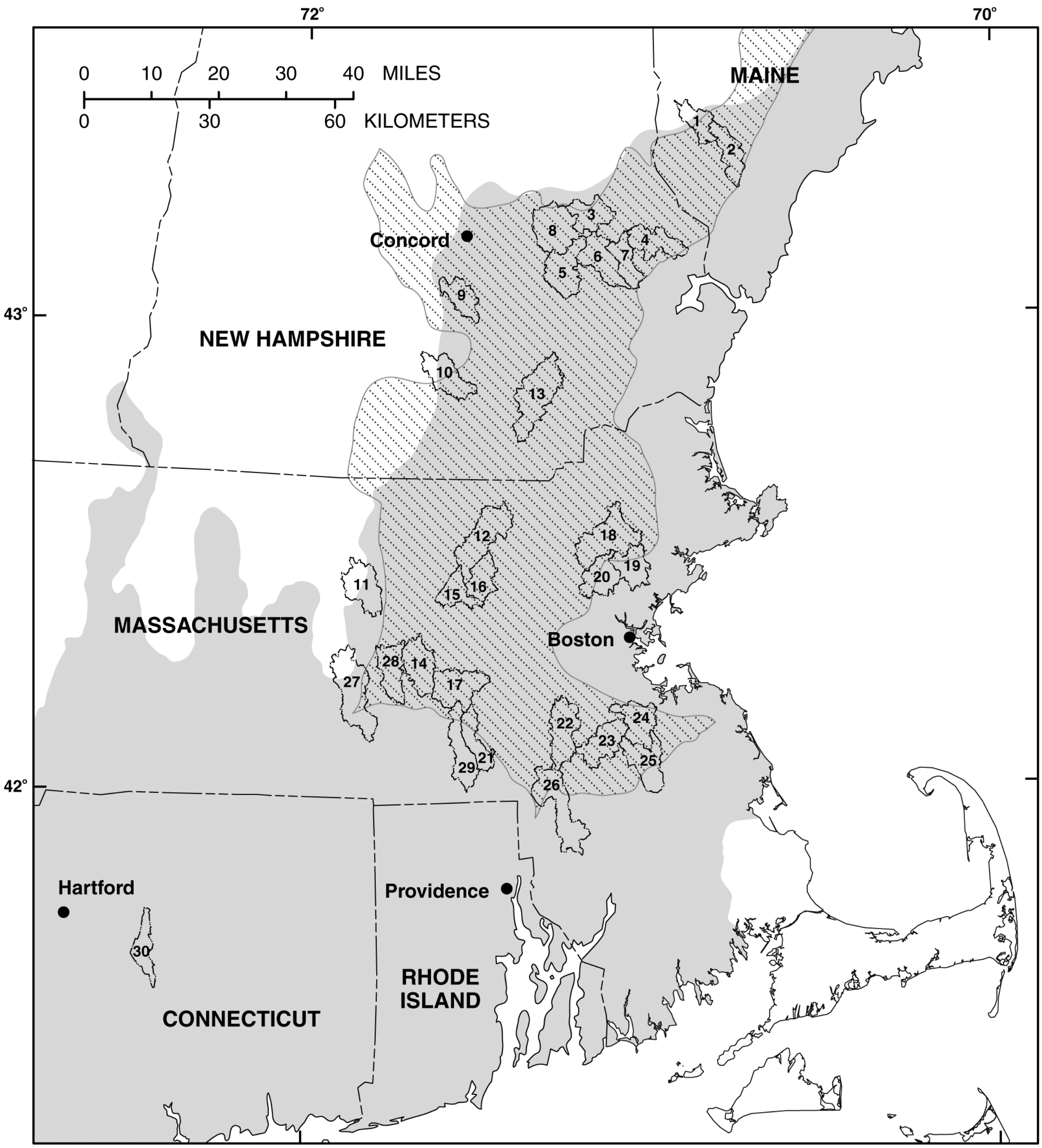
[Variables in bold type were selected for use in the urban index on the basis of their relation to 1999 population density. Explanations of the socioeconomic indices are a generalized characterization of the variables used in the indices. MRLC, Multi-Resolution Land Characteristic; ha, hectare; km, kilometer; km$^2$, square kilometer; m, meter]

| Abbreviation | Explanation | Spearman rank correlations with | |
|---|---|---|---|
| | | Drainage area | 1999 population density |
| | Socioeconomic variables | | |
| AVGBED90 | Average number of bedrooms in a housing unit, 1990 | 0.061 | 0.062 |
| ANNEX99 | Average annual household expenditures, 1999 | .252 | .230 |
| pWRK16 | Percentage of persons 16 and older in the workforce | .018 | -.421 |
| pPOV90 | Percentage of all persons below poverty level | -.084 | -.174 |
| MEDAGE99 | Median age of population, 1999 | -.102 | .553 |
| MEDHHI99 | Median household income, 1999 | .075 | .337 |
| **pFHHF90** | **Percentage of families with female head of household** | .031 | **.722** |
| **pHOUSL80** | **Percentage of 1990 housing units built before 1980** | .026 | **.751** |
| pOWN90 | Percentage of occupied housing units that are owner occupied | .133 | .401 |
| **pRENT90** | **Percentage of occupied housing units that are renter occupied** | .118 | **.690** |
| **pMINR99** | **Percentage of 1999 population of minorities** | .109 | **.811** |
| p65P90 | Percentage of 1990 population 65 years and over | .008 | .574 |
| pMALE99 | Percentage of 1999 population that is male | -.145 | -.399 |
| **POP99DEN** | **Population density, 1999** | .184 | **1.000** |
| **PDEN9099** | **Population density change from 1990–99** | .192 | **.927** |

## Biological Samples

Biological communities were sampled from August 1 to September 1, 2000. Standard NAWQA protocols for sampling of aquatic invertebrate communities (Cuffney and others, 1993), periphytic algae communities (Porter and others, 1993), and fish communities (Meador and others, 1993) were used. Aquatic invertebrates were collected for a quantitative sample from five riffle areas in each sampling reach. A Slack sampler with 0.25-m$^2$ sampling grid and a 425-micron mesh net was used to collect the samples. The five riffle samples were combined for a composite sample and designated as an invertebrate sample representing the richest targeted habitat (RTH) in the reach. A qualitative sample of invertebrates was also collected from the reach by using a 212-micron mesh dip net to sample the various microhabitats along the length of the reach. This composite sample was designated as an invertebrate-QMH (qualitative multihabitat) sample and was intended, in conjunction with the RTH sample, to provide a comprehensive list of invertebrate taxa in each sampling reach. Invertebrate RTH and QMH samples were preserved in 10-percent buffered formalin and sent to the USGS National Water-Quality Laboratory (NWQL) in Denver, CO for taxa identification and enumeration (Moulton and others, 2000).

Periphyton algae were collected quantitatively from five riffle areas in each sampling reach by selecting cobble-size stones and scraping the periphyton from them. The area scraped was determined with a foil template, and the five samples were composited to produce an algae-RTH sample. Aliquots of algae scrapings were taken from the algae-RTH sample to assess community structure and to measure algae biomass as chlorophyll *a* (Chl *a*) and ash free dry mass (AFDM). A second quantitative algae sample was collected from five depositional areas along the sampling reach by inverting a 47-mm petri dish, gently pressing it into the sediment surface, sliding a spatula underneath the petri dish, and then removing the petri dish full of sediment. The five samples were composited and designated an algae-DTH sample, to represent the depositional habitat. Analogous to the invertebrate-QMH sample, a qualitative algae sample (algae-QMH) was collected and composited from various microhabitats along the sampling reach where periphyton growth was observed. Algae RTH, DTH, and QMH samples were preserved in 5-percent buffered formalin and sent to the Academy of Natural Sciences of Philadelphia (ANSP) for taxa identification and enumeration (Charles and others, 2002). The aliquots taken from the algae-RTH samples for the AFDM and Chl *a* analyses were filtered through 45-micron glass-fiber filters to collect the algae cells. The filters were then sealed, packed on dry ice, and sent to the USGS NWQL for analysis.




Basemap from USGS and MassGIS digital graphic sources, with additional overlay information modified from U.S. Department of Agriculture Forest Service, 1995, and from Omernik, J.M., 1987

**EXPLANATION**

17 URBAN LAND-USE GRADIENT BASINS—Site name and location given on table 2

GULF OF MAINE COASTAL PLAIN—U.S. Forest Service Subsection 221Ai

NORTHEASTERN COASTAL ZONE, U.S. ENVIRONMENTAL PROTECTION AGENCY ECOREGION 59

AREA OF OVERLAP BETWEEN COASTAL PLAIN AND COASTAL ZONE

**Figure 1.** Study-area locations for the urban land-use gradient study, Northeastern United States.

**Table 2.** Location of urban land-use gradient sites, basin areas, and their urban index values.

[Site numbers correspond to those numbers given on figure 1. ULUG, Urban Land-Use Gradient; USGS, U.S. Geological Survey; $km^2$, square kilometers]

| USGS station number | ULUG site code | Site number | Site name and location | Basin area ($km^2$) | ULUG index |
|---|---|---|---|---|---|
| 01072540 | LIME | 1 | Little River near Lebanon, ME | 45.8 | 0.0 |
| 01072650 | GREA | 2 | Greatworks River near North Berwick, ME | 60.2 | 13.4 |
| 01072845 | ISIN | 3 | Isinglass River at Batchelder Road near center Strafford, NH | 59.4 | 6.4 |
| 01072904 | BELL | 4 | Bellamy River at Bellamy Road near Dover, NH | 68.5 | 18.3 |
| 01073260 | LAMP | 5 | Lamprey River at Cotton Road near Deerfield, NH | 83.1 | 1.2 |
| 01073458 | NORT | 6 | North River at Rt 152 near Nottingham, NH | 74.9 | 4.9 |
| 010734833 | LINH | 7 | Little River at Cartland Road at Lee, NH | 52.2 | 3.7 |
| 01089743 | LSUN | 8 | Little Suncook River at Blackhall Road at Epsom, NH | 101.4 | 5.7 |
| 01090477 | BLAB | 9 | Black Brook at Dunbarton Road near Manchester, NH | 53.7 | 1.1 |
| 01094005 | BABO | 10 | Baboosic River at Bedford Road near Merrimack, NH | 73.0 | 9.6 |
| 01095220 | STIL | 11 | Stillwater River near Sterling, MA | 78.7 | 14.4 |
| 01096544 | STON | 12 | Stony Brook at School Street at Chelmsford, MA | 107.7 | 36.5 |
| 010965852 | BEAV | 13 | Beaver Brook at North Pelham, NH | 121.7 | 38.1 |
| 01096710 | ASSA | 14 | Assabet River at Allen Street at Northborough, MA | 76.4 | 58.0 |
| 01096945 | ELIZ | 15 | Elizabeth Brook off White Pond Road near Stow, MA | 48.5 | 26.9 |
| 01097270 | FORT | 16 | Fort Pond Brook at River Road near South Acton, MA | 53.7 | 35.9 |
| 01097476 | SUDB | 17 | Sudbury River at Concord Street at Ashland, MA | 89.6 | 36.8 |
| 01101500 | IPSW | 18 | Ipswich River at South Middleton, MA | 115.3 | 61.1 |
| 01102345 | SAUG | 19 | Saugus River at Saugus Ironworks at Saugus, MA | 60.4 | 87.2 |
| 01102500 | ABER | 20 | Aberjona River (head of Mystic River) at Winchester, MA | 58.2 | 100.0 |
| 011032058 | CHAR | 21 | Charles River at Maple Street at North Bellingham, MA | 54.2 | 52.0 |
| 01105000 | NEPO | 22 | Neponset River at Norwood, MA | 84.9 | 55.9 |
| 01105500 | ENEP | 23 | East Branch Neponset River at Canton, MA | 72.9 | 61.3 |
| 01105581 | MONA | 24 | Monatiquot River at River Street at Braintree, MA | 71.2 | 73.0 |
| 01106468 | MATF | 25 | Matfield River at North Central Street at East Bridgewater, MA | 79.8 | 93.3 |
| 01109000 | WADE | 26 | Wading River (head of Threemile River) near Norton, MA | 113.4 | 38.1 |
| 01109595 | MIDD | 27 | Middle River off Sutton Lane at Worcester, MA | 124.7 | 55.7 |
| 01110000 | QUIN | 28 | Quinsigamond River at North Grafton, MA | 66.2 | 85.7 |
| 01112262 | MILL | 29 | Mill River at Summer Street near Blackstone, MA | 73.7 | 22.9 |
| 01193340 | BLAL | 30 | Blackledge River above Lyman Brook near North Westchester, CT | 49.2 | 19.0 |

Fish communities were sampled about a week after invertebrate and algae sampling to allow fish communities to recover from any disturbance caused by the invertebrate and algae sampling. Fish were collected from each reach by making two separate upstream passes with a backpack electro-shocker, producing a direct current (DC) to stun the fish. The stunned fish were then captured with 6-mm (0.25 in.) mesh nets. Captured fish were placed in live wells until they could be identified to the species level, measured for total length, checked for anomalies, and released back to the stream.

## Water Samples

Water-column-chemistry data were collected once in April (spring samples) and once in August (summer samples) 2000 to characterize water-quality conditions before biological samples were collected and at the time biological samples were collected. Water samples were analyzed for nutrients (USGS NWQL schedule 2702) and pesticides (USGS NWQL schedule 2001). Additional samples were collected for sulfate ($SO_4$) at 14 sites. The water samples were collected at equal-width increments across the stream channel and processed on site in accordance with standard NAWQA Program protocols (Shelton, 1994). The samples were then sent to the USGS NWQL for analysis. Field measurements of specific conductance, water temperature, dissolved oxygen, pH, and alkalinity were taken concurrently with the collection of the water samples. Measurements of these field parameters were made with a Wissenschaftlich-Technische-Werkstätten (WTW) ML P4 F multi-parameter field meter.

### Physical Characteristics

Usually within 1 day of collecting the invertebrate and algae samples, measurements of the physical-habitat characteristics of the sampling reaches were made in accordance with the NAWQA habitat protocol (Fitzpatrick and others, 1998). Habitat characteristics were measured at 11 equally spaced transects along the sampling reaches and included measurements of stream velocity, channel depth and width, aspect of flow, bed substrate, habitat cover, bank morphology, canopy closure, and bank vegetation. Means and coefficients of variation (CV) were calculated from these 11 values so that single variables could be used to represent the general habitat conditions of the reaches. Variables that represented overall reach conditions were considered meso-scale habitat characterizations, and variables that represented specific areas in the reach (such as riffles where the RTH samples were collected) were considered micro-scale habitat characterizations. The meso- and micro-scale habitat characterizations, therefore, are distinguished from the basin-scale characterization, which is the basis of the urban index.

Stream-stage measurements were recorded at hourly intervals by USGS streamflow-gaging stations (at 9 sites) or by stage transducers installed for this study (at 21 sites) from May 2000 through June 2001. At some sites, continuous measurements of stream stage were interrupted by low summertime flows that caused stages to fall below the level of the transducer or by freezing during the winter that caused erroneous readings in some instruments. Consequently, summary statistics for stream stage were calculated only for spring 2000, fall 2000, and spring 2001, the periods when all instruments were functioning properly. Summary statistics were based on the CV and skew of stage values, number of rising and falling changes in stage, and the duration of high- and low-stage pulses.

Water-temperature recorders were installed near the stage-measuring instruments at the time of biological sampling. Water temperature was recorded at hourly intervals and was monitored through August 2001 to characterize the annual thermal regime at each site and to compare thermal regimes among sites. Summary statistics were calculated for the year and for each season. Statistics included degree-day and daily mean, median, maximum, minimum, range, standard deviation, and CV.

## Data Analysis

The BPC data were analyzed with two approaches: (1) a multivariate-based analysis that examined the correspondence between ecological gradients derived from ordinations of BPC data and the *a priori* urban gradient defined by the urban index, and (2) a multimetric-based analysis that examined how specific characteristics (metrics) of BPC data responded to the urban index and the relative effectiveness of these metrics in representing urban intensity. Before doing the analyses, the data were reviewed for errors and formatted to a structure appropriate for the data type and the analysis. For the invertebrate- and algae-community data, it was also necessary to resolve the ambiguous taxa within each sample.

### Resolution of Taxa Ambiguities

Although all fish were identified to the species level, invertebrates and algae could not always be identified to consistent taxonomic levels. Taxonomic ambiguities occur when results (abundance or presence) are reported at multiple levels of a taxonomic hierarchy for a group of related taxa. For example, an ambiguity would exist in a sample when abundance data are reported for species (for example, *Hydropsyche sparna* and *Hydropsyche betteni*) and for the genus (*Hydropsyche*), family (Hydropsychidae), or other higher taxonomic levels associated with these species. In this example, the genus for both organisms is the same, but it cannot be determined if the organisms represented by the genus and family belong to these two species or to other unidentified species; therefore, *Hydropsyche* and Hydropsychidae represent ambiguous taxa. How taxonomic ambiguities are resolved will affect metrics based on taxa richness and abundance.

For the invertebrate data, resolving ambiguities was automated with the Invertebrate Data Analysis System software (IDAS; Cuffney, 2003). Ambiguities in the RTH data were resolved by specifying an option in the IDAS program that processes samples separately by site and then distributes the abundance of ambiguous parents among children in proportion to the relative abundance of each child. This procedure maximizes taxa richness without affecting taxa abundance. A more complete qualitative sample was created for each site by using an option in IDAS that creates a combined taxon on the basis of the content of the QMH and RTH samples. Consequently, the QMH data reported in this paper are a combination of taxa found in the QMH and RTH samples. Ambiguities in the QMH data were resolved by specifying an option in the IDAS program that processes samples separately by site and then deletes ambiguous parents and retains their children. Because the QMH data were important only for presence of taxa and not for abundance, it was necessary to retain only the children to preserve taxa richness.

Periphyton algae were generally identified by the ANSP to at least the species level (many were identified to variety), although a few taxa were identified only to genus. As the starting point for reducing ambiguities, lowest-level taxa identifications were set at the species level (that is, algae variety was not considered). Through a series of database queries, ambiguities in the RTH- and DTH-algae data were resolved by processing samples by site and then deleting children of ambiguous parents and adding their abundance to the parent. As was done with the QMH-invertebrate data, the QMH-algae data for each site were augmented with additional taxa found in the RTH and DTH samples. Taxonomic ambiguities in the QMH-algae data were resolved by using database queries to delete ambiguous parents and retain children, which is analogous to the procedure used for the QMH-invertebrate data.

## Multivariate Analysis

For each BPC data set, an indirect gradient analysis (ter Braak, 1995) was used to investigate the relation between an ecological gradient (latent environmental variables) derived from an ordination of the specific data set and (1) the urban index, (2) the derived ecological gradients from other BPC data sets, and (3) the individual variables used to derive the urban index. Each ordination represents the relative differences among sites along a derived ecological gradient that is specific to each BPC data set. These ecological gradients are expressed as ordination axes, where sites with similar compositions are close together along an axis and sites with increasingly different compositions are far apart. The primary ordination axis explains the most variation (structure) in the data. Each succeeding axis explains less of the variation. The strength of each ordination axis in explaining structure within the data is represented by an eigenvalue for each axis; the highest eigenvalue is associated with the primary axis. Because each successive axis explains less of the data structure, it has a correspondingly lower eigenvalue. For this study, responses associated with the first two axes of the ordinations (Axes 1 and 2) were examined on the basis of the premise that, if urban intensity were an important factor in determining community structure, it would be evident in the primary or secondary axes of the ordination.

Correspondence analysis (CA) was used to ordinate biological-community data because the underlying response model was assumed to be unimodal. Separate correspondence analyses were completed for fish, RTH invertebrates, QMH invertebrates, RTH algae, DTH algae, and QMH algae. For correspondence analyses that were based on taxa-abundance data (all but the QMH-data sets), absolute abundances were used for the RTH invertebrates, and relative (percent) abundances were used for the other data. All taxa were used in the ordination, the data were square-root-transformed, and rare species were downweighted before running the analyses with CANOCO 4.0 (ter Braak and Smilauer, 1998). An eigenvalue of 0.3 or greater was considered indicative of an ordination with an acceptable spread of data along the axis.

A PCA was used to ordinate the physical and chemical data because the underlying response model was assumed to be monotonic. The PCA procedure used for the physical and chemical data followed the same general approach as the CA procedure used for the biological-community data. Before running the PCA on the habitat data, however, the ranked variables were correlated (Spearman rank correlation) in relation to each other to test for colinearity. If two or more variables had a |rho| value greater than 0.800, they were reduced to a single proxy variable. To be selected as a proxy variable, a variable had to be a primary (measured) datum and (or) it had to be more normally distributed around the mean than the other variables. Similarly, the water-chemistry data had variables that were essentially redundant for the purpose of PCA analysis. Proxy chemical variables were selected (shown in **bold**) by dropping homologous variables (shown in parentheses): **oxygen saturation** (dissolved oxygen); **bicarbonate** (alkalinity); **total organic N + $NH_4$** ($NH_4$ and $NH_4$ + dissolved organic N); **$NO_2$ + $NO_3$** ($NO_2$); **total phosphorus** (dissolved phosphorus and dissolved $PO_4$). Total organic N + $NH_4$, as reported by the USGS NWQL, is often reported elsewhere as total Kjeldahl nitrogen (TKN), so the TKN notation is used in this report. Data were standardized and centered for the PCA before running the analyses with CANOCO 4.0 (ter Braak and Smilauer, 1998).

Axis 1 and 2 site scores from each BPC ordination were used to represent the two strongest ecological gradients that were derived from the data structure within each BPC data set. Site scores from each axis were scaled from 0 to 100 and their mathematical signs (+ or -) were standardized so that the scores increased or decreased relative to the urban index in an ecologically meaningful manner. In other words, the signs associated with ordination site scores are arbitrary, and it is possible for the values associated with a particular biological data set (for example, invertebrates) to increase as urban intensity increases, although it is recognized that the community structure decreased as urban intensity increased. In this example, the mathematical direction of the site scores would be reversed to reflect this response with urban intensity and to facilitate comparing responses with other data sets.

The indirect gradient analysis was finalized by plotting the standardized site scores (representing the derived ecological gradients) in relation to possible explanatory variables. If the derived gradient was related to the variable, the form (linear, linear with threshold, or curvilinear) and rate of the response were determined. This procedure was used to determine how well the BPC-derived gradients corresponded (1) to the urban index, (2) to each other, and (3) to the individual variables used to develop the urban index. This procedure was used to determine if a derived gradient responds more strongly to the urban index or to other variables that are more ecologically based. Furthermore, comparing the derived gradients to the variables used to develop the urban index helped (1) to identify variables that might provide a better ecological characterization of urban intensity, and (2) to compare results among BPC responses to determine if the same variables were important in defining each response. This approach was used to gain an understanding of the changes associated with urbanization, the effects of urbanization on stream ecology, and to provide information to develop strategies for effectively monitoring urbanization.

Addressing these objectives generated far more scatterplots than could be discussed, so it was necessary to present the results of these analyses in a more condensed fashion. Spearman rank correlation coefficients (rho) were used for this purpose because they provided an effective means of summarizing relations between variables, even when the underlying response was not linear (that is, linear with a threshold or curvilinear). General relations among variables are therefore summarized with correlation coefficients, and only a limited number of the most important relations are plotted. Correlations and scatterplots were done with SYSTAT 8 (SPSS, 1998) and were considered to be strong when |rho| was greater than or equal to 0.7.

## Multimetric Analysis

All metrics were extracted from the BPC data sets as either individual variables or combinations of variables that collectively emphasize specific characteristics, which are thought to be indicators of ecological response to environmental disturbances such as urbanization. Metrics calculated from the biological data were based on taxa composition as measures of abundance, richness, functional group, tolerance, and as indices of diversity. The IDAS program was used to calculate 126 invertebrate metrics that are commonly used in bioassessment (Rosenberg and Resh, 1993; Davis and Simon, 1995; Barbour and others, 1999). Of these metrics, all 126 were relevant to the invertebrate RTH data, and 58 were relevant to the QMH data. For the fish data, a total of 92 metrics were calculated, primarily on the basis of taxa characteristics described in Halliwell and others (1999). For the algae data, a total of 164 metrics were calculated, primarily on the basis of information provided by Stephen Porter (U.S. Geological Survey, written commun., 2002), Marina Potapova (Academy of Natural Sciences of Philadelphia, written commun., 2002), and Molloy (1992). Of these metrics, 146 were relevant to the RTH data, 134 were relevant to the DTH data, and 132 were relevant to the QMH data. These 146 metrics included algae biomass measurements of Chl *a* concentration and AFDM for the RTH data, and included biovolumes for the RTH and DTH data.

Metrics from the physical- and chemical-data sets included the measured variables and metrics derived through various combinations and transformations of the measured variables. Derived metrics also included indices that were developed by assigning categorical values to variables (either singly or in combination) to represent specific habitat characteristics, such as bank stability and habitat heterogeneity. Variables that were not included in the ordination analyses because of redundancy or colinearity were retained as metrics in the multimetric analyses, because unlike the PCA ordinations, there was no statistical justification for excluding these variables.

Each of the BPC metrics was correlated with the urban index by using Spearman rank correlation (rho) as a means of efficiently identifying metrics that might be useful in monitoring urban effects. When |rho| is greater than or equal to 0.7, a scatterplot was made to show the relation between the metric and the urban index. These response curves were examined for trends, thresholds, and statistical relations in the same manner that scatterplots were analyzed in the multivariate analyses. Correlations, scatterplots, and regression analyses were calculated by using SYSTAT 8 (SPSS, 1998).

## Description of the Responses

Identification of thresholds in response curves was an important aspect of the data analysis. Thresholds are points where the underlying relation between variables changes abruptly; therefore, they are important in the management and monitoring of aquatic resources. A hypothetical response curve with a threshold at an urban index value of 30 is shown in figure 2. This response curve indicates that it would not be cost-effective to invest resources to reduce urban intensity from an index value of 80 to 60, because there would be no predictable response and the mitigation effort would be considered a failure. In contrast, reducing urban intensity from an urban index value of 30 to 20 could be cost-effective because it would result in a large response (mitigation). Consequently, using this variable to monitor responses would be effective only at the low end of the urban gradient.

Lowess smoothing was used to help identify apparent thresholds in the response curves and to approximate the urban index values where there were thresholds. A two-slope linear-regression analysis was used to determine if a threshold was significant ($\alpha = 0.05$), and to describe and compare the response on both sides of the threshold. When variable responses were determined not to have thresholds, they were described with a single linear-regression analysis. So that the overall strength of responses could be compared among variables regardless of the form of the response, the second-order equation (nonlinear) was used to calculate a coefficient of determination ($r^2$) for the overall response of variables with thresholds. Lowess smoothing and regression analyses were done with SYSTAT 8.0 (SPSS, 1998).

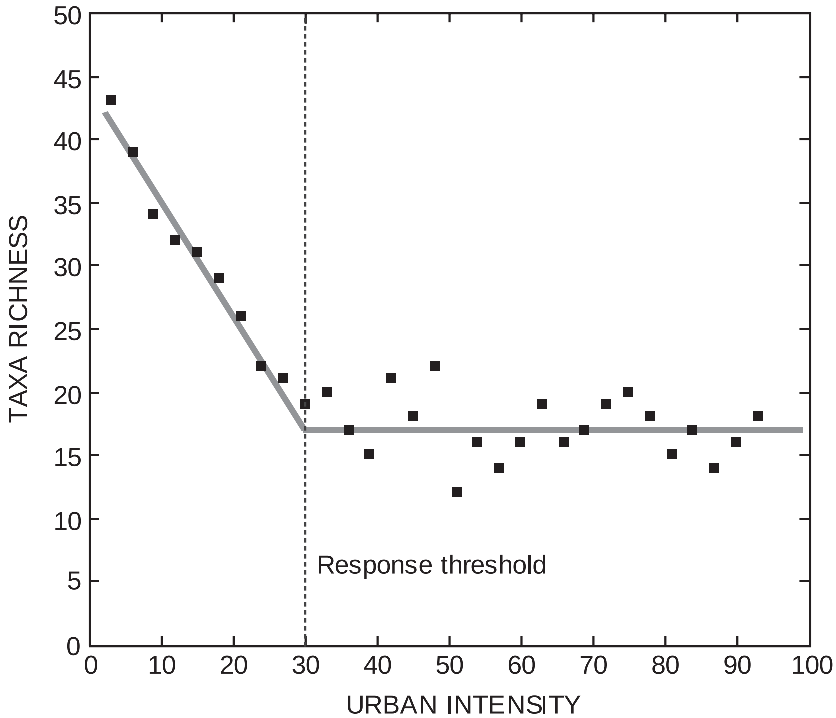



**Figure 2.** A response curve showing a hypothetical relation between taxa richness and urban intensity. This example shows a strong two-slope regression response with a threshold at an urban index value of 30. There is a strong relationship below 30 and a weak response above 30.

# Responses of Biological, Physical, and Chemical Characteristics to Urban Intensity

The biological, physical, and chemical components investigated in this study each had specific elements that responded to increases in urban intensity. Land cover changed from primarily forested to developed land, and concurrently the forested land became more fragmented as developed land patches coalesced. The multivariate ordinations all showed that the primary axis was most strongly related to urban intensity, although the form of the response between the primary axis and the urban index differed among BPC ordinations; often there was a threshold, above which the variable ceased to change with increasing urban intensity. Each of the BPC data sets had metrics that responded to urban intensity (often with a threshold), but the invertebrate metrics appeared to provide the best characterization of changes to the stream brought about by urbanization. Finally, although different categories of basin variables were used to derive the urban index (infrastructure, land cover, and socioeconomic), each category had specific variables that appeared to be important in characterizing urbanization.

## Land-Cover Changes

Urbanization is associated with several distinct patterns in the physical characteristics of forested and developed land-cover patches (table 3). As urban intensity increases (as measured by the urban index), forests are broken up into greater numbers of smaller patches. This change in land use is observed when forest-patch density (PD) increases and when mean forest patch size (MPS) and the percentage of the basin composed of the largest forest-patch (LPATp) decrease concurrently. The direct relation between urbanization and forest fragmentation is further indicated by the positive correlation between the urban index and mean nearest neighbor distance (MNN), indicating that the dispersion of forest patches increases across a drainage basin. Although not as strongly correlated as the MNN was with the urban index, the relation of the variability in nearest neighbor distance (NNCV) increasing with the urban index also indicated that forest fragmentation increased with urbanization. The increase in NNCV of forest patches with the urban index indicates that the distribution of forest patches within a basin becomes more random with urbanization.

**Table 3.** Rho values from Spearman correlations between the urban index and patch characteristics for forest and developed land.

[A patch is a discrete area within a basin that is composed of a single land-cover class (For example, forest and developed land, in this table); bold indicates an absolute value of rho greater than 0.7; MRLC, Multi-Resolution Land Characteristic]

| Variable | Patch characteristic for either forest or developed land-cover class | Developed (MRLC 2) | Forest (MRLC 4) |
|---|---|---|---|
| LANDp | Percentage of basin area in the land-cover class | **0.974** | **-0.977** |
| LPATp | Percentage of basin area comprising largest patch of the class | **.940** | **-.971** |
| PD | Patch density of the class | .268 | **.968** |
| MPS | Mean patch size of the class | **.949** | **-.971** |
| PSCV | Patch-size coefficient of variation for the class | **.860** | .078 |
| MNN | Mean nearest neighbor distance for patches of the class | **-.911** | **.828** |
| NNCV | Nearest neighbor coefficient of variation for the class | -.677 | .685 |

The relation between urbanization and the landscape arrangement of developed land is not entirely the inverse of the relation between forested land and urbanization. The LPATp and MPS for developed-land patches positively correlated with the urban index, but PD was not well correlated (table 3). Interpreted together, these responses indicate that as urban intensity increases, more developed-land patches appear on the landscape, but the overall density of patches (PD) is less evident because urban expansion causes existing patches to become larger and coalesce; in other words, the patches become less fragmented. This process is further supported by the positive correlation between urban index and patch-size coefficient of variation (PSCV), which indicates that the variability of developed-land patch size increases with urbanization. Additionally, MNN had a strong negative correlation with the urban index; this correlation indicates that an increase in urban intensity is associated with a less fragmented spatial structure of developed land-cover patches.

## Biological, Physical, and Chemical Ordinations and Urban Intensity

For each of the BPC ordinations, the primary axis site scores were much more strongly correlated with the urban index than were the secondary axis site scores, even when the eigenvalues of the axis ordinations were relatively low, such as with the algae RTH and QMH data. This finding indicates that urban intensity was the primary factor in defining the data structure in all of the BPC data sets, as represented by the primary derived ordination gradient. Secondary axes were not associated with the urban index; therefore, they represent other ecological gradients. The association between the urban index and the primary BPC gradients supports the use of this index (McMahon and Cuffney, 2000) as a tool for quantifying the intensity of urban development in the Boston area and relating it to changes in BPC characteristics of streams.

The strength of the biological-community responses to urban intensity varied, depending on community (invertebrates, fish, or algae) and sample type (RTH, DTH, or QMH) (table 4). Invertebrate communities, represented separately by RTH and QMH data, had a strong community gradient among sites, as indicated by their ordination eigenvalues. The similarity between the RTH and QMH ordinations is a strong indication that presence-absence data (QMH) are as effective as abundance data (RTH) in expressing the change in invertebrate communities along the urban-intensity gradient. This relation was not seen with the algae data, where the DTH and QMH ordinations had notably lower eigenvalues (0.145 and 0.182, respectively) than the RTH ordination (0.318). This finding may be a result of the accumulation of non-resident algae that washed in from upstream and settled into the depositional habitats. These algae would include sestonic forms coming from upstream impoundments and algae from upstream riverine habitats. The techniques used to count and identify diatoms (Charles and others, 2002) cannot distinguish between living and dead diatoms; therefore, it was not possible to restrict diatom counts in DTH habitats to living diatoms, which would have reduced the abundance of nonresident algae. The QMH samples were composed of algae collected from all habitats, so they would also include any nonresident taxa that were accumulating in depositional habitats. Consequently, the algae community in the QMH and DTH samples would not have been entirely composed of the taxa growing in the reach, whereas algae from the RTH sample were a better representation of the resident community.

The gradient defined by the primary axis of the fish-community ordination was the strongest gradient evident in any of the biological community data (eigenvalue, 0.489), but it did not have the highest correlation with the urban index (table 4). As with the other biological-community data, the primary ordination axis was closely associated with the urban index, but the secondary ordination axis was not. This finding strongly indicates that the distribution of fish, like invertebrates and algae, was affected by urban intensity as measured by the urban index.

In the water-chemistry data, strong gradients were evident among sites in samples collected in the spring and in the summer, although the data collected in the spring had a slightly stronger gradient (primary eigenvalue, 0.647) than the data

**Table 4.** Rho values from Spearman correlations of site scores from primary and secondary ordination axes derived from biological, chemical, and physical measures with the urban index.

[Abund, raw abundance; CA, correspondence analysis; DCA, detrended correspondence analysis; DTH, depositional targeted habitat; PCA, principal-components analysis; Pres/Abs, presence/absence; RelAbund, relative abundance; RTH, richest targeted habitat; QMH, qualitative multihabitat]

| Ordination axis | Ordination type | Ordination eigenvalue | Correlation coefficient |
|---|---|---|---|
| Invertebrates RTH 1 | DCA-Abund | 0.350 | -0.892 |
| Invertebrates RTH 2 | DCA-Abund | .130 | - .244 |
| Invertebrates QMH 1 | DCA-Pres/Abs | .324 | - .912 |
| Invertebrates QMH 2 | DCA-Pres/Abs | .162 | - .021 |
| Fish 1 | CA-RelAbund | .489 | -.805 |
| Fish 2 | CA-RelAbund | .241 | -.184 |
| Algae RTH 1 | CA-RelAbund | .318 | - .643 |
| Algae RTH 2 | CA-RelAbund | .242 | - .219 |
| Algae DTH 1 | CA-RelAbund | .145 | - .634 |
| Algae DTH 2 | CA-RelAbund | .137 | - .022 |
| Algae QMH 1 | CA | .182 | - .632 |
| Algae QMH 2 | CA | .151 | - .009 |
| Habitat 1 | PCA | .208 | - .820 |
| Habitat 2 | PCA | .155 | - .321 |
| Chemistry 1(spring) | PCA | .647 | .933 |
| Chemistry 2 (spring) | PCA | .160 | -.419 |
| Chemistry 1(summer) | PCA | .538 | .796 |
| Chemistry 2 (summer) | PCA | .163 | -.230 |

collected in the summer (primary eigenvalue, 0.538) (table 4). Furthermore, these chemical gradients were strongly associated with urban intensity, but the samples collected in the spring had a stronger association than those collected in the summer. This result may indicate that measuring chemical runoff from urban landscapes during the spring provides a better integration of urban effects than does measuring chemistry during the summer when runoff is dependent upon periodic storms that are not easily sampled.

The gradient described by the habitat data was relatively weak (primary eigenvalue, 0.208), which may have resulted from the criteria used to select sites and sampling reaches. Sampling reaches were selected to be physically consistent among sites so that differences in the biological communities could be attributed to urban intensities in drainage basins; therefore differences in habitat features among the sampling reaches were minimal and resulted in relatively weak habitat gradients among sites. Although the data structure indicated relatively low ecological gradients in the habitat, algae-DTH, and algae-QMH data (as represented by the ordination eigenvalues), the urban index correlated reasonably well with their primary axis site scores. Urban intensity, therefore, was a likely factor affecting the variance in these data, but the relation was not as strong as was seen in the invertebrate, fish, and chemical data.

## Relations between Biological, Physical, and Chemical Ordinations

Site scores derived from the biological-community data were correlated with site scores derived from chemical and habitat data to determine how well the physical and chemical gradients corresponded to the biological gradients (table 5). In all comparisons, the highest correlations were with the primary axes site scores (|rho|, 0.574–0.868) rather than with the secondary axes or with combinations of primary and secondary axes site scores (|rho|, 0.002–0.431). This analysis indicated that the primary gradients derived from BPC data were all closely associated with a common ecological gradient, such as the one represented by the urban gradient and quantified by the urban index. The lack of correspondence among the secondary axes indicates that the secondary gradients in the BPC data sets were not associated with a common gradient.

The rho values derived from individual correlations of the biological-community data with the urban index (table 4) were compared to the rho values derived from individual correlations of the biological-community data with the habitat and the water- chemistry data (table 5). These comparisons were made to determine if any of the biological communities were more strongly correlated with either the habitat or the water chemistry than with the urban index. Invertebrate and fish communities were more highly correlated with the urban index than with the habitat and water-chemistry data. In contrast, depending on the sample type, the algae communities were more strongly correlated with the habitat and chemical data than with the urban index. The QMH-algae data correlated more strongly with habitat data, and the RTH- and DTH-algae data correlated more strongly with habitat and with summer water-chemistry data. These correlations indicate that algae communities may be more sensitive to the micro- or meso-scale conditions represented by the habitat and water-chemistry data than are the longer-lived invertebrates and fish, which tend to correspond more closely to the basin-wide characteristics represented by the urban index.

The finding that the RTH- and DTH-algae data correlated somewhat better with the summer water-chemistry data (representing water-quality conditions at the time of sampling) than with the spring water-chemistry data suggests that these communities were responding to water chemistry conditions in the short term. The strong relation between the QMH-algae data and the habitat data is probably related to differences in microhabitats at the different sampling sites. Sites with greater diversity in microhabitat types would be expected to support a greater variety of algae, which would be detected in the QMH-algae sample, but not in the RTH- or DTH-algae

**Table 5.** Rho values from Spearman correlations of the biological communities with the chemistry and habitat data by use of the site scores from the primary and secondary ordination axes.

[Bold indicates an absolute value of rho greater than 0.7. DTH, depositional targeted habitat; QMH, qualitative multihabitat; RTH, richest targeted habitat]

| Biological community sample | Ordination axis | | | | | |
|---|---|---|---|---|---|---|
| | Habitat | | Chemistry (spring) | | Chemistry (summer) | |
| | Primary | Secondary | Primary | Secondary | Primary | Secondary |
| Invertebrates RTH 1 | **0.783** | 0.371 | **-0.868** | 0.371 | **-0.818** | -0.002 |
| Invertebrates RTH 2 | .100 | .246 | -.305 | .075 | -.269 | .168 |
| Invertebrates QMH 1 | **.814** | .329 | **-.859** | .431 | **-.785** | .057 |
| Invertebrates QMH 2 | -.106 | .176 | -.085 | .165 | -.164 | .117 |
| Fish 1 | **.766** | .425 | **-.761** | .388 | **-.719** | .065 |
| Fish 2 | -.002 | .138 | -.084 | -.126 | -.116 | -.044 |
| Algae RTH 1 | **.721** | .310 | **-.708** | .427 | **-.721** | .200 |
| Algae RTH 2 | .147 | .016 | -.155 | -.289 | -.234 | -.176 |
| Algae DTH 1 | .640 | .259 | **-.709** | .275 | **-.727** | .330 |
| Algae DTH 2 | .050 | .111 | .015 | -.084 | -.062 | -.199 |
| Algae QMH 1 | **.762** | .314 | -.624 | .411 | -.574 | .l59 |
| Algae QMH 2 | -.031 | -.110 | -.022 | .387 | .089 | .177 |

samples. Although the sampling reaches were chosen to meet specific criteria for the study, there were differences in the quality and scale of habitat features among the sites. For example, for the habitat features of reach gradient, channel sinuosity, percentage of pools, and hydrologic variability, the CV among sites for each of these was greater than 0.5, even though the variability in these features among sites was not strongly correlated with the urban index. Therefore, micro- or meso-scale habitat features of a sampling location can more strongly affect certain biological assemblages than the basin features can collectively.

## Form of Ordination Responses and Thresholds

The Lowess smoother applied to scatterplots was an effective tool for identifying the form of responses, especially when the relation was not linear. If a threshold was apparent, it was generally between urban index values of about 35–40 (fig. 3). On the basis of these observations, the five data values within this range (urban index, 35.9–38.1) were used as a starting point for determining if a response threshold was statistically significant. This process was done for each variable by taking the data points of the variable that corresponded to these five urban-index values and using them simultaneously as the upper-limit endpoints of a first regression line and as the lower-limit endpoints of a second regression line. The premise of using this two-step regression is that if a threshold existed, the change in slope of the response of the variable would be most pronounced around this range of urban-index values. Once the two separate regressions were calculated, their slopes were compared for a statistical difference ($\alpha$, 0.05) with the general linear model (GLM). When slopes were statistically different, the interpretation was that a threshold existed in the response of the variable to the urban index, meaning that the rate of the response changed at a specific urban-index value (table 6). Consequently, variables that showed a threshold response had two separate responses associated with them: a first response below the threshold value, and a second response above the threshold value. The urban-index value that corresponded to the intersection of the two response lines was then calculated as the urban-index threshold. The slope of the second response line (above the threshold) was tested to determine if it was significantly different from zero ($p > 0.05$). If it was not, then the slope was deemed indicative that urban intensity at values greater than the threshold value did not cause a response to the variable.

Regression statistics for the variable responses shown in figures 4–9 are described in table 6. For variables that had a threshold response, the urban-index threshold is given along with the linear coefficients of determination ($r^2$) for the responses below and above the threshold (first and second responses, respectively). The slopes of the two response lines are also given, which are standardized so that they may be compared among variables. If a slope was not significantly different from 0 ($p > 0.05$), it is shown as 0. A value of 0 was observed with the second-response slopes of most variables with a threshold, but not with any of the first-response slopes (table 6). Additionally, to express the overall responses of the variables that had a threshold, a non-linear coefficient of determination (nonlinear $r^2$) was determined by use of the second-order equation and by including all data values along the urban index. The regression statistics given for the variables without a threshold include the single linear coefficient of determination ($r^2$) and the standardized slope of the response line.

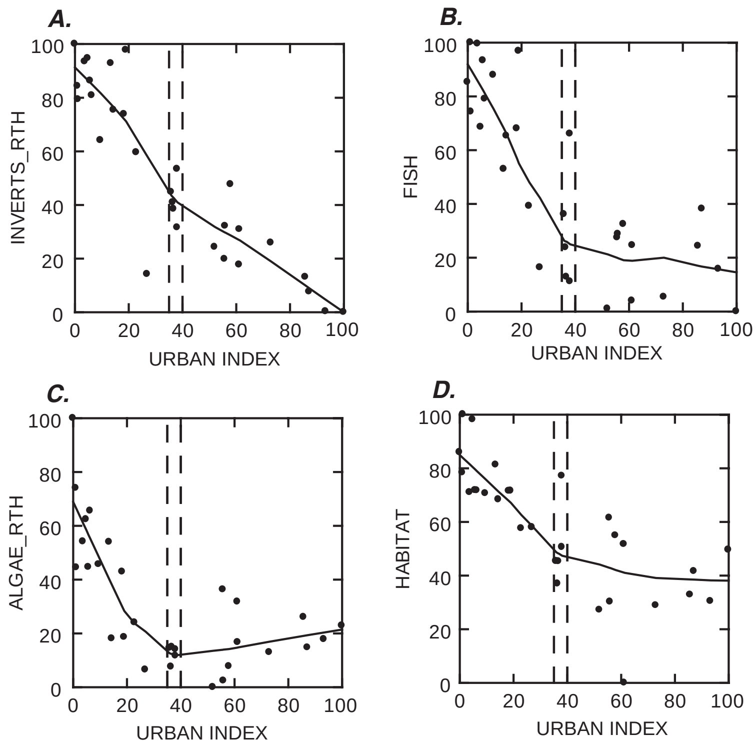



**Figure 3.** The response of biological communities and habitat ordinations in relation to the urban index, with the LOWESS (SPSS, 1998) regression smoother used to indicate trends. The vertical dashed lines represent urban index values between 35 and 40, which was the most consistent region where a threshold was seen.

Scatterplots of the BPC-ordination site scores (primary axis) with the urban index (fig. 4) showed that statistically significant threshold responses were evident in six of the nine components (table 6). These plots were scaled and standardized to represent either an increasing (positive) or decreasing (negative) relation between site scores and the urban index to show a trend that was ecologically relevant. Although the position of sites along the ordination axis (distance between sites) is important, the signs associated with site scores do not imply a relation with the urban index. Consequently, the signs of the site scores were standardized so that the direction of the derived gradient was consistent with improving or declining community conditions as indicated by community metrics.

The RTH- and QMH-invertebrate site scores had a strong negative response to the urban index (figs. 4*A* and 4*B*). The RTH data had a moderate threshold at an urban index value of 34.4, and the QMH data had no threshold (table 6). The slopes of the RTH response were significant on both sides of the threshold, indicating that impairment of the community continued at urban intensities greater than 34.4, but the rate of impairment was only about half of what it was at lower levels of urban intensity. The rate of impairment seen in the QMH-invertebrate data was uniform across the urban gradient and approximately equal to the slower rate exhibited for RTH data. The general forms of these responses indicate that the invertebrate communities changed predictably with urban intensity, and that there was no level of urban intensity (either increasing or decreasing) beyond which the community structure ceased to change. Site 15, which had an urban index value of 26.9 (table 2), had RTH- and QMH-invertebrate site scores that were indicative of sites that were somewhat more urbanized. This result was most pronounced in the ordination of RTH-invertebrates, indicating that taxa abundances were affected more than taxa richness. Site 15 was immediately below a wetland area and had a dissolved oxygen (DO) concentration at the time of sampling of only 66-percent saturation. This level was more characteristic of the DO saturation observed at high-intensity urban sites, and was second only to site 25 (urban index, 93.3), which had the lowest DO saturation. The water above the sampling reach of site 15 also had a sulfate concentration about 50 times greater than any other site, so the low dissolved oxygen saturation probably resulted from the reaction of dissolved oxygen with reduced sulphur compounds rather than from urban factors.

**Table 6.** Statistics characterizing the response of variables to the urban index.

[To determine which type of response each variable showed (either with or without threshold), response-variable data were grouped into two categories. The first category included data values where the urban index was less than 38. The second category included values where the urban index was greater than 35. This resulted in five data values that were included in both categories, and was based on the Lowess responses seen in figure 3. An interaction term of [Urban index*Category] was included in the general linear model (GLM) to test for homogeneity of slopes. This test was used to determine if the regression lines for the first and second responses were significantly different (P less than 0.05), thereby suggesting a threshold. For variables determined not to have a threshold, the response was characterized by a single linear regression. For variables determined to have a statistically significant threshold, the urban index value at the intersection of the first and second response lines was identified as the threshold value for the variable. Furthermore, a test was done to determine if the slope was significantly different from zero (slope = 0, where P was greater than 0.05). Slopes that were statistically 0 occurred only with a second response; the interpretation was that the variable does not respond predictably above the threshold value to higher urban intensity. To characterize the overall response strength of variables with a threshold, a non-linear regression was calculated by including all the data points of the variable so that the overall response could be compared with the response of variables without a threshold.]

| Variables with a threshold | | | | | | | Variables without a threshold | | |
|---|---|---|---|---|---|---|---|---|---|
| Variables | First response $r^2$ | Second response $r^2$ | Urban-index threshold | First response slope | Second response slope | Overall $r^2$ | Variables | Overall $r^2$ | Slope |
| Ordinations | | | | | | | | | |
| INVERTS_RTH | 0.659 | 0.752 | 34.4 | -1.5 | -0.7 | 0.835 | INVERTS_QMH | 0.856 | -0.8 |
| FISH | .641 | .086 | 37.4 | -1.7 | 0 | .737 | CHEM_SPRING | .791 | .6 |
| ALGAE_RTH | .718 | .110 | 33.9 | -1.6 | 0 | .686 | CHEM_SUMMER | .599 | .6 |
| ALGAE_DTH | .667 | .003 | 34.5 | -1.3 | 0 | .500 | | | |
| ALGAE_QMH | .511 | .010 | 33.0 | -1.3 | 0 | .478 | | | |
| HABITAT | .604 | .068 | 43.3 | -.9 | 0 | .677 | | | |
| Metrics | | | | | | | | | |
| CG_RICH_RTH | 0.641 | 0.008 | 37.8 | -1.7 | 0 | 0.689 | EPT_RICH_RTH | 0.777 | -0.8 |
| pCG_ABND_RTH | .603 | .003 | 35.2 | -1.5 | 0 | .650 | EPT_RICH_QMH | .767 | -.9 |
| DOM5_RTH | .681 | .293 | 38.1 | 1.5 | .4 | .776 | pNONINSR_QMH | .869 | 1.0 |
| MARGALEF | .574 | .338 | 43.3 | -1.2 | -.4 | .771 | pMOLCRUR_QMH | .824 | .9 |
| TOT_ABUND | .405 | .043 | 31.3 | -1.4 | 0 | .447 | RICHTOL_QMH | .856 | .8 |
| CYP_ABUND | .399 | .126 | 32.7 | -1.4 | 0 | .503 | pSILTMOTL_RTH | .613 | .7 |
| CYP_RICH | .648 | .101 | 35.0 | -1.8 | 0 | .688 | pEUTRO_QMH | .482 | .5 |
| LOTIC_ABUND_NT | .423 | .137 | 33.2 | -1.5 | 0 | .537 | BAHLS_RTH | .515 | -.7 |
| MOD_ABUND_AL | .399 | .139 | 31.5 | -1.2 | 0 | .500 | DEPTHm | .543 | .6 |
| BI_ABUND | .390 | .022 | 41.2 | -1.3 | 0 | .521 | BICARBONATE | .816 | .8 |
| TAXARICH_RTH | .535 | .188 | 37.1 | 1.2 | 0 | .651 | LOG_TKN | .633 | .6 |
| DIATRICH_RTH | .652 | .154 | 35.5 | 1.5 | 0 | .694 | | | |
| pACHNMN_RTH | .662 | .017 | 38.5 | -1.8 | 0 | .715 | | | |
| SW_DIATOM | .763 | .021 | 36.5 | 1.8 | 0 | .738 | | | |
| pFAC_HET_QMH | .459 | .032 | 41.7 | 1.0 | 0 | .552 | | | |
| BFWID_DEPm | .420 | .295 | 33.6 | -1.1 | -.3 | .587 | | | |
| DEG_DAYm | .735 | .198 | 4.0 | 1.0 | 0 | .735 | | | |
| SPECIF_COND | .758 | .319 | 36.8 | 1.4 | .6 | .768 | | | |
| CHEM_WQ_NDX | .811 | .529 | 36.2 | 1.5 | .6 | .868 | | | |

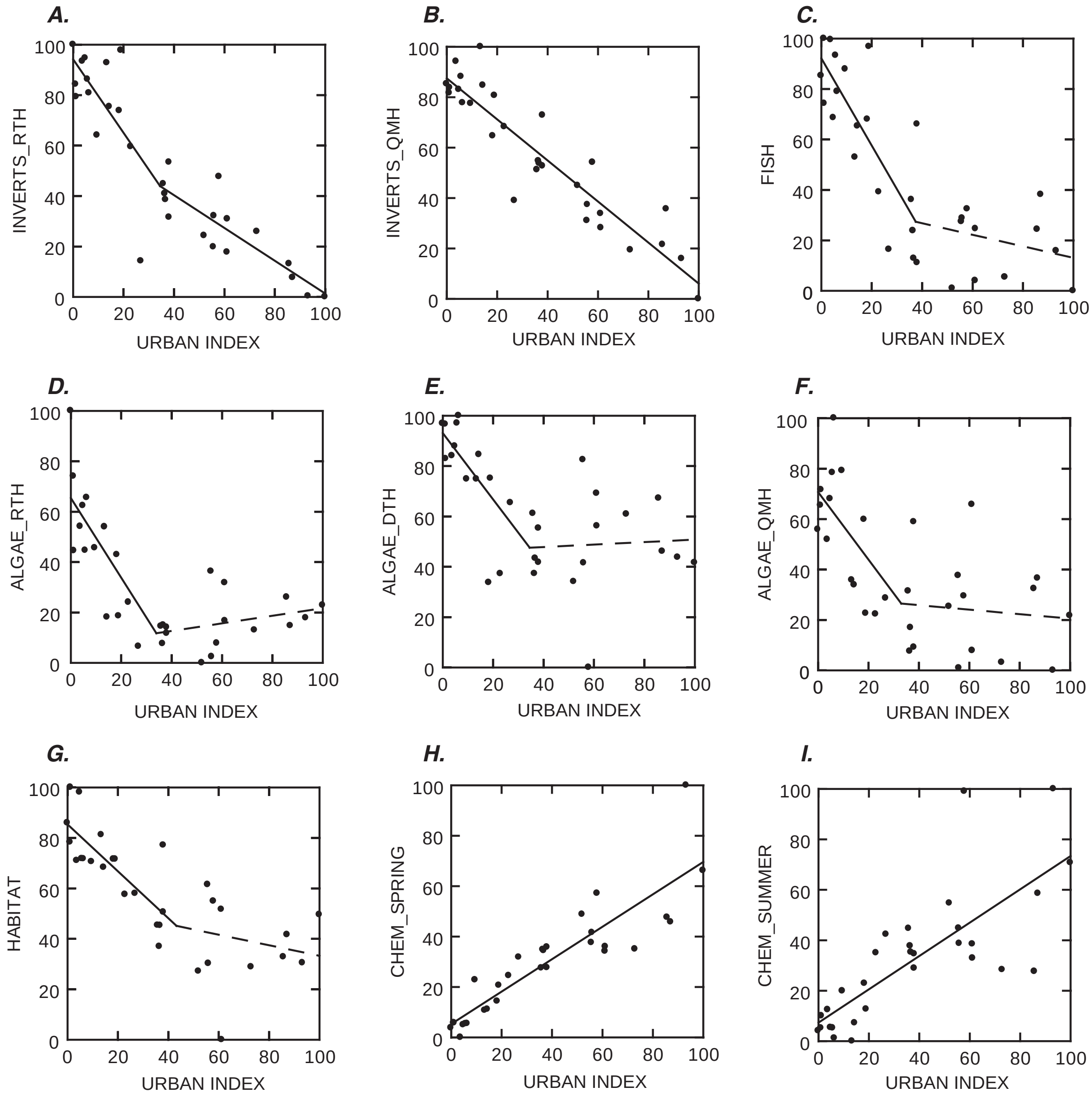



**Figure 4.** The responses of the primary axes site scores from the biological, physical, and chemical ordinations to the urban index. A two-slope regression line is shown in the plots if a threshold value of urbanization was significant ($p < 0.05$). If a regression-line slope is not significantly different from zero, the line is dashed. Refer to table 6 for regression statistics.

To determine if the RTH-response threshold was caused by leverage imposed by site 15, the data set was reanalyzed without data from site 15, but the resulting threshold was still statistically significant. Furthermore, the level of urban intensity associated with this threshold was similar to the threshold for other BPC data sets, so this response is probably valid rather than an artifact of the data. More importantly, this situation exemplifies how a natural environmental factor can strongly affect a biological community, and consequently complicate the interpretation of the response of the community to urban intensity.

Fish-site scores had a clearly negative response to the urban index (fig. 4*C*) and a threshold at the urban index value of 37.4 (table 6). At urban intensities below the threshold value, the slope of the response (-1.7) was similar to what was observed for RTH-invertebrates (-1.5). Unlike the invertebrate data, the fish-community data essentially ceased to show significant changes at urban intensities above the threshold (the second-response slope was not significantly different from 0). Furthermore, at levels of urban intensity above the threshold value, the fish-community data exhibited a high degree of variability at higher levels of urban intensity.

This heteroscedasticity (deviation in variance along the response line) was evident in a much lower coefficient of determination for the response above the threshold for the fish data than for the RTH-invertebrate data (second response $r^2$, 0.086 and 0.752, respectively, table 6).

Algae-site scores for RTH, DTH, and QMH data had negative responses to urban intensity and with thresholds (figs. 4*D*, 4*E*, and 4*F*). Like the response seen with the fish-community data, the algae-community data ceased to change predictably at levels of urban intensity above the threshold value. All algae responses had similar thresholds (urban index, 33.0–34.5) and slopes (-1.3 to -1.6), which compares to the values observed in RTH-invertebrate and fish-community data (table 6). A high degree of variability in the DTH- and QMH-algae data was evident at high levels of urban intensity (figs. 4*E* and 4*F*), and was evident in low coefficients of determination for the responses above the threshold (second response $r^2$, 0.003 and 0.010, respectively, table 6). This increase in variance may indicate that other environmental factors (such as components of water chemistry) or deposition of nonresident algae (such as sestonic algae from upstream impoundment) may affect community structure in DTH and QMH samples.

The habitat-site scores had a negative response with the urban index (fig. 4*G*) and with a threshold (43.3) that was slightly higher than those for biological communities (33.0–37.4, table 6). The rate of response below the threshold was low (slope, -0.9) compared to biological responses. As with the fish and algae data, the response slope was 0 at levels of urban intensity above the threshold. Site 23 (table 2) was an outlier in the scatterplot (fig. 4*G*), with an urban index value of 61.3 and a habitat-site score of 0. The reason for this outlier became apparent when the habitat survey found that the sampling reach was channelized many years ago. Although the site met the selection criteria for choosing sampling sites, the homogeneity among several of the features measured at 11 transects within the reach was the highest of any sampling site. Consequently, the uniformity among certain transect features at this site accounted for its position as an outlier on the habitat scatterplot, but evidently this did not cause the site to be an outlier on the biological-community scatterplots.

The chemistry-site scores from the samples collected in the spring and the summer both had a positive relation with urban intensity and were without thresholds (figs. 4*H* and 4*I*). The chemistry data collected in the spring were more strongly related to the urban index than were chemistry data collected in the summer ($r^2$ = 0.791 and 0.599, respectively, table 5). Heteroscedasticity of sites from the response line increases at higher urban index values; this increase indicates that the response of water chemistry becomes less predictable at higher urban intensities. This pattern is more evident in the data collected in the summer than in the data collected in the spring. Water-chemistry data collected in the spring is probably a better representation of conditions in the drainage basin over time, because these samples were collected during the period of springtime flush. Chemicals that had accumulated in the basin over the winter would therefore be conveyed in the spring water-chemistry samples. Conversely, the water-chemistry samples collected in the summer could be strongly affected by more proximate or point sources, especially at low flows that are typical during summer. For example, sites 14 and 25, with urban index values of 58.0 and 93.3 respectively, (table 2), were high outliers on the summer scatterplot (fig. 4*I*). There were sewage-treatment plants upstream of the sampling reaches that presumably had strongly affected the water chemistry. For a biological community that is particularly sensitive to water chemistry or that has a short generation time (such as algae), the community structure could change in the short term by responding to a change in water-quality conditions. As previously discussed, for example, the strongest correlation seen in the DTH-algae site scores was with water-chemistry data from the samples collected in the summer (table 5).

## Multimetric Analyses

Of the 126 invertebrate-, 92 fish-, and 164 algae-community metrics examined, only about 20 percent (43 invertebrate, 11 fish, and 29 algae) showed potential as indicators of urban intensity on the basis of Spearman correlations with the urban index with |rho| greater than 0.7 (tables 7, 8, and 9). Richness, tolerance, dominance, and diversity metrics were the best indicators of urban intensity for invertebrates. Flow preference, abundance of moderately tolerant taxa, and richness and abundance of minnows were the best indicators for fish. Taxa richness, diversity, tolerance, a few autoecological indices, and relative abundance of some diatom taxa were the best indicators for algae. Of the 89 habitat metrics examined, only 5 were strongly associated with urban intensity and 4 of these were dependent on stream depth (table 10). Of the 38 water-chemistry metrics examined, 12 were strongly associated with urban intensity. Of these, the best indicators were the field-measured properties and constituents (table 11). Only metrics with |rho| greater than 0.7 or metrics pertinent to the discussion with |rho| below this criterion are included in tables 7–11.

**Table 7.** Rho values from Spearman correlations of invertebrate metrics with the urban index.

[Shown only are metrics having a correlation (absolute value of rho, |rho|) greater than 0.7 with either RTH or QMH data. The number following each metric category is the total number of metrics tested for the category. Bold indicates |rho| greater than 0.7. QMH, qualitative multihabitat; RTH, richest targeted habitat; USEPA, U.S. Environmental Protection Agency; NA, metric not applicable; --, no correlations met the acceptance criterion where |rho| was greater than 0.7]

| Invertebrate metrics | Description | Richest targeted habitat | Qualitative multihabitat |
|---|---|---|---|
| | Richness metrics (23) | | |
| RICH | Total richness (number of non-ambiguous taxa) | **-0.853** | **-0.862** |
| EPTR | Richness composed of EPT (mayflies, stoneflies, caddisflies) | **-.898** | **-.885** |
| EPEMR | Richness composed of mayflies | **-.850** | **-.826** |
| PLECOR | Richness composed of stoneflies | **-.809** | **-.815** |
| TRICHR | Richness composed of caddisflies | **-.839** | **-.860** |
| ODONOR | Richness composed of odonates | -.466 | **-.703** |
| DIPR | Richness composed of Diptera | **-.812** | **-.788** |
| CHR | Richness composed of Chironomidae midges | **-.808** | **-.740** |
| ORTHOR | Richness composed of Orthocladiinae midges | **-.789** | **-.738** |
| NONINSR | Richness composed of noninsects | .633 | **.773** |
| MOLCRUR | Richness composed of molluscs and crustaceans | **.765** | **.781** |
| GASTROR | Richness composed of Gastropoda | .517 | **.710** |
| | Percent richness metrics (22) | | |
| pEPTR | Percentage of total richness composed of EPT (mayflies, stoneflies, caddisflies) | -0.669 | **-0.837** |
| pEPEMR | Percentage of total richness composed of mayflies | -.690 | **-.702** |
| pPLECOR | Percentage of total richness composed of stoneflies | **-.723** | **-.782** |
| pTRICHR | Percentage of total richness composed of caddisflies | -.340 | **-.763** |
| pNONINSR | Percentage of total richness composed of noninsects | **.841** | **.910** |
| pODIPNIR | Percentage of total richness composed of nonmidge Diptera and noninsects | **.839** | **.853** |
| pMOLCRUR | Percentage of total richness composed of molluscs and crustaceans | **.869** | **.899** |
| pGASTROR | Percentage of total richness composed of Gastropoda | .625 | **.822** |
| pBIVALR | Percentage of total richness composed of Bivalvia | .643 | **.765** |
| | Abundance metrics (23) | | |
| EPEM | Abundance of mayflies | **-0.764** | NA |
| PLECO | Abundance of stoneflies | **-.833** | NA |
| | Percent abundance (composition) metrics (22) | | |
| pEPEM | Percentage of total abundance composed of mayflies | **-0.763** | NA |
| pPLECO | Percentage of total abundance composed of stoneflies | **-.850** | NA |
| pCOLEOP | Percentage of total abundance composed of Coleoptera | **-.773** | NA |
| pORTHO | Percentage of total abundance composed of Orthocladiinae midges | **-.744** | NA |
| | Functional group richness metrics (6) | | |
| PR Rich | Richness composed of predators | **-0.760** | -0.645 |
| CG Rich | Richness composed of collector-gatherers | **-.743** | **-.761** |
| | Percent functional group richness metrics (6) | | |
| -- | -- | -- | -- |
| | Functional group abundance metrics (6) | | |
| -- | -- | -- | -- |
| | Percent functional group abundance metrics (6) | | |
| pCG Abund | Percentage of total abundance composed of collector-gatherers | **-0.754** | NA |
| pSC Abund | Percentage of total abundance composed of scrapers | **-.713** | NA |

**Table 7.** Rho values from Spearman correlations of invertebrate metrics with the urban index.—Continued

[Shown only are metrics having a correlation (absolute value of rho, |rho|) greater than 0.7 with either RTH or QMH data. The number following each metric category is the total number of metrics tested for the category. Bold indicates |rho| greater than 0.7. QMH, qualitative multihabitat; RTH, richest targeted habitat; USEPA, U.S. Environmental Protection Agency; NA, metric not applicable; --, no correlations met the acceptance criterion where |rho| was greater than 0.7]

| Invertebrate metrics | Description | Richest targeted habitat | Qualitative multihabitat |
|---|---|---|---|
| | Tolerance metrics (2) | | |
| RICHTOL | Average USEPA tolerance values for sample based on richness | **0.851** | **0.897** |
| ABUNDTOL | Abundance-weighted USEPA tolerance value for sample | **.816** | NA |
| | Percent abundance of dominant taxa (5) | | |
| DOM1 | Percentage of total abundance represented by the most abundant taxa | **0.731** | NA |
| DOM2 | Percentage of total abundance represented by the two most abundant taxa | **.798** | NA |
| DOM3 | Percentage of total abundance represented by the three most abundant taxa | **.827** | NA |
| DOM4 | Percentage of total abundance represented by the four most abundant taxa | **.844** | NA |
| DOM5 | Percentage of total abundance represented by the five most abundant taxa | **.861** | NA |
| | Diversity and evenness indices (5) | | |
| MARGALEF | Margalef's Diversity | **-0.856** | NA |
| MENHINIC | Menhinick's Diversity | **-.774** | NA |
| SIMPDOM | Simpson's Dominance | **.807** | NA |
| SHANDIV | Shannon's Diversity | **-.847** | NA |
| SHANEVEN | Shannon's Evenness | **-.806** | NA |

## Invertebrate Metrics

Invertebrate-richness metrics associated with Ephemeroptera, Plecoptera, Trichoptera (EPT), Diptera, midge, non-insects, mollusks and crustaceans generally responded strongly to urban intensity for the RTH and QMH data, but there were some differences when richness was expressed as number of taxa or percentage of total taxa richness (table 7, fig. 5). Richness metrics for total taxa, EPT, dipteran, and midge taxa decreased as urban intensity increased, whereas richness metrics for non-insect, mollusk + crustacean, and gastropod taxa increased with urban intensity. The EPT richness for the RTH and QMH data (figs. 5*A* and 5*B*) exhibited some of the strongest responses to urban intensity of any of the invertebrate metrics, steadily declining as urban intensity increased and did not have a threshold. Metrics based on EPT-taxa richness have become a standard in biomonitoring programs worldwide, and the results of this study confirm their utility as an indicator of disturbance. Despite their importance, EPT richness never exceeded 36 percent of total richness in any of the samples. The percentage of total taxa richness composed of non-insects and the mollusks + crustaceans also had a strong response to urban intensity, especially with the QMH data, but the trend increased with increases in urban intensity (figs. 5*C* and 5*D*). Together, mollusks and crustaceans accounted for about half of the richness of the non-insect invertebrates in the RTH and QMH data. The oligochaetes accounted for the majority of the remainder of the richness. The taxa in the mollusk + crustacean communities were primarily the gastropods, amphipods, and bivalves, with only a few decapods and isopods.

Invertebrate-abundance metrics, expressed as both density and percentage abundance, generally did not correspond as well to urban intensity as did the richness metrics. Only six abundance metrics (mayfly abundance and percentage abundance, stonefly abundance and percentage abundance, midge percentage abundance, and Coleoptera percentage abundance) responded strongly to urban intensity, but the strength of this response was typically less than for richness (table 7). These metrics all decreased as urban intensity increased. The EPT abundance did not relate strongly with urban intensity because mayfly and stonefly abundance decreased with increasing urban intensity while caddis-fly abundance increased. Stonefly abundance decreased primarily over the low to moderately low range of urban intensity (urban index less than 20). Stoneflies (a taxon intolerant of disturbances) were only a small fraction (< 1 percent) of the abundance found in RTH samples at sites with an urban index value above 20. The increase in caddisfly abundance with urban intensity was primarily caused by an increase in the abundance of a few tolerant hydropsychid taxa. The abundance of other caddisfly taxa decreased as urban intensity increased.

**Table 8.** Rho values from Spearman correlations of fish metrics with the urban index.

[Shown only are metrics with a correlation (absolute value of rho, |rho|) greater than 0.7, except in cases where metrics not meeting this criteria are discussed. The number following each metric category is the number of metrics tested for the category. Bold indicates |rho| greater than 0.7. --, no correlations met the acceptance criterion where |rho| was greater than 0.7]

| Fish metrics | Description | Rho value |
|---|---|---|
| | Abundance (22) | |
| TOT_ABUND | Abundance of all individuals | -0.638 |
| CYP_ABUND | Abundance of cyprinids (minnows) | **-.796** |
| pCYP_ABUND_AL | Percentage of total abundance composed of cyprinids among all individuals | **-.772** |
| pCYP_ABUND_NT | Percentage of total abundance composed of cyprinids among non-transient individuals | **-.773** |
| pBASS_ABUND_NT | Percentage of total abundance composed of centrarchids among non-transient individuals | .629 |
| | Richness (11) | |
| CYP_RICH | Number of cyprinid species | **-0.798** |
| pCYP_RICH _AL | Percentage of total richness composed of cyprinid species among all species | **-.765** |
| pCYP_RICH _NT | Percentage of total richness composed of cyprinid species among non-transient species | **-.785** |
| | Feeding abundance (8) | |
| BI_ABUND | Abundance of benthic insectivore individuals | **-0.729** |
| | Feeding richness (8) | |
| -- | -- | -- |
| | Temperature preference abundance (2) | |
| -- | -- | -- |
| | Flow preference abundance (2) | |
| LOTIC_ABUND_NT | Abundance of lotic individuals, among non-transient individuals | **-0.827** |
| pLOTIC_ABUND_NT | Percentage of total composed of lotic individuals, among non-transient individuals | **-.752** |
| | Tolerance abundance (12) | |
| MOD_ABUND_AL | Abundance of moderately tolerant individuals, among all individuals | **-0.707** |
| MOD_ABUND_NT | Abundance of moderately tolerant individuals, among non-transient individuals | **-.704** |
| | Tolerance richness (12) | |
| -- | -- | -- |
| | Tolerance index (2) | |
| -- | -- | -- |
| | Anomalies (8) | |
| -- | -- | -- |
| | Diversity and evenness indexes (5) | |
| -- | -- | -- |

**Table 9.** Rho values from Spearman correlations of algal metrics with the urban index.

[Shown only are metrics having a correlation (absolute value of rho, |rho|) greater than 0.7 with either RTH, DTH, or QMH data, except in cases where metrics not meeting this criteria are discussed. The number following each metric category is the total number of metrics tested for the category. Bold indicates |rho| greater than 0.7; DTH, depositional targeted habitat; QMH, qualitative multihabitat; RTH, richest targeted habitat; NA, metric not applicable]

| Algal metrics | Description | RTH | DTH | QMH |
|---|---|---|---|---|
| | Composition (abundance—RTH, DTH; richness—QMH) within taxa (109) | | | |
| TOTAXA | Total taxa abundance (RTH and DTH) and richness (QMH) | 0.621 | 0.304 | 0.578 |
| pNAVI001AF | Percentage of total composed of Naviculaceae | **.793** | .482 | .158 |
| pNITZ001AF | Percentage of total composed of Nitzschiaceae | **.772** | .455 | .249 |
| pDIAT026AG | Percentage of total composed of *Synedra* | **.707** | .239 | -.073 |
| pNAVI036AG | Percentage of total composed of *Gomphonema* | **.741** | -.101 | -.159 |
| pNAVI050AG | Percentage of total composed of *Navicula* | **.709** | .607 | .376 |
| pNITZ003AG | Percentage of total composed of *Nitzschia* | **.772** | .459 | .237 |
| | Richness (3) | | | |
| RICHF | Family richness | 0.551 | 0.149 | 0.141 |
| RICHG | Genera richness | **.817** | .198 | .451 |
| RICHS | Taxa richness | **.821** | .358 | .578 |
| | Biomass (13) | | | |
| BIOVOLTOT | Biovolume total | 0.547 | 0.236 | NA |
| BIOVOLDIAT | Biovolume total diatoms | .661 | .281 | NA |
| AFDM | Ash free dry mass | .340 | NA | NA |
| CHLA | Chlorophyll *a* concentration (Chl *a*) | .383 | NA | NA |
| AI | Autotrophic index ([AFDM] . [Chl $a$]$^{-1}$) | -.162 | NA | NA |
| | Diversity and eveness indices (5) | | | |
| MARGALEF | Margalef's Diversity | **0.799** | 0.262 | NA |
| SIMPSONDOM | Simpson's Dominance | **-.768** | -.345 | NA |
| SHANDIV | Shannon's Diversity | **.787** | .338 | NA |
| | Composition based on diatom autecology (13) | | | |
| pSILTMOTL | Percentage of diatoms composed of taxa motile in silt | **0.846** | 0.685 | 0.529 |
| pFAC_HET | Percentage of diatoms composed of facultative nitrogen heterotrophics | **.784** | .169 | **.701** |
| pAUTO | Percentage of diatoms composed of nitrogen autotrophs | **-.710** | -.339 | -.434 |
| pEUTRO | Percentage of diatoms composed of eutrophic taxa | .613 | .486 | **.751** |
| | Diatom tolerance indices (6) | | | |
| pLBVTOL | Percentage of diatoms composed of Lange-Bertalot Pollution Tolerant Class 1 | **0.776** | 0.349 | -0.311 |
| pLBTOL2A | Percentage of diatoms composed of Lange-Bertalot Pollution Tolerant Class 2a | **.741** | .476 | .562 |
| pLBTOL2B | Percentage of diatoms composed of Lange-Bertalot Pollution Tolerant Class 2b | **.715** | .626 | .253 |
| pLBTOL3A | Percentage of diatoms composed of Lange-Bertalot Pollution Tolerant Class 3a | **-.709** | -.506 | .198 |
| BAHLS | Bahl's Pollution Index | **-.705** | .023 | .027 |
| | Diatom community diversity (15) | | | |
| DIATRICH | Diatom taxa richness | **0.846** | 0.407 | 0.462 |
| pDOMDIAT | Percentage of diatom abundance composed of the most abundant taxa | **-.726** | -.449 | NA |
| pDOM | Percentage of diatom abundance composed of taxa that occurred in greater than 10 percent abundance | **-.771** | -.443 | NA |
| SWDIAT | Shannon-Wiener Diversity Index for diatoms | **.801** | .446 | NA |
| pACHNMN | Percentage of diatom abundance composed of *Achnanthidium minutissimum* | **-.713** | .511 | NA |
| pNAVICULA | Percentage of diatom abundance composed of *Navicula* spp. | **.715** | NA | NA |
| pFILAMENT | Percentage of diatom abundance composed of filamentous diatoms | .598 | NA | NA |
| pERECT | Percentage of diatom abundance composed of erect diatoms | **.751** | NA | NA |
| pBIRAPHID | Percentage of diatom abundance composed of biraphid, prostrate, and nonmotile diatoms | **.752** | NA | NA |
| pSTALKED | Percentage of diatom abundance composed of stalked diatoms | **.745** | NA | NA |

**Table 10.** Rho values from Spearman correlations of habitat metrics with the urban index.

[Shown only are metrics having a correlation (absolute value of rho, |rho|) greater than 0.7 or those used in the principal components analysis. Variables listed in bold were used in the principal components analysis. Bold indicates |rho| greater than 0.7; CV, coefficient of variation; DTH, depositional targeted habitat; RTH, richest targeted habitat]

| Variables | Definition | Rho value |
|---|---|---|
| | Measured variables (61) | |
| **SEGLENG** | Segment length | -0.544 |
| **SINUOS** | Segment sinuosity | -.405 |
| **SEGGRAD** | Segment gradient | -.234 |
| **ORDER** | Stream order | .374 |
| **LINK** | Stream link | .389 |
| **RCHGRAD** | Reach gradient | -.214 |
| **SINUOSITY** | Reach sinuosity | -.291 |
| **POOLp** | Percentage of sampling reach composed of pools | .043 |
| **RUNp** | Percentage sampling reach composed of run | -.067 |
| **CHWIDTHm** | Wet channel width (mean) | -.204 |
| **BFWIDTHm** | Bankfull channel width (mean) | -.312 |
| **LUUR** | Land-use percent urban residential | .493 |
| **LUSW** | Land-use percent shrubs and woods | -.407 |
| **LURR** | Land-use percent rural residential | -.532 |
| **LURW** | Land-use percent right-of-way | -.141 |
| **DEPTHm** | Wet stream depth (mean) | **.742** |
| **BFDEPm** | Bankfull depth (mean of transects) | .656 |
| **BNKANGm** | Bank angle (mean) | -.240 |
| **BNKSUBSTRm** | Bank substrate (mean) | .348 |
| **BNKVEGm** | Bank vegetation cover (mean) | -.468 |
| **BANKEROSp** | Percent bank erosion | .187 |
| **BARWD** | Total bar width | -.160 |
| | Measured variables (61)—Continued | |
| **SHELFWD** | Total shelf width | -0.535 |
| **FLOVELm** | Flow velocity (mean) | .433 |
| **SILTp** | Percent silt on substrate | -.182 |
| **EMBEDDEDp** | Percent embeddedness | .369 |
| **SUBSTRATEm** | Substrate size (mean) | -.156 |
| **SHADEm** | Riparian shading (mean) | .057 |
| **CANANGLEm** | Canopy closure angle (mean) | .105 |
| **HABCOVERp** | Percent habitat cover | .412 |
| **DEPmRTH** | Depth at RTH (mean) | .605 |
| **DEPmDTH** | Depth at DTH (mean) | .099 |
| **VELmRTH** | Flow velocity at RTH (mean) | .435 |
| **DOMSUBmRTH** | Dominate substrate at RTH (mean) | -.134 |
| **SUBSUBmRTH** | Subdominant substrate at RTH (mean) | -.310 |
| **EMBEDpRTH** | Percent embeddedness at RTH | .409 |
| | Derived metrics (28) | |
| **DEG_DAYm** | Stream temperature—degree-days (mean) | **.869** |
| **HYDRVAR** | Hydrologic variability—skew of stage | .588 |
| CHWID_DEPm | Wet channel width to wet depth ratio (mean) | **-.735** |
| BFWID_DEPm | Bankfull channel width to wet depth ratio (mean) | **-.810** |
| HYDRADIUS | Hydraulic radius | **.733** |

Invertebrate functional-group metrics were generally not good indicators of urban intensity, regardless of whether the metrics were expressed as richness, percentage richness, abundance, or percentage abundance. Only 4 (predator richness, scraper percentage abundance, collector-gatherer richness, and collector-gatherer percentage abundance) of 24 metrics had a strong relation with urban intensity (table 7). As urban intensity increased, collector-gatherer richness (fig. 5*E* for RTH, but also for QMH) and percentage abundance (fig. 5*F*) decreased, with thresholds at urban index values of 37.8 and 35.2, respectively (table 5). There was no response to levels of urban intensity above these thresholds. Predator richness and scraper percentage abundance also decreased as urban intensity increased, but this response was seen only with the RTH data where |rho| was greater than 0.7. Through an automated process within the IDAS program that uses taxa-attribute data from USEPA (Barbour and others, 1999), functional groups were assigned to the taxa that represented 91–100 percent of taxa richness and 92–100 percent of abundance. When a taxon did not have a functional group identified in the USEPA data, then IDAS would make a functional group assignment to the taxon from a higher taxonomic level, if one existed in the USEPA data.

Invertebrate pollution-tolerance metrics that were based on an average of the USEPA tolerance values (Barbour and others, 1999) had strong positive responses to urban intensity, especially when the metrics were based on taxa richness of the QMH data (fig. 5*G*). This response was linear and without a response threshold. The pollution-tolerance metrics responded positively to disturbance because the USEPA scores pollution tolerances from 0 for the most pollution-sensitive taxa to 10 for the most pollution-tolerant taxa. Consequently, the metric value increases as the number and abundance of pollution-sensitive taxa decline and more tolerant taxa increase. With the IDAS program, USEPA data were used to assign tolerance values to taxa that represented 95–100 percent of taxa richness and 94–100 percent of abundance, which is the same approach that was used to assign functional groups to taxa.

**Table 11.** Rho values from Spearman correlations of chemical variables with the urban index.

[Variables in bold were used in the principal components analysis. Rho values in bold are greater than an absolute value of 0.7; ND, not detected]

| Chemical constituents | Spring | Summer |
|---|---|---|
| Field-measured parameters and constituents | | |
| **Specific conductance** | **0.860** | **0.852** |
| **pH** | **.796** | .546 |
| Air temperature | .108 | .191 |
| Water temperature | .637 | .439 |
| Dissolved oxygen | **-.745** | -.414 |
| **Dissolved oxygen saturation** | -.415 | -.346 |
| Alkalinity, dissolved | **.903** | **.868** |
| **Bicarbonate, dissolved** | **.904** | **.863** |
| Nutrients | | |
| Inorganic nitrogen ammonia, dissolved | 0.671 | 0.600 |
| Nitrogen ammonia and organic, dissolved | **.772** | .592 |
| **Total organic nitrogen + ammonia (TKN)** | **.837** | .583 |
| **Nitrite+nitrate, dissolved** | **.779** | .683 |
| Nitrogen, nitrite | .460 | .589 |
| Dissolved inorganic nitrogen | **.781** | **.703** |
| Phosphorus, dissolved | .581 | .321 |
| Phosphorus, ortho | .327 | .444 |
| **Total phosphorus** | .639 | .465 |
| Herbicides | | |
| Atrazine | 0.685 | 0.237 |
| Benfluralin | .510 | ND |
| Cyanazine | ND | .097 |
| Deethyl atrazine | .147 | .028 |
| Metolachlor | .205 | -.048 |
| Pendimethalin | .432 | ND |
| Prometon | .692 | .243 |
| Pronamide | ND | .075 |
| Simazine | .318 | -.032 |
| Tebuthiuron | .247 | ND |
| Terbuthylazine | ND | .161 |
| Trifluraline | .519 | -.031 |
| **Total herbicides** | **.823** | .349 |
| Insecticides | | |
| Carbaryl | 0.263 | 0.365 |
| Chlorpyrisfos | .309 | .290 |
| Diazinon | **.838** | .694 |
| Lindane | ND | .290 |
| Malathion | ND | .290 |
| Azinphos-methyl | ND | -.032 |
| P, P’, DDE | .277 | .032 |
| **Total insecticides** | **.841** | .639 |

Metrics that measured the percentage abundance of dominant invertebrate taxa (single most dominant through five most dominant) showed that taxa dominance increased with increasing urban intensity for all five metrics (table 6). Furthermore, correlations with urban intensity became progressively stronger as the dominance metric included more taxa. The highest correlation was with the metric represented by the five most abundant taxa (fig. 5*H*). This metric (DOM5_RTH) had a response threshold at an urban index value of 38.1(table 6), and the rate of response was almost four times higher at urban intensities below this threshold (first slope, 1.5) than above (second slope, 0.4).

All five of the invertebrate-community diversity and evenness indices responded strongly to urban intensity. Diversity and evenness decreased and dominance increased (table 7). Of these metrics, Margalef’s diversity had the strongest response to urban intensity and a threshold at 43.3 (fig. 5*I*, table 6). The rate of change below the threshold was three times greater (first slope, -1.2) than above the threshold (second slope, -0.4). Regarded in total, the responses of these metrics were a strong indication that as urban intensity increased, invertebrate diversity decreased and the abundance of certain tolerant taxa increased.

## Fish Metrics

Compared to the invertebrate metrics, few fish metrics (11 of 92) had a response (table 8) with a |rho| value greater than 0.7 with the urban index. Furthermore, when a metric was biased to certain taxa, the responses to urban intensity improved marginally at best. A biased metric was used when a specific characteristic of the fish community (such as abundance or richness) was measured relative to introduced taxa, salmonid taxa, or nontransient taxa. For example, the metric pCYP_ABUND_AL measured the percent abundance of minnows relative to all taxa; pCYP_ABUND_NT was measured relative only to the nontransient taxa (eels, sea lampreys, and stocked salmonids were excluded). The relation between the urban index and metrics that excluded these groups did not differ strongly from the relation observed when all taxa were used, probably because these groups accounted for relatively small percentages of the total richness and abundance at most sites.

Most of the fish metrics that responded strongly to the urban index were based on or primarily affected by one or more of the cyprinid (minnow) taxa. Cyprinids as a group were the most dominant taxon overall, and they best represented changes in the fish communities among the sites. This result can be seen when the responses are compared between total and cyprinid abundances. Although it was not strongly correlated with urban intensity (|rho|, 0.638), total abundance did show a tendency to decrease as urban intensity increased, with a sharp shift in the rate of change at a threshold of 31.3 (fig. 6*A*, table 6).

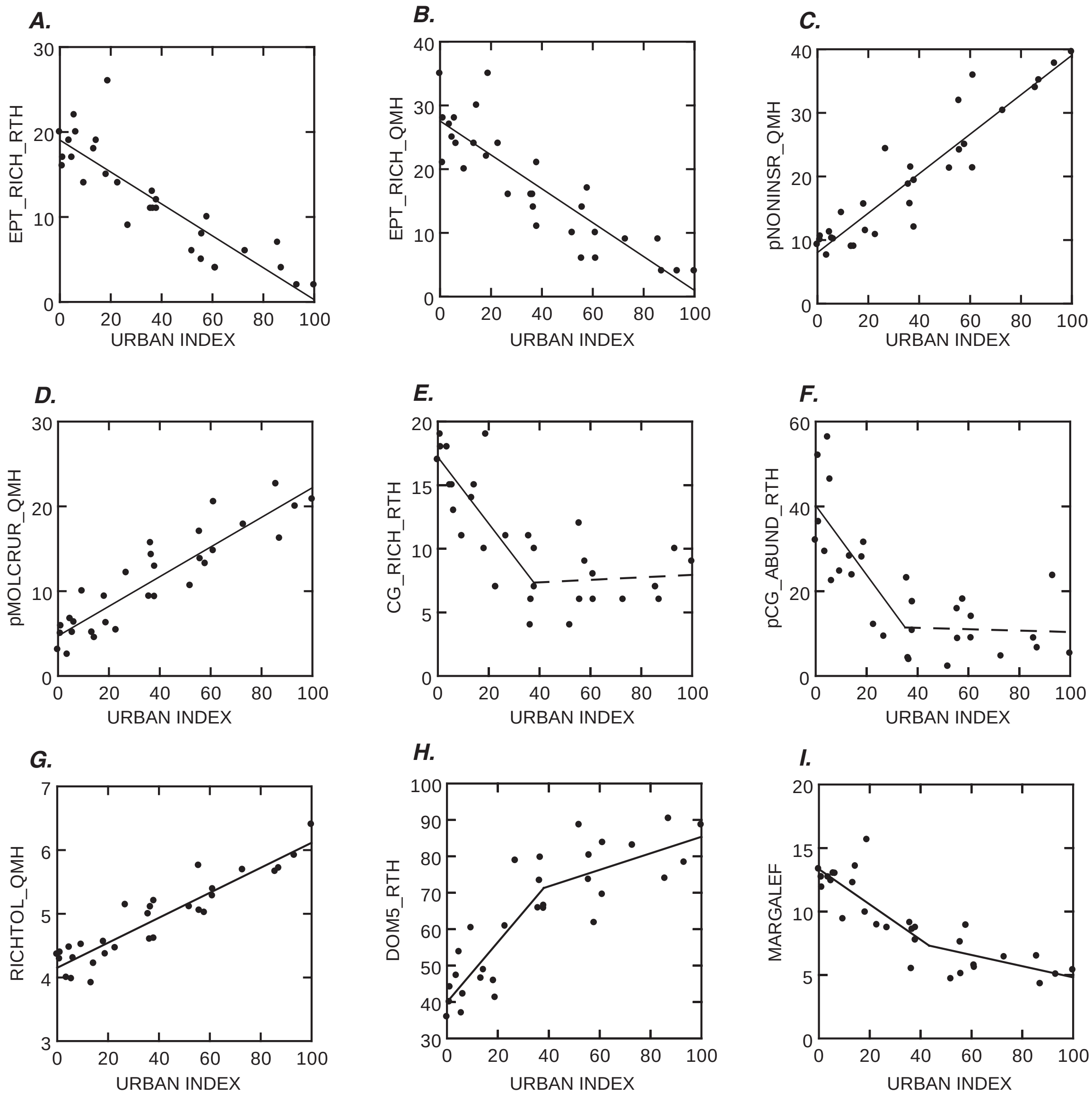



**Figure 5.** The responses of selected invertebrate metrics to the urban index. A two-slope regression line is shown in the plots with a significant ($p < 0.05$) threshold value of urbanization. If a regression-line slope is not significantly different from zero, the line is dashed. Refer to table 6 for regression statistics. Refer to table 7 for variable definitions.

The response of cyprinid abundance (fig. 6*B*) was virtually the same as the response of total taxa, with similar abundance values (Y-axis scale), a threshold only slightly higher (32.7), and the same response rate below the threshold (-1.4). Cyprinid richness also decreased with increasing urban intensity and had a threshold at 35.0 (fig. 6*C*, table 6). The centrarchids (sunfish) were the second-most dominant taxon, but metrics associated with them were not strongly correlated with the urban index (table 8). For example, the relative abundance of centrarchids (*Lepomis*, in particular) was highest at sites near the mid-range of urban index values.

The abundance of fluvial-dependent fish (expressed as lotic) decreased as urban intensity increased, and the response had a threshold at 33.2 (fig. 6*D*, table 6). This response indicates that the flow regime of the streams change from more lotic to more lentic with increases in urban intensity. Taxa that affected this metric were primarily minnow species, and secondarily, margined madtoms and tessellated darters. Salmonid abundance was not used in calculating this metric because trout and salmon have been stocked in all streams where they were collected, but it is possible that at least some of the individuals came from self-reproducing populations.

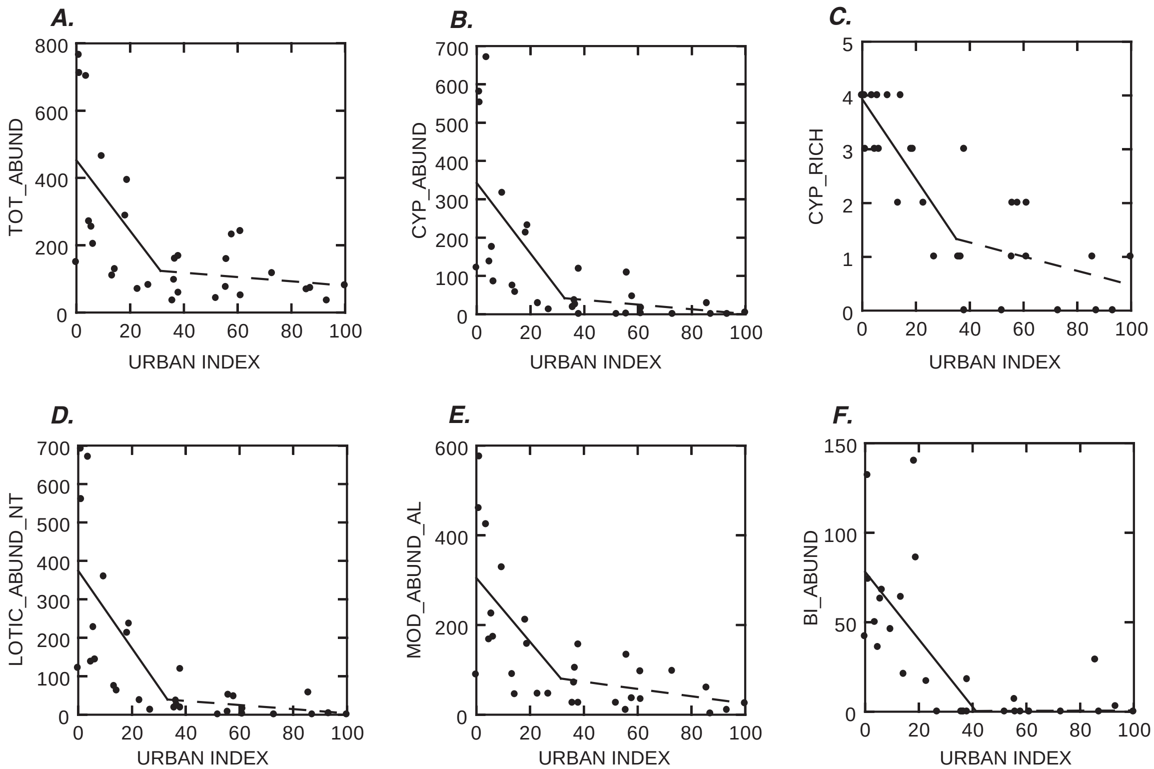



**Figure 6.** The responses of selected fish metrics to the urban index. A two-slope regression line is shown in the plots with a significant ($p < 0.05$) threshold value of urbanization. If a regression-line slope is not significantly different from zero, the line is dashed. Refer to table 6 for regression statistics. Refer to table 8 for variable definitions.

The abundance of moderately tolerant fish (whether based on all or nontransient taxa) were the only tolerance metrics that responded strongly to urban intensity. The response is represented by the abundance of moderately tolerant fish that include all taxa; the abundance decreased with increasing urban intensity to a threshold at 31.5, above which there was no significant change (fig. 6*E*, table 6). The taxa that affected this metric were primarily in the cyprinid and the centrarchid families, and secondarily, margined madtoms, yellow perch, and tessellated darters. Abundance and richness of pollution-intolerant fish, mainly limited to the salmonids, were relatively low among all sites. Salmonids were collected at several sites that varied in urban intensity. These fish, however, are stocked for recreational fishing and were collected in some of the more urbanized streams where it is unlikely that they would reproduce. Consequently, the correlation between intolerant taxa and the urban index was low. Fish that were pollution-tolerant (primarily white sucker, blacknose dace, and bluegill) were fairly widespread throughout the sampling network, and their cosmopolitan nature was apparent in the weak correlations between the metrics for pollution-tolerant taxa and the urban index.

The abundance of benthic insectivores was the only trophic (feeding) guild metric that had a strong relation with the urban index. Abundance of benthic insectivores decreased with increasing urban intensity and had a threshold at 41.2, above which there was no significant change (fig. 6*F*, table 6). The taxa characterized by this metric were longnose dace (60 percent), margined madtom (30 percent), and tessellated darter (10 percent). Furthermore, the combined abundance of these taxa relative to total abundance was low (less than 13 percent) when compared to the abundances of taxa that affected other fish metrics that also had a strong correlation with the urban index. The response of this metric, however, was not as dependent on the more dominant taxa that strongly affected other metrics.

Metrics summarizing temperature preferences, tolerance indices, anomalies, community diversity, evenness, and dominance were not strongly correlated with urban intensity; this finding indicates that the responses based on these metrics were, as a whole, dependent on other environmental factors. Furthermore, the lack of strong correlations between diversity and evenness indices and the urban index may be indicative of segregation among fish communities within the site network.

Unlike the motile adult stage of aquatic insects, fish populations are likely restricted in their movements by the numerous dams within the basins of the sampling network. Conversely, impoundments can serve as refuges; for example, if an impoundment is in close proximity to a sampling site, fish species (such as yellow perch) that tolerate a more lentic environment are usually in greater abundances at that sampling site.

## Algae Metrics

Only 29 of 164 algae metrics had a strong correlation (|rho| > 0.7) with the urban index. Of these metrics, 28 were associated with RTH data, 2 with QMH data, and 0 with DTH data (table 9). All were strongly affected by diatoms, which were the predominant group in all samples. This predominance is represented in the RTH data, where an average of 87.9 percent of taxa richness (CV, 10.8 percent) consisted of diatoms, and there was an almost 1:1 correspondence between total and diatom taxa richness across the urban gradient. Furthermore, total taxa richness and diatom richness related strongly to urban intensity (figs. 7*A* and 7*B*). Both measures of taxa richness increased with urban intensity, and the responses had a threshold at 37.1 for total richness and 35.5 for diatom richness; above these thresholds there was no significant response (table 6).

Taxa richness of genera also had a strong correlation with the urban index, but the response was slightly less than that of total taxa richness, which was based on the lowest taxonomic level (primarily species). At the family level, however, there was a much weaker relation of taxa richness with urban intensity. Algae richness, therefore, appears more responsive (at least to urban intensity) when based on the lower taxonomic levels, and consequently underscores the importance of a high degree of taxonomic resolution when algae are used for biological assessments.

The majority of algae metrics (109 of 164) was based on composition within various taxa. In addition to total taxa abundance (RTH, DTH) and richness (QMH), percent abundance (RTH, DTH) and percent richness (QMH) were determined within each division, family and genus. Out of 327 possible combinations of metrics and sample types (109 multiplied by 3), only 6 were strongly correlated with the urban index, and all were correlated with diatoms associated with RTH samples (table 9). In all six cases, relative abundance increased with increases in urban intensity, but the responses were not as uniform as with other metrics. For example, the strongest correlation observed in this group of metrics was the increase in the relative abundance (percentage) of the family Naviculaceae (|rho|, 0.793). Irrespective of the relatively high coefficient of determination , the form of the response was not especially useful in contributing to an understanding of the effects of urbanization because heteroscedasticity increased with increasing urban intensity (fig. 7*C*).

Thirteen metrics were derived from data on biomass estimates (Chl *a*, AFDM, and biovolume) obtained from RTH and DTH samples. None of these metrics were strongly related to urban intensity, regardless of the sample type considered. Only total biovolume and diatom biovolume came close (|rho|, 0.547 and 0.661, respectively) to meeting the criterion value. Furthermore, the autotrophic index (AI), which measures the ratio of AFDM to Chl *a*, was not related to urban intensity. A response in the biomass metrics to urban intensity might have been expected, however, because nutrients increased with urban intensity in water-chemistry data collected in both the spring and summer.

Several of the algae-community diversity and evenness indices responded strongly to urban intensity as shown in the RTH data, but not the DTH data. For the total algae community, Margalef’s and Shannon’s Diversity increased as urban intensity increased, but Shannon’s Dominance decreased. This finding indicates that the number of algae taxa increased along the urban gradient, and the distribution of abundances among taxa became more uniform. Shannon-Wiener diatom diversity increased as urban intensity increased until a threshold was reached at 36.6, above which there was no significant response (fig. 7*D*, table 6). This pattern observed with algae diversity and dominance is in marked contrast to that observed for fish and invertebrates.

Of the 13 diatom autoecological metrics, 4 responded strongly to urban intensity (percentage of motile forms—RTH, percentage of nitrogen heterotrophs—RTH and QMH, percentage of nitrogen autotrophs—RTH, and percentage of eutrophic taxa—QMH). The percentage of motile forms (the relative abundance of motile diatoms able to move through fine sediment and silt) increased as the urban intensity increased (fig. 7*E*). This association would indicate that siltation increases as urban intensity increases, but there is insufficient information on sediment characteristics to determine if siltation increases along the urban gradient. There is some evidence, however, based on the finding that stream depth increases with urban intensity (described below), that the hydraulic energy at the water-substrate interface of the channel decreases as urban intensity increases. If this is the case, then it would suggest that fine particles might accumulate within the epilithic periphyton as urban intensity increases. This condition would favor silt-motile diatoms over other taxa at higher levels of urban intensity.

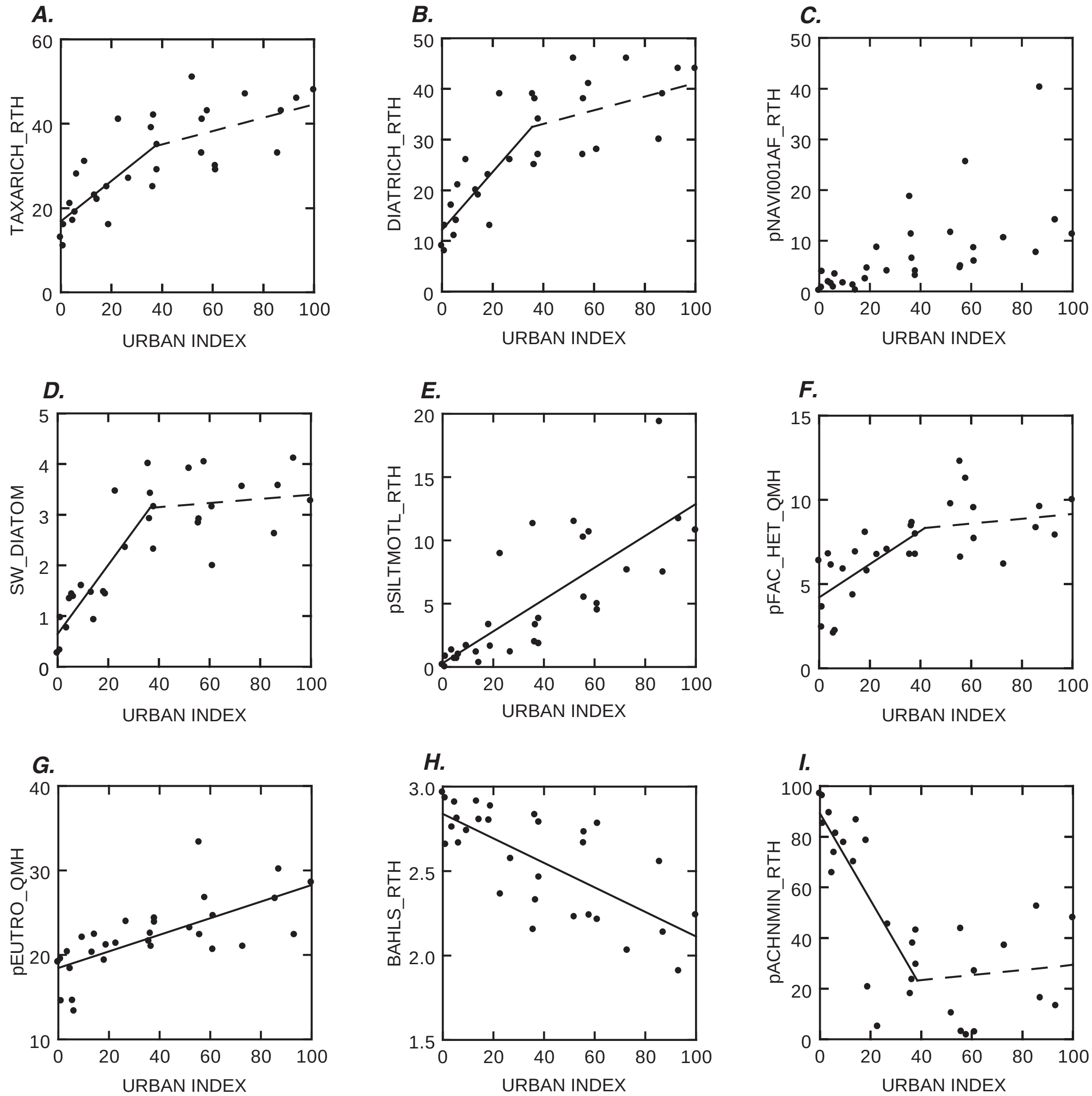



**Figure 7.** The responses of selected algae metrics to the urban index. A two-slope regression line is shown on the plots with a significant ($p < 0.05$) threshold value of urbanization. If a regression-line slope is not significantly different from zero, the line is dashed. *C*, shown without a regression line, illustrates the increasing spread of data with increasing urbanization, even when the Spearman correlation coefficient was high (|rho| = 0.793). Refer to table 6 for regression statistics. Refer to table 8 for variable definitions.

The percentage of facultative nitrogen heterotrophs (RTH and QMH) and eutrophic taxa (QMH) increased with urban intensity (figs. 7*F* and 7*G*). In contrast, the percentage of nitrogen autotrophs decreased. This pattern is consistent with the increase in concentrations of nitrogen compounds that accompanies increasing urban intensity. Facultative nitrogen heterotrophs accounted for a relatively small percentage of the algae abundances. These diatoms are associated with light-limiting habitats (Tuchman, 1996), a condition that can occur when streams become deeper and more sluggish, which appears to happen as urban intensity increases. Light-limiting conditions could also occur if high algae standing crops produced a situation of self-shading. Diatom biomass was moderately correlated with the urban index (|rho|, 0.661, determined only from the RTH data), so it is possible that the self-shading effect may increase as urban intensity increases. Conditions favoring nitrogen heterotrophs would also occur where there are high concentrations of reduced nitrogen compounds, which could be used as an energy source under low-light conditions. This condition appears to be somewhat related to urban intensity, where both TKN and light limitation (as indicated by depth) increase with the urban index. The increase in percent abundance of eutrophic diatoms along the urban gradient indicates a response similar to that of the facultative nitrogen heterotrophs. Overall, however, the relative abundance of the eutrophic diatoms was several times higher at sampling sites.

Tolerance metrics were generally good indicators of urban intensity, but only when the RTH data are considered. The Lange-Bertalot Pollution Tolerance Classes and Bahl's Pollution Index had a strong response to urban intensity (table 9). Lange-Bertalot Pollution Tolerance Classes 1, 2a, and 2b (polysaprobic, α-meso/polysaprobic, and α-meso-saprobic, respectively) represent pollution-tolerant algae, which increased in relative abundance as urban intensity increased. In contrast, the relative abundance of Class 3a declined as urban intensity increased because this class represents less tolerant (β-mesosaprobic) taxa. Bahl's Pollution Index, which ranges from 1 (all tolerant taxa) to 3 (all sensitive taxa), declined steadily as urban intensity increased, indicating that the condition of algae communities declined (fig. 7*H*). Nearly all of the diatom taxa in the RTH samples were associated with a tolerance category from the Bahl's index, so this metric was not dependent on just a few selected taxa.

The percentage abundance of the diatom *Achnanthidium minutissimum* in RTH samples was strongly associated with urban intensity. This diatom decreased as urban intensity increased (fig. 7*I*) to a threshold at 38.5, above which there was no further response (table 6). *Achnanthidium minutissimum* was the dominant diatom taxon at sites with low urban intensity (urban index values less than 20), where it accounted for 65–97 percent of diatom abundance. At higher levels of urban intensity, the relative abundance of *A. minutissimum* varied widely (0–50 percent of abundance). A similar trend was seen in the relative abundance of algae composing the Lange-Bertalot Pollution Tolerance Class 3a (β-mesosaprobic). The similarity in response of these two metrics arises because *A. minutissimum* accounts for almost all of the abundance categorized by the Lange-Bertalot Pollution Tolerance Class 3a metric. Considered an early-successional species (Peterson and Stevenson, 1992), *A. minutissimum* is often the first species to colonize a streambed after it has been scoured. Its dominance in streams with low urban intensity may be related to the observation that streams at the low end of the urban gradient were generally shallower than streams at the high end. Consequently, scouring could be more frequent at the low end of the urban gradient, which would favor *A. minutissimum*, while the deeper streams at the more urbanized end of the gradient would favor other taxa.

## Habitat Metrics

Only 5 of the 89 habitat metrics responded strongly to urban intensity (table 10). The suite of metrics summarized the stream-reach characteristics that were measured at the sampling site (such as channel morphology and stream depth) and the stream-segment characteristics that were GIS-derived (a stream segment is the section of stream from the sampling reach and extending upstream to the first major feature of hydrologic discontinuity, such as an impoundment or perennial tributary). In cases where there were multiple measurements taken within a reach (such as water depth, velocity, and substrate), the central tendency (mean) and variability (typically CV) were used as metrics. Additionally, various mathematical combinations of measured variables were used to derive metrics that quantified habitat features that could not be fully expressed by a single variable (such as hydraulic shear stress). Despite developing a comprehensive set of metrics, only a few of them had a strong correlation with the urban index. In part, this result arises from the site-selection process, which sought to select sites with minimal natural variability and with similar reach features (such as riffles and intact riparian zones). Because sites were relatively consistent in physical characteristics, habitat responses to urban intensity may have been minimized.

Although site selection may have limited the response of variables such as substrate type and size, there were some important differences in the habitat features that were related to urban intensity. Of the five habitat metrics that had a strong relation with the urban index, four were related to stream depth. Mean stream depth, the only primary (measured) variable that had a strong correlation with the urban index, increased with urban intensity (fig. 8*A*). The ratio of bankfull channel width to mean depth decreased as urban intensity increased with a threshold at an urban index value of 33.6 (fig. 8*B*, table 6). The rate of response below the threshold was about four times higher (slope, -1.1) than above the threshold (slope, -0.3). This response indicates that the channel becomes more constrained as urban intensity increases, but the active flowing water depth becomes greater. The hydraulic radius, which characterizes the cross-sectional shape of the stream channel, is closely related to this metric. Hydraulic radius was also strongly correlated with the urban index, which indicates that stream channels become deeper and narrower with increasing urban intensity.

Two of the derived metrics were based on long-term monitoring of stream conditions: water temperature, expressed as degree-days, and hydrologic variability, expressed as a statistical skew of stage. Water temperature was strongly correlated with the urban index, indicating that the mean annual water temperature increased with increasing urban intensity (fig. 8*C*). Changes in water temperature have important implications for water chemistry (Bowie, and others, 1985; Chapra, 1997) and biological communities (Ward, 1976; Coutant, 1977). In contrast, hydrologic variability was not strongly correlated with the urban index (|rho|, 0.558), and other variables derived from the stage data had even weaker correlations. Urbanization was expected to have a strong effect on hydrologic variables, such as the flashiness of streams (Schueler, 1994; Booth and Jackson, 1997). Based on this study, however, there was no strong evidence that stream flashiness increased with urban intensity. This fact may be related to the mitigating effects of impoundments that function like stormwater-detention ponds and reduce hydrologic variability. The number of impoundments upstream from the sampling sites was greater at the more urbanized sites, and this condition may have mitigated the variability of runoff from impervious surfaces after moderate storms. Impounded waters are also heat sinks that can raise the water temperature at the outfall. This process, along with runoff coming from paved surfaces (also heat sinks), may have resulted in the relation between water temperature and urban intensity.

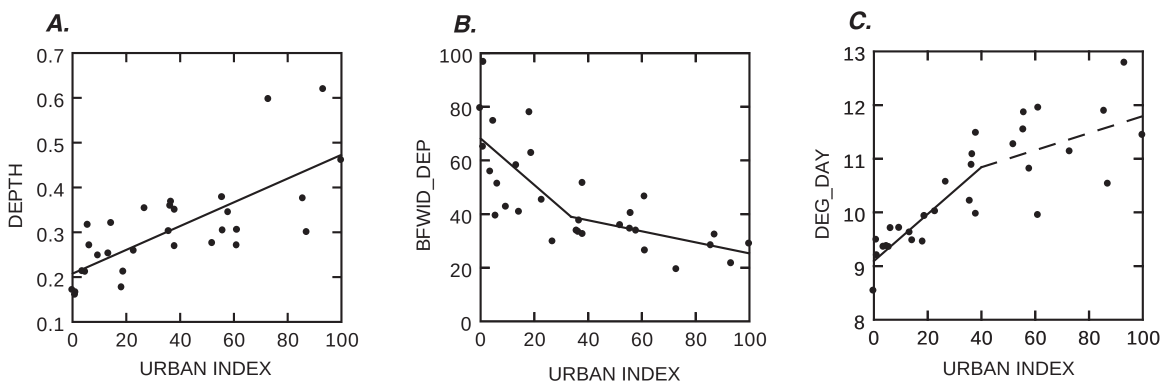



**Figure 8.** The responses of selected physical habitat metrics to the urban index. A two-slope regression line is shown in the plots with a significant ($p < 0.05$) threshold value of urbanization. If a regression-line slope is not significantly different from zero, the line is dashed. Refer to table 6 for regression statistics. Refer to table 10 for variable definitions.

## Water-Chemistry Metrics

Of the 38 water-chemistry metrics, 12 responded strongly to urban intensity (table 11). The responses to urban intensity were typically stronger in data collected in the spring than in data collected in the summer. As discussed earlier, this result is probably associated with the higher, more widespread springtime runoff, which better integrated conditions throughout the drainage basin. Summer low flows are typically more affected by conditions that are closer to the sampling site rather than conditions over the basin. The individual water-chemistry metrics, however, generally responded similarly between samples collected in the spring and summer, but the metrics from data collected in the summer had weaker correlations.

Relations between spring pesticide concentrations and the urban index were mixed. Total insecticides correlated strongly with urban intensity (|rho|, 0.841), but only six sites had concentrations above 0.010 parts per million (ppm). There were no insecticides detected at sites with an urban index value of less than 20 (12 sites). Where insecticides were detected, diazinon was the most predominant, and it accounted for most of the total insecticide concentration at all sites but one. Herbicides were detected at all but six sites, and the response of total herbicides with the urban index indicated that herbicide use increased with urban intensity. Prometon (a broad-spectrum herbicide used to control vegetation along rights of ways) and atrazine were the most frequently detected herbicides; each was detected at 18 sampling sites, and both were detected at 16 of those sites. There were relatively high total herbicide concentrations at sites with moderately low levels of urbanization. In descending order, the predominant contributors to total herbicide concentrations at these particular sites were atrazine, metolachlor, simazine, and prometon. This finding may have resulted from local agricultural applications or springtime spraying of rights of ways.

Of the spring nutrient data, TKN had the strongest response; concentrations, expressed as log transformed (fig. 9*A*), increased with urban intensity. Site 25 had the highest concentration (2.60 ppm) of all sampling sites (median, 0.34 ppm). The high concentration at site 25 was presumably caused by the discharge from a sewage-treatment plant that was within 5 km upstream from the sampling site. In contrast, spring phosphorus concentrations did not show a strong response to the urban index, although most of the lowest concentrations (less than 0.01 ppm) were observed at 9 of the 10 sites with urban index values between 0 and 15.

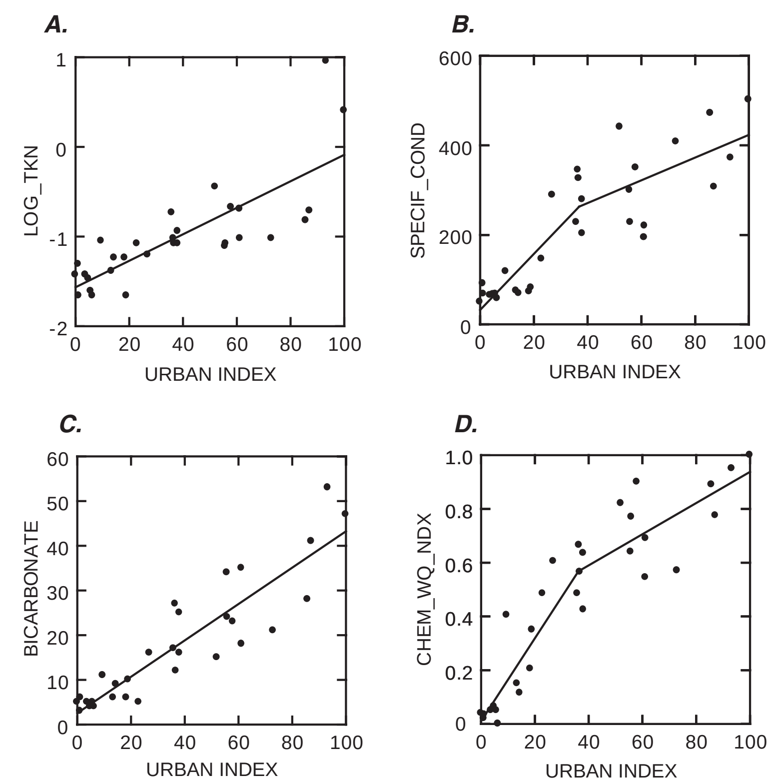



**Figure 9.** The responses of selected water-chemistry metrics to the urban index. A two-slope regression line is shown in the plots with a significant ($p < 0.05$) threshold value of urbanization. Refer to table 6 for regression statistics. Refer to table 11 for variable definitions.

Several of the field-measured constituents collected in the spring were strongly correlated with the urban index. Of these constituents, the strongest responses were observed with specific conductance and bicarbonate (figs. 9*B* and 9*C*). Specific conductance had a response threshold at an urban index value of 36.8 (table 6). The rate of response below the threshold (slope, 1.4) was more than two times greater than the rate of the response above the threshold (slope, 0.6). Alkalinity also had a strong response to urban intensity, but alkalinity was essentially a measure of the bicarbonate ions in the water from the study sites ($r^2$ of alkalinity with bicarbonate, 0.999). Consequently, the responses of alkalinity and bicarbonate with the urban index were redundant.

A chemical water-quality index (WQ index) was calculated on the basis of the spring water-quality constituents used in the ordination analyses. Redundant chemical variables were removed; their removal left a subset of 19 variables (specific conductance, pH, bicarbonate, TKN, nitrate + nitrite, total phosphorus, and the 13 pesticides that were detected) that were used in calculating the WQ index. Each variable was ranked in ascending order (1 through 30) on the basis of its measured value so that all constituents were equally weighted. The WQ-index value for each site was then determined by summing the ranked values of the constituents. The response of the WQ index with the urban index indicated that the overall water quality changed markedly with urban intensity (fig. 9*D*). Furthermore, the strength of the overall response ($r^2$, 0.868) was stronger than that of any of the individual variables (table 6). This response showed the importance of the water-chemistry constituents that individually were not always well correlated with the urban index; but when standardized and combined additively, they provided a response variable that reliably indicated the overall state of water quality.

## Basin Variables Used in Deriving the Urban Index

Spearman rank correlations were used to correlate the primary and secondary axes site scores from the BPC multivariate ordinations individually with the 53 basin variables (table 1) that were initially tested in deriving the urban index. The strongest correlations were with the primary axis site scores (table 12), as was observed with the correlations of the urban index with the BPC ordinations (tables 4 and 12). There were no strong correlations between the basin variables and any secondary ordination axes, with |rho| values less than 0.490. Furthermore, of the 29 basin variables that were not used in the urban index, none had a strong correlation with the primary axes (table 12); this result confirmed that the 24 variables used in the urban index were the most appropriate to include. However, the relative importance of each of the 24 variables to the urban index differed and was indicated by the number of correlations that were significant with the nine BPC primary axes. The urban index variables with no significant correlations were PSCNTDEN and SEI_5; variables with one significant correlation were pMRLC_42, pMRLC_82, and pRENT90; variables with two significant correlations were pMRLC_43 (refer to table 1 for variable definitions). Conversely, nine of the urban index variables (ROADDEN, pBUF_2, pBUF_4, pMRLC_2, pMRLC_4, pMRLC_21, SEI_2, pMINR99, and POP99DEN) had significant correlations with at least six of the primary axes. This frequency compares with correlations of the urban index itself, which had significant correlations with six of the nine primary axes. Generally, the urban index and its variables had |rho| values less than 0.7 for correlations with RTH, DTH, and QMH algae. The variable SEI_2, however, was not significantly correlated with the fish primary axis, but was correlated with the RTH algae (rho, -0.711). The variable pMINR99 had significant correlations with seven primary axes, which included the same six axes as the urban index and the QMH algae (rho, -0.767).

The relative strength of each urban index variable, in quantifying its relation among the nine BPC ordinations as a whole, was expressed by using the mean of the absolute rho values across all nine BPC primary axes (table 12). The 1999 population density (POP99DEN) was used as the initial surrogate for urbanization in deriving the urban index; this variable had a mean rho of 0.756, however, compared to the urban index mean rho of 0.785. Furthermore, the individual rho values across all nine primary axes were higher with the urban index than with POP99DEN. The multivariable urban index, therefore, was more effective in explaining the responses in the BPC ordinations than the population density. Conversely, road density (ROADDEN) had a mean rho of 0.800, which was slightly higher than the mean rho for the urban index, although the individual rho values were not consistently higher with ROADDEN (RTH and QMH invertebrates, RTH algae, and spring chemistry were higher with the urban index).

Of the land-use variables based on specific land use (MRLC Level II), the percentage of drainage basin in low-intensity residential areas (pMRLC21) had the highest mean rho of 0.749. Of the aggregated land-use variables (MRLC Level I), the percentage of developed land (pMRLC2) and percentage of forested land (pMRLC4) had comparable mean rho values of 0.749 and 0.752, respectively. Furthermore, these two urban-index variables had inversely similar rho values with all the BPC primary axes; the inverse relation of the rho values indicated that forested land is being replaced with developed land in the region. Of the urban index variables other than POP99DEN that were based on census data, the percentage of 1999 minority population (pMINR99) had the highest mean rho (0.751) and the highest frequency of significant correlations with the primary axis (seven). The individual rho values of pMINR99 associated with the nine primary axes, however, were not as high as those for the urban index, except for the RTH, QMH, and DTH algae.

**Table 12.** Rho values from Spearman correlations between the basin variables used to derive the urban index and the site scores of ordination primary axis for biological, chemical, and physical data.

[Bold indicates an absolute value of rho (|rho|) greater than 0.7. Mean |rho| is relative strength of basin variable among all axes. Variables that are bold were used in deriving the urban index, based on |rho| > 0.6 with POPDEN99 (first data column of table)]

| Basin variable | Site scores for ordination primary axis of biological, chemical, and physical data | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | POPDEN99 | Invertebrates | | Fish | Algae | | | Habitat | Chemistry | | Mean \|rho\| |
| | | RTH | QMH | | RTH | DTH | QMH | | Spring | Summer | |
| Urban intensity index | | | | | | | | | | | |
| Urban Index | 0.948 | **-0.892** | **-0.912** | **-0.805** | -0.643 | -0.634 | -0.632 | **-0.820** | **0.933** | **0.796** | 0.785 |
| Infrastructure variables | | | | | | | | | | | |
| **ROADDEN** | 0.964 | **-0.886** | **-0.897** | **-0.817** | -0.640 | -0.669 | -0.688 | **-0.874** | **0.912** | **0.814** | 0.800 |
| **PSCNTDEN** | .613 | -.574 | -.618 | -.523 | -.338 | -.369 | -.364 | -.517 | .621 | .436 | .484 |
| **DAMDEN** | .621 | **-.717** | -.590 | -.611 | -.648 | -.554 | -.473 | -.557 | **.742** | **.747** | .627 |
| **TRIDEN** | .858 | **-.840** | **-.891** | **-.773** | -.522 | -.576 | -.574 | **-.757** | **.839** | .663 | .715 |
| Land-cover variables | | | | | | | | | | | |
| **pBUF_2** | 0.942 | **-0.853** | **-0.844** | **-0.777** | -0.628 | -0.620 | -0.547 | **-0.740** | **0.904** | **0.749** | 0.740 |
| **pBUF_4** | -.865 | **.824** | **.846** | **.707** | .552 | .561 | .492 | **.748** | **-.860** | **-.736** | .703 |
| pBUF_9 | .064 | -.087 | -.168 | -.192 | -.262 | -.141 | -.247 | -.359 | .042 | .107 | .178 |
| pMRLC_1 | .031 | -.142 | -.132 | .012 | .099 | -.002 | .185 | -.216 | .046 | .029 | .096 |
| **pMRLC_2** | .965 | **-.873** | **-.882** | **-.771** | -.597 | -.612 | -.565 | **-.777** | **.913** | **.748** | .749 |
| pMRLC_3 | .256 | -.241 | -.098 | -.201 | -.190 | -.271 | .025 | -.099 | .356 | .184 | .185 |
| **pMRLC_4** | -.939 | **.868** | **.891** | **.741** | .583 | .642 | .554 | **.783** | **-.921** | **-.781** | .752 |
| pMRLC_5 | .247 | -.226 | -.192 | -.168 | -.096 | -.280 | -.066 | -.098 | .319 | .252 | .189 |
| pMRLC_6 | .160 | -.163 | -.066 | -.123 | -.467 | -.213 | -.377 | -.170 | .269 | .270 | .235 |
| pMRLC_8 | -.366 | .374 | .416 | .330 | .000 | -.062 | .088 | .361 | -.219 | -.141 | .221 |
| pMRLC_9 | -.146 | .043 | .014 | -.042 | -.023 | -.097 | .030 | -.131 | -.130 | -.021 | .059 |
| pMRLC_11 | .031 | -.142 | -.132 | .012 | .099 | -.002 | .185 | -.216 | .046 | .029 | .096 |
| **pMRLC_21** | .963 | **-.853** | **-.876** | **-.760** | -.604 | -.626 | -.565 | **-.784** | **.906** | **.763** | .749 |
| **pMRLC_22** | .888 | **-.759** | **-.737** | -.653 | -.392 | -.526 | -.388 | -.618 | **.826** | .586 | .609 |
| **pMRLC_23** | .872 | **-.881** | **-.823** | **-.810** | -.560 | -.565 | -.499 | -.683 | **.854** | .686 | .707 |
| pMRLC_31 | .194 | -.160 | -.118 | -.036 | -.277 | -.049 | -.066 | -.153 | .257 | .183 | .144 |
| pMRLC_32 | .557 | -.545 | -.497 | -.630 | -.519 | -.576 | -.554 | -.619 | .604 | .520 | .563 |
| pMRLC_33 | .037 | .007 | .157 | .109 | .082 | -.117 | .275 | .208 | .147 | -.025 | .125 |
| pMRLC_41 | -.229 | .173 | .200 | -.059 | -.313 | -.076 | -.122 | .087 | -.089 | .031 | .128 |
| **pMRLC_42** | -.717 | .628 | .679 | .596 | .570 | .566 | .628 | .681 | **-.732** | -.696 | .642 |
| **pMRLC_43** | -.767 | .693 | .689 | .581 | .507 | .550 | .420 | .563 | **-.798** | **-.726** | .614 |
| pMRLC_51 | .247 | -.226 | -.192 | -.168 | -.096 | -.280 | -.066 | -.098 | .319 | .252 | .189 |
| pMRLC_61 | .160 | -.163 | -.066 | -.123 | -.467 | -.213 | -.377 | -.170 | .269 | .270 | .235 |
| pMRLC_81 | -.411 | .349 | .371 | .308 | -.087 | .102 | .035 | .241 | -.318 | -.127 | .215 |
| **pMRLC_82** | -.680 | .675 | **.741** | .575 | .356 | .253 | .422 | .688 | -.633 | -.583 | .547 |
| **pMRLC_85** | .695 | -.645 | **-.701** | -.620 | -.664 | -.534 | -.680 | **-.708** | **.747** | .648 | .661 |
| pMRLC_91 | -.378 | .259 | .266 | .209 | .230 | .138 | .315 | .152 | -.352 | -.207 | .236 |
| pMRLC_92 | .414 | -.443 | -.524 | -.544 | -.502 | -.479 | -.490 | -.649 | .443 | .387 | .496 |

**Table 12.** Rho values from Spearman correlations between the basin variables used to derive the urban index and the site scores of ordination primary axis for biological, chemical, and physical data.—Continued

[Bold indicates an absolute value of rho (|rho|) greater than 0.7. Mean |rho| is relative strength of basin variable among all axes. Variables that are bold were used in deriving the urban index, based on |rho| > 0.6 with POPDEN99 (first data column of table)]

| Basin variable | Site scores for ordination primary axis of biological, chemical, and physical data | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | POPDEN99 | Invertebrates | | Fish | Algae | | | Habitat | Chemistry | | Mean \|rho\| |
| | | RTH | QMH | | RTH | DTH | QMH | | Spring | Summer | |
| Socio-economic indices | | | | | | | | | | | |
| SEI_1 | -0.100 | 0.016 | 0.053 | 0.013 | -0.402 | -0.161 | -0.129 | -0.090 | 0.007 | 0.202 | 0.119 |
| **SEI_2** | .707 | **-.721** | **-.739** | -.624 | **-.711** | -.545 | -.600 | **-.725** | **.733** | **.749** | .683 |
| **SEI_3** | -.878 | **.751** | **.791** | **.759** | .435 | .509 | .612 | **.746** | **-.764** | -.612 | .664 |
| SEI_4 | .232 | -.321 | -.354 | -.069 | -.204 | .046 | -.102 | -.223 | .290 | .331 | .216 |
| **SEI_5** | -.712 | .665 | .663 | .558 | .352 | .374 | .322 | .486 | -.679 | -.606 | .523 |
| SEI_6 | .478 | -.411 | -.431 | -.504 | -.660 | -.587 | -.545 | -.553 | .494 | .507 | .521 |
| Socio-economic variables | | | | | | | | | | | |
| AVGBED90 | 0.062 | -0.117 | -0.100 | -0.116 | -0.348 | -0.176 | -0.221 | -0.147 | 0.108 | 0.289 | 0.180 |
| ANNEX99 | .230 | -.241 | -.196 | -.359 | -.486 | -.469 | -.331 | -.361 | .261 | .285 | .332 |
| pWRK16 | -.421 | .335 | .392 | .207 | -.099 | .037 | .072 | .325 | -.332 | -.143 | .216 |
| pPOV90 | -.174 | .242 | .199 | .331 | .515 | .201 | .350 | .245 | -.185 | -.283 | .283 |
| MEDAGE99 | .553 | -.589 | -.618 | -.438 | -.461 | -.233 | -.514 | -.556 | .531 | .550 | .499 |
| MEDHHI99 | .337 | -.371 | -.381 | -.328 | -.616 | -.410 | -.430 | -.446 | .411 | .532 | .436 |
| **pFHHF90** | .722 | **-.781** | **-.830** | **-.717** | -.397 | -.561 | -.443 | **-.749** | .685 | .633 | .644 |
| **PHOUSL80** | .751 | **-.783** | **-.834** | **-.763** | -.553 | -.436 | -.632 | **-.754** | .690 | **.735** | .687 |
| pOWN90 | .401 | -.391 | -.388 | -.364 | -.554 | -.349 | -.451 | -.471 | .400 | .499 | .430 |
| **pRENT90** | .690 | -.553 | -.579 | -.578 | -.407 | -.603 | -.464 | -.517 | **.703** | .493 | .544 |
| **pMINR99** | .811 | **-.709** | **-.810** | **-.780** | -.656 | -.690 | **-.767** | **-.802** | **.811** | **.731** | .751 |
| p65P90 | .574 | -.453 | -.502 | -.376 | .027 | -.076 | -.277 | -.391 | .431 | .210 | .305 |
| pMALE99 | -.399 | .386 | .378 | .414 | -.018 | .053 | .091 | .344 | -.302 | -.145 | .237 |
| **POP99DEN** | 1.000 | **-.849** | **-.877** | **-.745** | -.584 | -.614 | -.630 | **-.808** | **.921** | **.772** | .756 |
| **PDEN9099** | .927 | **-.781** | **-.803** | -.618 | -.563 | -.625 | -.494 | **-.738** | **.891** | .670 | .687 |

In summary, invertebrate-, fish-, habitat-, and chemistry-site scores, which represent the derived ecological gradients, had the strongest correlations with the urban index (|rho|, 0.814–0.912); the algae site scores had moderate correlations (|rho|, 0.640–0.688). In general, these results indicate that increasing urban intensity, as measured by the urban index, is associated with a decline in biological, physical, and chemical stream conditions. Similar associations exist among individual measures of urban development and stream condition. Among the infrastructure variables, road density had the strongest (negative) correlation with the site scores for the BPC data. Among the land-cover variables, developed and forested land cover had the strongest correlations with the site scores, although they responded inversely because the two variables were essentially mutually exclusive. Additionally, as the landscape becomes more urbanized, forested land becomes more fragmented but developed land coalesces into larger patches. Several individual socioeconomic variables, including population density, percentage of female-headed households, percentage of minority population, and percentage of housing stock built before 1980, had strong negative correlations with invertebrate-, fish-, habitat-, and chemical-site scores. Individual socioeconomic variable results are difficult to interpret. In conjunction with the socioeconomic index results, it appears that basins with older housing stock and high household and population density (SEI_2) adversely affect the BPC characteristics of streams, and basins with predominantly rural populations (SEI_3) have a positive association with the BPC characteristics.

# Relations among Landscape Changes, the Urban Index, and Biological, Physical, and Chemical Characteristics

The results of this study showed that increases in urban intensity were associated with declining BPC conditions, similar to the results seen in other studies (May and others, 1997; Kennen, 1999; Morley and Karr, 2002). Of the biological components analyzed, the greatest number and strongest responses were seen with the invertebrates, including the invertebrate community ordination and many invertebrate metrics. Of the physical and chemical components analyzed, the strongest responses were seen with the ordination of water-chemical constituents and with certain water-quality parameters. On the basis of the cumulative responses shown by the BPC characteristics, it was determined that the approach used to define an *a priori* urban index was effective in identifying the features that represent an urban gradient.

At the onset of the study, there had been no validation that the urban index or any of the 53 basin variables would relate to the BPC characteristics of a stream. The urban index was constructed from 24 of the basin variables that were highly correlated with (and included) population density. Although population density was considered a driving factor behind urbanization, there was no contention that it alone would be equated with stream disturbance. In addition to population density, the urban index represents differences among the study basins in terms of land cover and factors such as infrastructure, socioeconomic characteristics, and housing that are not generally quantified by simple measures of land cover (McMahon and Cuffney, 2000). This type of composite index of urban intensity, therefore, combines a number of individual environmental-condition measures that provide distinct information about different dimensions of complex urban systems. A closer inspection of the response of each of the 53 basin variables to the BPC characteristics provides insight into landscape changes that are associated with urbanization.

## Landscape Indicators of Urbanization

In place of an index to quantify urbanization, other studies have used the amount of impervious surface in a basin as a surrogate for urban intensity. Several studies have shown impervious surface to decrease ground-water recharge (Dunne and Leopold, 1978; Klein, 1979; Arnold and Gibbon, 1996), change discharge patterns (Espey and others, 1965; Leopold, 1968; Seaburn, 1969; Hirsch and others, 1990), increase stream-water temperature (Galli, 1991; LeBlanc and others, 1997, Talmage and others, 1999), and increase the delivery of contaminants to streams (Howard and Haynes, 1993; Lenat and Crawford, 1994; Porcella and Sorenson, 1980; U.S. Geological Survey, 1999; Neal and Robson, 2000). Originally, because direct measures of impervious surface were not available for the basins studied, we planned to estimate impervious surface area from land-cover data (McMahon and Cuffney, 2000). It was determined, however, that the procedure for estimating impervious-surface area relied on a subset of the basin variables used to derive the urban index. Therefore, estimated impervious area was redundant with other variables used in the urban index; consequently, it was not a suitable explanatory variable.

Despite the exclusion of impervious surface in the urban index, it was still important to relate urban intensity to impervious surface to facilitate comparisons with other published studies, such as Arnold and Gibbons (1996) and Booth and Jackson (1997). This process was accomplished by regressing the urban index in relation to estimated impervious surface area for the 206 candidate sites considered in the site-selection process (fig. 10). The resulting regression equation was used to estimate the percentage of impervious surface area in the basin on the basis of the urban index. This is an estimate of total impervious surface rather than effective or connected impervious surface area. Although impervious surface is an important consequence of urbanization, Karr and Chu (2000) concluded that total impervious area can be independent of many activities associated with urbanization that affect stream characteristics. This study, therefore, used the multimetric urban index as a more comprehensive indicator of urban intensity.

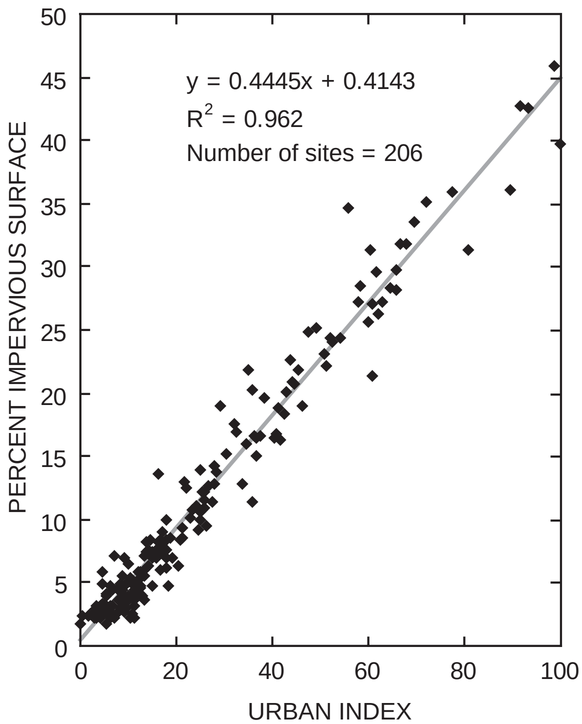



**Figure 10.** Relation between the urban index and percentage of impervious surface derived from the 206 candidate sites. This relation was used to estimate impervious surface at various levels of urban intensity.

Among the infrastructure variables that were tested, road density had the strongest association with the BPC conditions. In addition, it had the strongest correlations with BPC conditions of any of the candidate variables. Roadways can account for significant impervious surface area in a drainage basin, which has an acute effect on stormwater runoff (Arnold and Gibbons, 1996). The urbanization process is intrinsically linked with development of roadway networks, and May and others (1997) suggested that road density could be used as a surrogate for total impervious surface area. Roads can affect the rate of stormwater overland flow, and this runoff may flush contaminants that have accumulated on road surfaces into streams, including salt from wintertime deicing (Buckler and Granato, 1999; Granato and Smith, 1999; Forman, 2002). Some of the observed responses between water chemistry metrics and the urban index (such as increasing specific conductance) may therefore have been related, in part, to road density.

Among the MRLC land-cover variables, forested and developed land had the strongest correlations with the BPC conditions. Especially when measured at the general scale (level-1 classification), these two land-cover variables had strong associations with invertebrate-, fish-, habitat-, and chemical-site scores (|rho| > 0.7), but the correlations of these two land-cover variables with the BPC site scores have opposite signs (developed land shows a negative response and forested land shows a positive response). This result is not surprising, not only because these two variables have a strong negative correlation with each other, but also because of the inverse relation often observed between these two land-cover variables and stream conditions (Kennen, 1999). The direction of the responses of developed and forested land with the BPC site scores also held true at the more specific land-cover-classification scale (level-2 classification), although the correlations were generally weaker, especially with specific forest types. Landscape changes from urbanization were, therefore, most apparent at the general-scale (level 1) classification.

Urbanization profoundly affects patch characteristics of the landscape, particularly when examined for developed and forested patches at a level-1 classification. When patch dynamics are considered collectively for the land classes, the percentage of basin area consisting of the largest patch decreases (landscape fragmentation is increasing), the variability in patch size increases (fewer large forested patches, many more smaller patches of forest), patches become more isolated, and the homogeneity decreases. Together, these changes indicate that the landscape becomes more fragmented and disconnected as urbanization increases, and implies that streams become more associated with developed areas as the forest fragments.

The association between the socioeconomic indices and the BPC response was not unexpected, because each of these indices has a dimension related to population density, which is the primary variable used in deriving the urban index. SEI_2, which is associated with college-educated residents living in areas of high housing and population density, had a negative correlation with stream conditions. SEI_3 is associated with rural households in areas of reduced population density and had a positive correlation with stream conditions. The association of the individual socioeconomic variables with stream conditions is difficult to interpret. It is unlikely that there is a causative relation between any of these individual variables (such as percentage of female-headed households and percentage of population composed of minorities ) and stream conditions. For example, the SEI_2, percentage of female-headed households, and percentage of population composed of minorities all responded positively to an increase in urban intensity, although they likely represent different socioeconomic sectors of society. A more likely explanation, therefore, is that these variables are associated with a physical development pattern (such as high housing density) that also affects stream conditions.

## Biological Responses to Urban Intensity

The biological metric and multivariate analyses were effective in characterizing urban intensity, but the ability of metrics to detect responses was strongly affected by the biological community (fish, invertebrates, and algae), by the sample type (RTH, DTH, QMH), and by the characteristics of the metric. Only a small percentage of the metrics had a strong correlation with the urban index. Richness metrics were the best for examining invertebrate responses across urban intensity. Considered together, these metrics indicated that with urbanization, the EPT taxa are replaced with non-insect taxa, and that this shift is associated with taxa that are progressively more pollution-tolerant. Only a few abundance-based metrics (mayflies, stoneflies, orthoclad midges, and Coleoptera) responded strongly to urban intensity. Many of the invertebrate metrics that were based on quantitative data (RTH) displayed threshold responses, but metrics based on qualitative (QMH) data did not. These thresholds, including the one from the RTH ordination, ranged in urban index values from 34.4 to 43.3, which equates to an impervious-surface estimate from 15.7 to 19.7 percent.

The response of fish communities to urban intensity contrasted with the response of invertebrate communities in several ways. The fish metrics and ordination had responses to urban intensity that showed thresholds at urban-index values that ranged from 31.3 to 41.2, which equates to impervious-surface estimates from 14.3 to 18.7 percent. This range of values corresponds closely to the threshold values for the invertebrate responses. However, almost all of the fish metrics that were strongly related to urban intensity were based on abundance rather than on richness, and the strongest responses were associated with one taxon (cyprinids). In all cases, the relation between fish metrics and urban intensity was not as strong as the relation for invertebrates. Compared to other parts of the eastern United States, the number of fish species in New England is relatively low and often only a few species are found in a particular basin. A partial explanation for this is that migration of fish between basins can be deterred when patterns of hydraulic connections within drainage networks are restricted (Osborne and Wiley, 1992; Frothingham and others, 2002). Most of the drainage basins sampled in this study are relatively isolated from one another either because of interruptions associated with dams or because the basins drain directly into the sea. Under these circumstances, migration is reduced and the fish community at a site can become dominated by cosmopolitan species with high abundances (such as white sucker).

Limited fish migration was probably a factor affecting the community structure at many of the sites; therefore, the invertebrate communities were a more reliable indicator of urbanization. Armstrong (2001) reported that fish communities in the Ipswich River (site 18 in our study) were likely defined by the frequency of continuously flowing riffles in that system. Although only reaches that met a pre-determined set of criteria were sampled in our study, it is very likely that the frequency of quality reaches (those with coarse substrate riffles and intact riparian zones) decreases as basins become more urbanized. This observation is supported by the difficulty in finding appropriate sampling sites at the high end of the urban-intensity gradient during the site-selection process. Furthermore, the decrease in the fluvial-dependent fish taxa with increasing urban intensity is consistent with the increase in basin impoundments with urban intensity, and is probably associated with a transition from a lotic- to a lentic-flow regime, as indicated by stream depth.

The response of the algae community was best expressed by the RTH data, both in the community ordination and in the metrics. The algae associated with the DTH and QMH data had slightly weaker relations to urban intensity on the basis of the community ordinations, and showed almost no strong relations based on metrics. Algae, therefore, are probably responding to other environmental factors at least as strongly as they are responding to the basin factors used to derive the urban index. This assessment is based on the comparisons of the algae ordinations with those of habitat and water chemistry (table 5). For all representations of algae communities (RTH, DTH, QMH), the correlations were stronger with habitat than with the urban index, especially with QMH algae (|rho|, 0.762), which is strongly dependent on microhabitat diversity by definition. Similarly, the RTH- and DTH-algae correlations were much stronger with the summer water-chemistry data (|rho|, 0.721 and 0.727). Nevertheless, the algae ordinations and metrics that responded to urban intensity did so with threshold values ranging from 33.0 to 41.7. These threshold values equate to impervious-surface estimates from 15.1 to 19.0 percent, and were similar to the values seen in the other biological communities. Unlike the fish and invertebrate communities, algae taxa richness and diversity increased as urban intensity increased from low- to mid- range of the urban index, and remained high throughout increases in urban intensity. A possible explanation for this pattern may have been that as urban intensity increased, the increase in nutrients and water temperature was beneficial for the algal growth.

Diatoms dominated the algae community; therefore, diatom metrics exhibited the strongest responses to urban intensity. Diatom richness and Shannon-Wiener diatom diversity responded positively to the urban index, and Bahl's index responded negatively. These results indicate that pollution-sensitive diatom taxa decreased with urban intensity, and that diatom richness and diversity increased. These findings are an example of increasing diversity that is not indicative of improving conditions. Furthermore, the positive response of the relative abundance of the facultative nitrogen heterotrophic diatoms and the eutrophic diatoms with the urban intensity is consistent with nutrient enrichment of the water that often accompanies urbanization.

## Physical and Chemical Responses to Urban Intensity

Changes to the physical and chemical environment caused by urbanization can profoundly affect biota. It is important, therefore, to understand how habitat, water temperature, stream stage, and water chemistry change with urbanization, and then relate these findings to other studies of urbanization. Results from this study showed that effects of urbanization on water-chemistry variables were more frequent and stronger than the effects on the habitat variables. Furthermore, the ordination of habitat variables (eigenvalue 0.208) was not as strong as the gradient from the spring and summer chemical-data sets (eigenvalues 0.647 and 0.538). As discussed, the relatively weak pattern in the habitat ordination may be a result of the site-selection process. Maintaining consistency among habitat features enhances the ability to detect changes associated with basin-wide patterns of urbanization, although subtle habitat changes caused by urbanization were likely. Rogers and others (2002) point out that differences in habitat features can mask the effects of chemical stressors on biota; therefore, the efforts to hold habitat features relatively constant for this study enhanced the ability to detect chemical effects of urbanization. In this study, this consistency in habitat features was primarily recognized by observing that only a few of the habitat metrics were strongly correlated with urban intensity, and these metrics were associated with either stream depth or temperature. Furthermore, physical features peripheral to the flowing water did not change appreciably with urban intensity.

Stream depth increased as urban intensity increased and was not related to basin size. Depth was the prevailing variable in the bankfull width to depth ratio (negative response to increasing urban intensity) and to hydraulic radius (positive response to increasing urban intensity). Together, these two metrics indicated that the stream channel was becoming more U-shaped as urban intensity increased. This pattern is suggestive of the erosional phase in channel changes associated with urbanization (Paul and Meyer, 2002). Although the sampling reaches for this study were not obviously channelized, there were indications that the streambanks at several of the more urbanized sites had been constrained by large boulders and fill material. Consequently, high streamflows could cause the streambed to degrade and create a deeper channel.

Of the two physical characteristics that were measured over time (water temperature and stream stage), only water temperature had a strong relation to the urban index. The average daily temperature, calculated over a 1-year period, indicated that water temperature increases as urban intensity increases. The loss of streamside shading caused by removing riparian forests, heating of runoff water from roads and parking lots, and the heat islands formed by cities all contribute to increases in water temperature (Galli, 1991; LeBlanc and others, 1997; Paul and Meyer, 2002). An increase in water temperature is important because temperature can affect the distribution of aquatic organisms (Vannote and Sweeney, 1980; Hogg and others, 1995; Rathert and others, 1999; Giorgi and Malacalza, 2002; Rosenfeld and others, 2002; Schiemer and others, 2002; Smol and Cummings, 2002).

The lack of any definitive response between stage variability and the urban index was unexpected on the basis of other studies (Paul and Meyer, 2000). The increases in impervious surface area and stormwater discharges associated with increasing urban intensity were expected to result in an increase in the flashiness of streams (higher peak discharges with shorter durations, more low flows, more flow variability) (Espey and others, 1965; Leopold, 1968; Seaburn, 1969; Hirsch and others, 1990). Measures of hydrologic variability showed no strong relation with urban intensity. Morley and Karr (2002) also found weak correlations between urban intensity and hydrologic disturbance. The large number of natural and artificial ponds and wetlands in the study basins may be partially responsible for ameliorating the extremes in discharge normally associated with urbanization. On the basis of the density of dams (number of dams/100 $km^2$ of basin area), the number of impoundments increases as urban intensity increases. These impoundments seem capable of mitigating hydrologic effects, although retention ponds in the Pacific Northwest did not fully mitigate hydrologic effects associated with urbanization (Booth and Jackson, 1997). More intensive monitoring of stage and discharge, along with a more detailed evaluation of wetlands and flow-control structures, would be needed to develop a better understanding of the complex hydrologic relations associated with urbanization and flow characteristics in these basins.

Changes in water chemistry were strongly associated with changes in urban intensity, especially in the ordination of the data collected in the spring. Water chemistry in the spring may be more representative of urban impacts than water chemistry in the summer because a larger proportion of streamflow in the spring originates from surface runoff rather than during summer flows, when ground-water flows dominate. Consequently, samples collected in the spring are more directly linked to the modifications of the landscape. Heteroscedasticity increased as urban intensity increased, particularly among the 11 sites with a urban index value greater than 50 (impervious surface, about 22 percent). As with habitat, this tendency probably indicates an increase in the variability of anthropogenic factors, such as wastewater-treatment plants and storm sewers, which affected water chemistry at higher urban intensities. Specific conductance and the chemical water-quality index (composite of chemical parameters) both showed a response threshold at a urban index value of about 36 (impervious surface, about 16 percent). Of the nutrients, TKN had the strongest positive response to the urban index. This reduced form of nitrogen often originates from domestic sources. Specific conductance and bicarbonate were the two field parameters that best responded (positively) to the urban index. Effluent from wastewater-treatment plants and road-salt runoff could account for the increase in specific conductance (Granato and Smith, 1999; Heisig, 2000). The increase in bicarbonate associated with increases in urban intensity may be tied to the slow erosion of concrete structures, which are lime- (calcium-carbonate) based, and to the application of lime to lawns.

## Attributes of the Multivariate Analyses

Indirect-gradient analysis (ordination coupled with regression) provided a useful tool for analyzing, interpreting, and comparing BPC responses to urban intensity. One advantage of this approach is that it provides a consistent method not only for comparing diverse biological assemblages (fish, invertebrates, and algae), but also for comparing physical and chemical responses. In contrast, analytical consistency is difficult to obtain by using metrics. This difficulty exists because, with few exceptions (some multimetric indices), metrics are tailored to represent ecological attributes of specific assemblages of organisms (EPT for invertebrates, nitrogen heterotrophs for algae, insectivores for fish) or physical and chemical characteristics (temperature, nitrogen). Ordination, by focusing on the pattern of sites along an axis of variation (derived ecological gradient), provides a consistent approach that facilitates comparisons of responses among BPC data sets. Ordination also considers the response of the whole community rather than segments of the community, so it can be considered a more holistic approach to assessing biological responses.

Indirect-gradient analysis provides three measures that are important for interpreting BPC responses to urban intensity: (1) eigenvalues measure the overall strength of the ecological gradient derived from the data; (2) correlations between site scores and the urban index measure the association between the derived ecological gradient and the *a priori* urban gradient, and thus indicate whether the BPC components respond to urban intensity; (3) correlations between site scores (such as fish-site scores in relation to habitat-site scores) measure the association between ecological gradients to determine if the BPC components respond to each other. In all cases, the primary ordination axis (axis 1) was closely associated with the *a priori* gradient of urban intensity (the urban index), indicating that BPC changes were associated with urbanization. Eigenvalues for DTH algae, QMH algae, and habitat ordinations were, however, below 0.3, indicating that the derived gradients in these ordinations were relatively weak. Despite this result, their relations with the urban gradient were still fairly strong.

Of the biological data sets, RTH- and QMH-invertebrate data showed the strongest overall responses to urban intensity (table 6). The QMH response was essentially linear, whereas the RTH data showed a threshold response at an urban index value of 34.4. The response of the fish community showed a threshold at an urban index value of 37.4; however, above this value, changes in the fish community no longer appeared related to urban intensity. Similar patterns were seen for the algae (RTH, DTH, and QMH), which showed thresholds at urban index values between 33.0 and 34.5. Overall, the biological data sets that showed a threshold response did so at similar urban index values (33.0–37.4), which equates to impervious-surface estimates from 15.1 to 15.7 percent.

Ordination of the habitat data showed a response threshold that was slightly higher than that observed for the biological data sets. Habitat changed rapidly up to an urban index value of 43.3, equating to an impervious-surface estimate of 19.7 percent, above which habitat no longer responded to changes in urban intensity. The eigenvalue associated with the habitat ordination (0.208), however, indicated that the environmental gradient derived from the habitat data was not strong. As discussed, this result is probably associated with the design of this study to minimize differences among sites that were related to substrate, riparian, and channel conditions. Nevertheless, the habitat gradient showed a strong relation with the urban index (|rho|, 0.820). Ordination of the water-chemistry data showed a strong ecological gradient for the spring and summer water-chemistry samples (eigenvalues, 0.647 and 0.538, respectively). The response to the urban index was linear in both cases, with the spring water-chemistry sample having a stronger correlation (|rho|, 0.933) than the summer water-chemistry sample (|rho|, 0.796); however, the summer water-chemistry sample showed important relations in different respects. The correlations of the RTH- and DTH-algae ordinations with the ordination of the summer water-chemistry data were the strongest these algae had with any other data set (including the urban index); this finding indicated that these algae communities are probably responding to environmental factors in the short term. Further analysis would be necessary to determine the relative effect that land use, water chemistry, and habitat have in defining algae communities at the study sites.

## Implications for Water-Resource Monitoring and Management

All response variables began to change as soon as conditions departed from background, which strongly indicates that the biological communities were not resistant to urbanization and that urban intensity at any level alters the BPC characteristics of these streams. One of the expected results of this study was that at least some of the biological indicators would resist change at low to moderate levels of urban intensity, and that this delayed biological response (resistance to change) would be followed by a rapid response at moderate levels of urban intensity (an initial threshold response), and finally taper off at high levels of disturbance. This type of response pattern has been observed for agricultural gradients (Cuffney and others, 2000), but this was not the case in this study.

For the BPC indicators of water quality displaying a strong response to urban intensity, the rate of change was either uniform across the gradient of urban intensity or most rapid at levels from 0 to 30–45 percent of the maximum level of urban intensity (estimated impervious surface, about 14–20 percent). This latter response resulted in a threshold that was characterized by a rapid rate of change at low levels of urban intensity, followed by a much slower rate of response or no response at moderate to high levels of urban intensity. In a study of the effects of urbanization on small streams in the Puget Sound Ecoregion, May and others (1997) observed a rapid initial decline in the invertebrate communities as total impervious area (TIA) increased from 0 to 10 percent; as TIA increased above this range, the response of the invertebrate communities decreased. Although May and others (1997) did not describe the response as a threshold, it appeared comparable to the threshold response described in our study. Threshold responses have important implications for monitoring and mitigating urban effects. Metrics that exhibit a threshold response (such as diatom richness) may not be the most suited for monitoring urban effects because their responses are not consistent over the range of conditions being monitored, but metrics that respond uniformly (such as EPT richness) would be effective for monitoring urban effects. Depending on the purpose of the monitoring effort, it is often best to choose a metric that responds uniformly over the disturbance gradient; otherwise, there is a risk that detectable changes would occur only at relatively modest levels of urban intensity.

The uniformity in threshold levels across response variables was one of the more unanticipated results of this study. Thresholds were evident for biota (fish, invertebrates, and algae) and for habitat and chemistry. In all cases, urban-intensity levels at which the threshold was detected were fairly consistent (31.3–43.3), regardless of whether the response was for biological, physical, or chemical data sets. The reason for this uniformity in the responses is unclear. The observation that biological thresholds would follow changes in the physical and (or) chemical variables was expected; however, biological thresholds that were coincident with thresholds in driving variables were not expected. This finding may indicate that some unmeasured factor may be driving physical and chemical changes at the urban-intensity level around the threshold values. Further examination of basin variables used to characterize urbanization could help explain what might be responsible for the observed thresholds, and additional studies that are focused on sites with urban index values of 30–40 (14–20 percent estimated impervious surface) are needed to resolve the ecological implications of these responses.

Although the urban index did not explain responses in many of the variables tested, the response variables that had a clear relation to the urban index have potential use for monitoring stream conditions in urbanizing basins. Overall, the multivariate analyses were a more powerful tool for assessing patterns of change than were community metrics. Fish, invertebrates, algae, chemistry, and habitat all showed a clear response to urban intensity on the basis of indirect-gradient analyses, but relatively few metrics showed a definitive response. The use of multivariate analyses, however, has disadvantages. Primarily, discussions of multivariate analyses can be viewed as somewhat esoteric and bewildering to nontechnical audiences, so the challenge in using multivariate analyses is to present the interpretation in an intuitive manner. In this regard, simple scatterplots of site scores against an external gradient are useful, provided that a more descriptive name, such as “fish-condition index,” is applied to the derived ecological gradient represented by the site scores.

Compared to multivariate analyses, community metrics are widely used (Barbour and others, 1999) in water-quality monitoring and can be more clearly understood by nontechnical audiences, especially when the response is linear over the entire gradient of disturbance. From almost 700 metrics examined in this study, only 98 showed a meaningful response to urban intensity and most of these (57) were associated with invertebrates. These findings indicate the usefulness of invertebrates for biological monitoring, and the differences in life history of the three biological groups (fish, long life spans; invertebrates, moderate life spans; algae, short life spans) were likely a factor in the invertebrate metrics responding most consistently to urban intensity. As shown by results from this study, the most consistent (linear) biological indicators of urbanization were metrics associated with invertebrate taxa richness. These metrics included total and EPT richness, percentage richness of non-insect taxa, and richness of tolerant taxa, all of which showed a very strong linear relation across the gradient of urban intensity. Because these metrics are based on richness rather than on abundance, the QMH data (presence-absence) generally were best at showing changes in urban intensity. Neither the fish nor the algae metrics showed as strong a linear response as did these invertebrate metrics; this difference indicates that multimetric indices for these communities are probably more appropriate. With regards to water quality in this study, bicarbonate (or alkalinity) and specific conductance responded most consistently to urban intensity. Furthermore, these two constituents are relatively easy to measure in the field and are recognized as important in monitoring water quality (Cordy, 2001).

As mentioned previously, this study represents one of three completed pilot studies as part of the USGS NAWQA program to address the effect of urbanization on biological, physical, and chemical characteristics of streams. Additionally, 11 similar studies have been started throughout the United States to determine the effects of urbanization on stream ecosystems (Couch and Hamilton, 2002). Findings from these studies may be integrated to determine how urbanization changes the landscape in geographically diverse regions of the country, and how BPC characteristics respond to urbanization in the different regions. On the basis of these findings, it may be determined that there are specific elements of urbanization common throughout the country that are important driving variables that adversely affect stream ecology. Furthermore, additional data analyses could determine whether there are specific variables that explain changes in the biological responses that were not particularly well correlated to urban intensity. As discussed in this report, for example, the response of algae communities was more strongly related to habitat and water chemistry than to urban intensity. Variables from these data sets may be evaluated to determine if specific habitat and water-chemistry characteristics might be causing changes to the algae community, and how these characteristics might be related to urbanization.

## Summary and Conclusions

Urbanization of river basins is known to cause changes in aquatic ecosystems, but how aquatic ecosystems change in response to increasing urban intensity is less clearly understood. First, understanding how biological, physical, and chemical (BPC) components of a stream respond to increasing urban intensity involves identifying which landscape and land-use features within drainage basins have a quantifiable effect on the BPC ecological components. Second, the response of specific characteristics within each of the BPC components to drainage-basin change is determined. This determination is important to water-resource managers who assess and monitor effects brought about by urbanization.

Single-variable surrogates for urban intensity, such as population density or measures of impervious surface, are often used to represent urban intensity in a drainage basin. Many factors, however, can be associated with ecological disturbances, which make it difficult to predict how ecological components of a stream will respond to specific aspects of urbanization, particularly in different geographical locations across the United States. These issues are effectively assessed by developing and implementing a nationally standardized study design, and then using a combination of univariate, multivariate, and multimetric approaches to analyze and model responses to urbanization. To address the effects of urbanization on BPC characteristics of streams nationwide, the National Water-Quality Assessment (NAWQA) Program of the U.S. Geological Survey (USGS) designed nationally consistent studies to investigate the relations between the varying urban intensities of drainage basins and stream ecology in distinct environmental settings associated with major metropolitan areas. The design of these Urban Land-Use Gradient (ULUG) studies was tested with three pilot studies that were conducted in the humid Northeast around Boston, Massachusetts; the humid Southeast around Birmingham, Alabama; and the semiarid West around Salt Lake City, Utah. This report reviewed the findings of the ULUG study in the humid Northeast around Boston, Massachusetts.

This report presents the results of using a multimetric index of urban intensity to characterize the urban intensity among streams in a relatively homogenous environmental setting and the responses of various BPC conditions of the stream to urban intensity. The report (1) discusses the effectiveness of the urban index to represent the aspects of urban intensity that accounted for changes to the BPC characteristics of streams; (2) describes how the BPC characteristics of streams responded to urban intensity; (3) discusses which BPC variables were useful indicators of urban intensity; and (4) identifies specific characteristics of urbanization that were most strongly associated with BPC responses, so that those characteristics may be used in constructing better indices of urban intensity. The study used a network of 30 sites within 80 mi of Boston, Massachusetts. The sites represented a gradient of low to high urban intensity that was quantified by the urban index. Furthermore, the sites were within a relatively homogenous natural setting so that changes to the BPC characteristics among sites could be attributed to urban intensity rather than to natural variability.

To ensure that natural variability was relatively homogenous among the sampling sites, the sites selected were within a single United States Environmental Protection Agency Ecoregion and within contiguous United States Forest Service Ecological Units. An urban index that represented urban intensity for the 30 study sites was developed from 24 land-use and census-based variables. These variables were included in the urban index because they were strongly correlated with population density of the 30 drainage basins, and because they were thought to be a better representation collectively of urban intensity than any single variable.

During 2000, biological (invertebrates, fish, and algae), physical, and chemical data were collected at the sites. Biological data were collected once in 2000. Physical data were collected once for all sites, although stage and temperature data were monitored continuously for a year. Chemical data were collected twice in 2000. Data were analyzed with the use of multimetric and multivariate techniques. Multimetric analysis was accomplished by individually regressing the BPC variables with the urban index. Multivariate analysis was accomplished by first using indirect-gradient analysis (CA and PCA ordinations) to identify and derive ecological gradients of changing conditions within each BPC data set, and then regressing these gradients (quantified by site scores) with the urban index. These two types of analysis were used because the multimetric approach indicates which BPC variables are most responsive to changes in urbanization, and the multivariate approach indicates the extent that the structure within the multivariable data set changes with urbanization.

The study results indicate that the urban index effectively defined an urban gradient. Furthermore, the urban index can be used for further understanding of how urban intensity affects stream characteristics and how these effects relate to changes on the drainage-basin landscape. Generally, increases in urban intensity, as measured by the urban index, were associated with declining BPC conditions of the streams. Results of the multivariate analysis, which used site scores to represent derived BPC ecological gradients, showed that the invertebrate-, fish-, habitat-, and chemistry-site scores had the strongest correlations with the urban index (|rho|, 0.814–0.912); the algae site scores had more moderate correlations (|rho|, 0.640–0.688). Results of the multimetric analysis showed the variables most responsive to urban intensity were EPT (Ephemeroptera, Plecoptera, Trichoptera) taxa for invertebrates; cyprinid taxa for fish; diatom taxa for algae; bicarbonate, conductivity, and nitrogen for chemistry; and water depth and temperature for physical habitat. Many of the responses showed a threshold at around an urban index value of 35. This threshold indicates that the variable was not responding predictably to higher levels of urban intensity and that aquatic health changed the most between low and moderate levels of urban intensity. The ability to discern and interpret threshold responses may be useful to land-use managers and other resource managers, because a biological, physical, or chemical variable that is used to characterize stream health over a gradient of urban intensity would not be a reliable indicator if it does not respond predictably.

Associations between the individual variables that were used in deriving the urban index and the measures of stream condition, as expressed by the BPC gradients, were stronger in some cases than in others. The importance of this finding is that the urban index variables that showed the stronger correlations were likely more important in causing responses observed in the BPC data sets, and these variables could be used in constructing more specific urban indices. Among the infrastructure variables used in the urban index, road density had the strongest (negative) correlation with the BPC gradients. Among the land-cover variables, developed and forested land cover had the strongest correlations, although they responded inversely because the two variables were essentially mutually exclusive. In general, as the landscape becomes more urbanized, forested land becomes more fragmented, but developed land coalesces into larger patches. Several socioeconomic variables, including population density, percentage of female-headed households, percentage of minority population, and percentage of housing stock built before 1980, had strong negative correlations with the BPC gradients. It is unlikely, however, that there is a causative relation between any of these census-based variables and stream conditions. It is more probable that these variables are associated with a development pattern in a drainage basin (such as high-housing density) that affects stream conditions.

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