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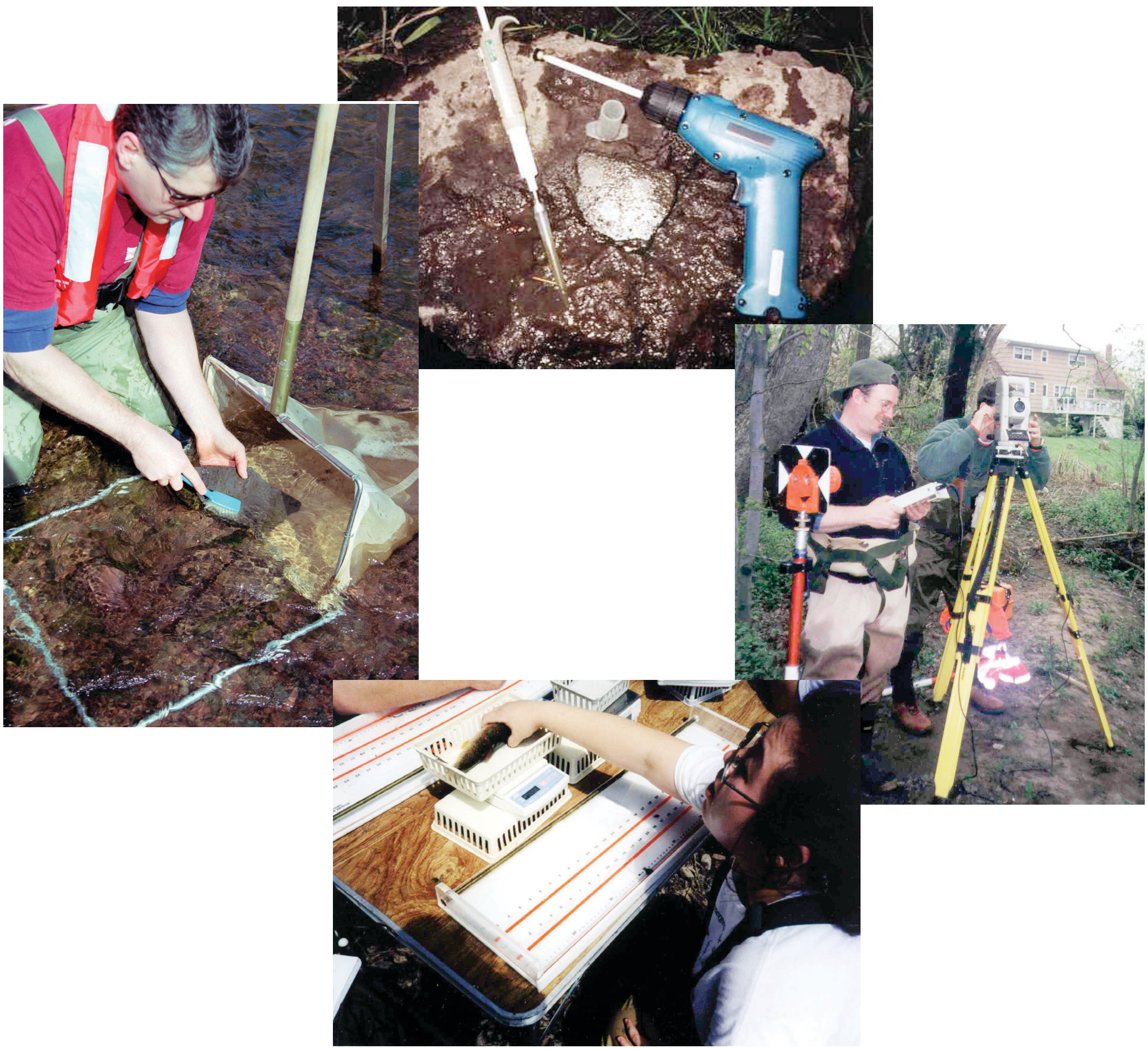
# RELATION OF ENVIRONMENTAL CHARACTERISTICS TO THE COMPOSITION OF AQUATIC ASSEMBLAGES ALONG A GRADIENT OF URBAN LAND USE IN NEW JERSEY, 1996-98

## Water-Resources Investigations Report 02-4069



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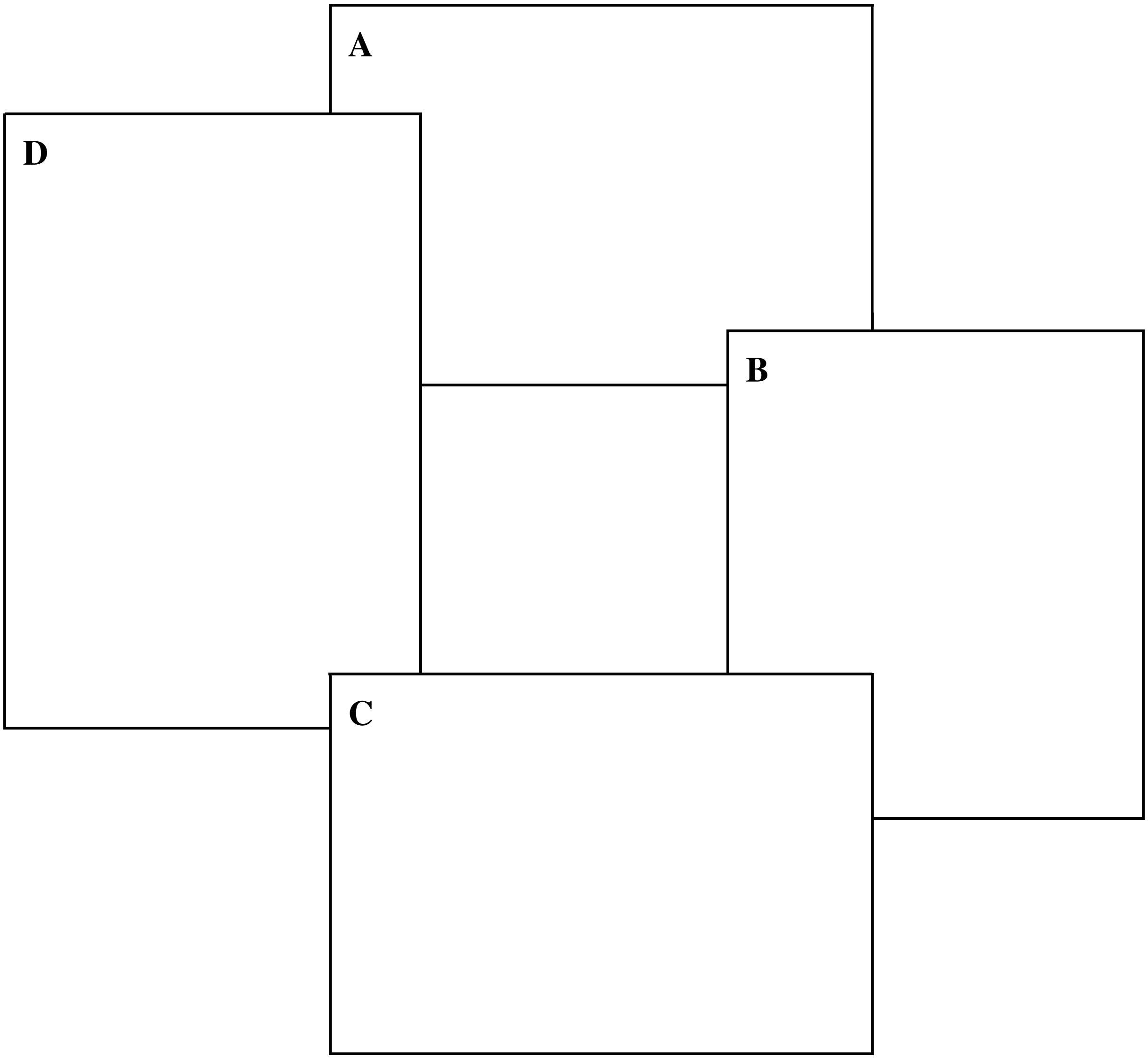


**A.** Algae sampling equipment (pipettor, microalgal periphyton brush attached to end of hand drill, and a modified-syringe sampling device)

**B.** Surveying channel cross sections (Photograph by Ming Chang, U.S. Geological Survey)

**C.** Determining length and weight of a white sucker

**D.** Collecting aquatic macroinvertebrates (Photograph by Denis Sun, U.S. Geological Survey)

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# RELATION OF ENVIRONMENTAL CHARACTERISTICS TO THE COMPOSITION OF AQUATIC ASSEMBLAGES ALONG A GRADIENT OF URBAN LAND USE IN NEW JERSEY, 1996-98

***By Jonathan G. Kennen and Mark A. Ayers***

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**U.S. GEOLOGICAL SURVEY**

**Water-Resources Investigations Report 02-4069**

## National Water-Quality Assessment Program

**West Trenton, New Jersey**

**2002**

USGS
science for a changing world

**U.S. DEPARTMENT OF THE INTERIOR**

**Gale A. Norton, *Secretary***

**U.S. GEOLOGICAL SURVEY**

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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. 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. 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. 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. 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.

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

Robert M. Hirsch
Associate Director for Water

## CONVERSION FACTORS, VERTICAL DATUM, AND ABBREVIATIONS

| Multiply | By | To obtain |
|---|---|---|
| | ***Length*** | |
| micrometer (mm) | 0.00003937 | inch |
| millimeter (mm) | 0.03937 | inch |
| centimeter (cm) | 0.3937 | inch |
| meter (m) | 3.281 | foot |
| kilometer (km) | 0.6214 | mile |
| | ***Area*** | |
| square centimeter ($cm^2$) | 0.155 | square inch |
| square meter ($m^2$) | 10.76 | square foot |
| square kilometer ($km^2$) | 0.3861 | square mile |
| | ***Volume*** | |
| liter (l) | 0.2642 | gallon |
| milliliter (ml) | 0.0338 | ounce, fluid |
| dram (dr) | 0.125 | ounce, fluid |
| | ***Flow*** | |
| centimeter per second (cm/s) | 0.0328 | foot per second |
| | ***Mass*** | |
| gram (g) | 0.03527 | ounce, avoirdupois |
| | ***Pressure*** | |
| kilopascal (kPa) | 0.1450 | pound-force per square inch |

Temperature Conversion
Degree Celsius (°C) may be converted to degree Fahrenheit (°F) by using the following equation:
°F = 9/5(°C) + 32

Sea level: In this report "sea level" refers to the National Geodetic Vertical Datum of 1929—a geodetic datum derived from a general adjustment of the first-order level nets of the United States and Canada, formerly called Sea Level Datum of 1929.

## Other abbreviations used in this report:

| Abbreviation | Description |
|---|---|
| ANS | Academy of Natural Sciences, Philadelphia, Pennsylvania |
| DCA | Detrended correspondence analysis |
| DOC | Dissolved organic carbon |
| GIS | Geographic Information System |
| ISC | Impervious surface cover |
| LINJ | Long Island-New Jersey Coastal Drainages Study Unit |
| LU/C | Land use and land cover |
| MLR | Multiple linear regression |
| NAWQA | National Water-Quality Assessment Program |
| NJDEP | New Jersey Department of Environmental Protection |
| NJSOP | New Jersey Office of State Planning |
| NJIS | New Jersey impairment score |
| NWQL | National Water-Quality Laboratory |
| PCA | Principal components analysis |
| pCCA | Partial canonical correspondence analysis |
| RTH | Richest targeted habitat |
| SOC | Suspended organic carbon |
| TDI | Tolerant diatom index |
| TWINSPAN | Two-Way INdicator SPecies ANalysis |
| ULUG | Urban land-use gradient |
| USEPA | U.S. Environmental Protection Agency |
| USGS | U.S. Geological Survey |
| VIFs | Variance inflation factors |
| VOCs | Volatile organic compounds |
| > | greater than |
| < | less than |

# GLOSSARY

**Algae—**Chlorophyll-bearing nonvascular, primarily aquatic species that have no true roots, stems, or leaves; most algae are microscopic, but some species can be as large as vascular plants.
**Anthropogenic—**Occurring because of, or influenced by, human activity.
**Aquatic Invertebrates—**Insects, worms, crayfish, snails, clams, and other organisms without a backbone that inhabit lakes, streams, rivers, or oceans.
**Base flow—**Sustained, low flow in a stream; ground-water discharge is the source of base flow in most streams.
**Benchmark site—**See reference site.
**Benthic—**Organisms living on or burrowing into aquatic substrates.
**Drainage basin—**The portion of the surface of the Earth that contributes water to a stream through overland runoff, including tributaries and impoundments.
**Embeddedness—**The degree to which gravel-sized and larger substrate particles are surrounded or enclosed by finer sized particles.
**Ichthyofauna—**Refers to fish.
**Impervious surfaces—** Hard, non-porous surfaces in a watershed such as asphalt, concrete, buildings, and other compacted or impermeable surfaces that prevent precipitation from infiltrating into the ground, thus increasing surface runoff.
**Index of Biotic Integrity (IBI)—**An aggregated number, or index, based on several attributes or metrics of a fish community that provides an assessment of the health of aquatic systems.
**Integrator site—**Stream-sampling site located at an outlet of a drainage basin that contains multiple environmental settings. Most integrator sites are on major streams with relatively large drainage areas.
**Interstitial space—**The space between substrate particles (substrate porosity) that provides a refuge for aquatic organisms.
**Intolerant organisms—**Organisms that are not adaptable to human alterations to the environment and commonly decline in number where human alterations occur. See also tolerant species.
**Motile—**Refers to organisms that have the ability to move, often rapidly and spontaneously.
**Periphyton—**Algae attached to an aquatic substrate (also known as benthic algae).
**Reference site—**A sampling site selected for its relatively undisturbed conditions.
**Relative abundance—**The number of organisms of a particular kind present in a sample relative to the total number of organisms in the sample.
**Retrospective analysis—**The review and analysis of available water-quality and ecological data to provide a historical perspective of water quality and biological integrity, to assess strengths and weaknesses of available information, and to evaluate implications for water-quality management and study design.
**Richest targeted habitat (RTH)—**A targeted habitat (usually a riffle or woody snag where the taxonomically richest algal or invertebrate community theoretically is located) identified in a reach from which discrete samples of algae or invertebrates are collected and later combined to form a composite sample. The composited sample is referred to as an "RTH sample."
**Riparian buffer—**A vegetated protective area adjacent to rivers and streams with a high density, diversity, and productivity of plant and animal species. It serves as a barrier against runoff and as a habitat corridor for many terrestrial animals.
**Runoff—**Rainwater or snowmelt that is transported to streams by overland flow, tile drains, or ground water.
**Sinuosity—**The ratio of the channel length between two points on a channel to the straight-line distance between the same two points; a measure of meandering.
**Tolerant species—**Those species that are adaptable to (tolerant of) human alterations to the environment and commonly increase in number when human alterations occur.
**Urban gradient study—**A study designed to measure physical, chemical, and biological responses along

gradients of urban land-use intensity and identify the factors most responsible for controlling water-quality conditions.

**Volatile organic compounds (VOCs)—**Organic chemicals that have a high vapor pressure relative to their water solubility. VOCs include components of gasoline, fuel oils, and lubricants, as well as organic solvents, fumigants, some inert ingredients in pesticides, and some by-products of chlorine disinfection.

**Watershed—**See Drainage basin.

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***By Jonathan G. Kennen and Mark A. Ayers***

## ABSTRACT

Community data from 36 watersheds were used to evaluate the response of fish, invertebrate, and algal assemblages in New Jersey streams to environmental characteristics along a gradient of urban land use that ranged from 3 to 96 percent. Aquatic assemblages were sampled at 36 sites during 1996-98, and more than 400 environmental attributes at multiple spatial scales were summarized. Data matrices were reduced to 43, 170, and 103 species of fish, invertebrates, and algae, respectively, by means of a predetermined joint frequency and relative abundance approach. White sucker (Catostomus commersoni) and Tessellated darter (Etheostoma olmstedi) were the most abundant fishes, accounting for more than 20 and 17 percent, respectively, of the mean abundance. Net-spinning caddisflies (Hydropsychidae) were the most commonly occurring benthic invertebrates and were found at all but one of the 36 sampling sites. Blue-green (for example, Calothrix sp. and Oscillatoria sp.) and green (for example, Protoderma viride) algae were the most widely distributed algae; however, more than 81 percent of the algal taxa collected were diatoms.

Principal-component and correlation analyses were used to reduce the dimensionality of the environmental data. Multiple linear regression analysis of extracted ordination axes then was used to develop models that expressed effects of increasing urban land use on the structure of aquatic assemblages. Significant environmental variables identified by using multiple linear regression analysis then were included in a direct gradient analysis. Partial canonical correspondence analysis of relativized abundance data was used to restrict further the effects of residual natural variability, and to identify relations among the environmental variables and the structure of fish, invertebrate, and algal assemblages along an urban land-use gradient. Results of this approach, combined with the results of the multiple linear regression analyses, were used to identify human population density (311-37,594 persons/km$^2$), amount and type of impervious surface cover (0.12-1,350 km$^2$), nutrient concentrations (for example, 0.01-0.29 mg/L of phosphorus), hydrologic instability (for example, 100-8,955 ft$^3$/s for 2-year peak flow), the amount of forest and wetlands in a basin (0.01-6.25 km$^2$), and substrate quality (0-87 percent cobble substrate) as variables that are highly correlated with aquatic-assemblage structure. Species distributions in ordination space clearly indicate that tolerant species are more abundant in the streams impaired by urbanization and sensitive taxa are more closely associated with the least impaired basins. The distinct differences in aquatic assemblages along the urban land-use gradient demonstrate the deleterious effects of urbanization on assemblage structure and indicate that conserving landscape attributes that mitigate anthropogenic influences (for example, stormwater-management practices emphasizing infiltration and preservation of existing forests, wetlands, and riparian corridors) will help to maintain the relative abundance of sensitive taxa. Complementary multiple linear regression models indicate that aquatic community indices were correlated with many of the anthropogenic factors that were found to be significant along the urban land-use gradient. These indices appear to be effective in differentiating the moderately and severely impaired streams from the minimally impaired streams. Evaluation of disturbance thresholds for aquatic assemblages indicates that

moderate to severe impairment is detectable in New Jersey streams when impervious surface cover in the drainage basin reaches approximately 18 percent.

## INTRODUCTION

Human-induced alterations of the landscape associated with population growth and urban development have been consistently linked to water-quality deterioration and contribute to the degradation of streams and rivers throughout the United States (for example, Klein, 1979; Garie and McIntosh, 1986; Scott and others, 1986; Booth and Jackson, 1997; Jones and Clark, 1987; Clements and others, 1988; Wear and others, 1998; Kennen, 1999; Ayers and others, 2000) and other regions of the world (for example, Prowse, 1987; Gilbert, 1989; Wright and others, 1995). This degradation is exemplified by the replacement of climax habitat and undeveloped lands with residential and commercial developments, and an associated general decline of biological diversity in aquatic and terrestrial habitats (for example, Morgan, 1987; Moyle and Williams, 1990; Allan and Flecker, 1993; Angermeier, 1994).

Streams in urban and urbanizing areas are affected by many anthropogenic factors that directly or indirectly modify stream geomorphology, chemistry, and aquatic-assemblage structure. Urban development alters flow regimes by straightening, deepening, widening, and diverting natural channels (Leopold, 1968; Klein, 1979) and by increasing the amount of impervious surface area in a watershed. Changes in hydrology, including rapid changes in water levels and peak flows, have been found to cause measurable physical and biological changes in streams (Riley, 1998).

Equally important is the relation between impervious surface cover and nonpoint-source runoff, which has been shown to adversely affect stream-water quality and aquatic biota (Garie and McIntosh, 1986; Schueler, 1994). Although impervious surfaces themselves do not generate contamination, they can induce dramatic hydrologic changes in a watershed that promote many of the physical, chemical, and biological changes that adversely affect urban streams (May and others, 1997). As impervious cover increases, surface runoff increases in volume and velocity while infiltration and ground-water recharge rates decrease. Moreover, flooding can result from increased runoff volume moving swiftly through artificial water-conveyance devices such as pipes, gutters, and straightened channels (Arnold and Gibbons, 1996). Floods and other fluvial processes can be part of a natural disturbance regime that helps shape stream communities and increase biodiversity (Reice, 1994; Ward and Stanford; 1983, Resh and others, 1988); however, urban development dramatically alters this regime by increasing flood frequency and magnitude, and by altering the response of stream channels to floods as a result of changes in stream structure and morphology (Schroeder and Savonen, 1997). Consequently, aquatic assemblages in urban streams are stressed continually and rarely reach stable population levels. Additionally, reduction in base flow resulting from changes in surface- and ground-water-use practices greatly affects the suitability of a stream for many types of aquatic fauna (Klein, 1979). Fluctuations in base flow and repeated exposure of areas along stream margins have been shown to result in slow recovery and decreased production of macroinvertebrate communities (Perry and Perry, 1986; Blinn and others, 1995). In addition, modification of the hydrologic regime can alter the composition, structure, and function of aquatic ecosystems through their effects on other environmental and habitat characteristics, including water temperature, oxygen content, water chemistry, and substrate size (Richter and others, 1996; Ward and Stanford, 1989). Changes in hydrology also are accompanied by well-documented changes in water quality; concentrations of dissolved substances, including nutrients and suspended sediments, which may sorb metal and organic contaminants, increase (for example, Prowse, 1987; Marsalek, 1991).

Increases in point and nonpoint sources of chemicals and chemical mixtures, nutrients, suspended sediment, wastewater effluent (municipal and industrial), trace elements, and pesticides in urban areas have been shown to affect water quality and aquatic assemblage structure (Porcella and Sorensen, 1980; Culp and others, 1986; Clements and others, 1988; Newcombe and MacDonald, 1991; Wright and others, 1995; Kennen, 1999;

Rabeni and Smale, 1995). Most of these effects are compounded greatly by increases in impervious surface cover that increase the velocity of stormwater, increase the washoff of toxic substances and particulates, and provide a continuous pathway along which many of these constituents can be transported (Hoffman and others, 2000).

Reduction or elimination of riparian vegetation, streambank alterations, and construction activities can reduce shade, modify streambank stability, and affect soil profiles. Stream corridors that are left relatively undisturbed typically contain greater vegetative cover and soils with higher infiltration capacity, which helps to mitigate the deleterious effects of other anthropogenic factors on stream health. Richards and others (1997) found that streams draining basins with well-drained soils and high infiltration capacity tend to be stable and dominated by ground-water discharge, whereas streams draining basins with poorly drained soils commonly are “flashy” and are dominated by surface runoff. Stream riparian buffers consistently have been shown to mitigate the effects of erosion and stabilize streambanks, ultimately reducing the amount of sediment entering the stream channel (Schlosser and Karr, 1981). In addition, the value of wetlands in protecting surface-water quality by providing a natural filter has been demonstrated extensively; riparian wetlands also support high rates of denitrification, thus reducing high loads of nitrate in surface water (Johnston, 1991; Osborne and Kovacic, 1993; Lowrance, 1998).

The effect of the conversion of various land uses to urbanized environments on aquatic assemblages is an emerging issue of local and global importance. Although many individual landscape and environmental factors have been shown to affect aquatic assemblages in urban areas, few investigators have attempted to examine these factors over the full gradient of urbanization (Blair, 1996). Little is known about the ultimate consequences of urbanization on aquatic ecosystems because, historically, most studies have concentrated on pristine environments (Cairns, 1988; Blair, 1996), have evaluated only a selected few environmental factors, have assessed only one component of the aquatic community, or have failed to eliminate natural gradients from the assessment. Therefore, in 1996 the U.S. Geological Survey (USGS), as part of its National Water-Quality Assessment (NAWQA) Program, initiated a 3-year integrated study of 36 watersheds in northern New Jersey (fig. 1) to determine biological responses of aquatic assemblages to environmental degradation along an urban land-use gradient (ULUG).

## Purpose and Scope

This report evaluates the most sensitive environmental and biological features in impaired basins, develops predictive models of stream degradation for urban watersheds, and identifies the important environmental factors that affect the distribution of fish, invertebrate, and algal assemblages along an urban land-use gradient. It includes a description of field, analytical, and statistical methods, lists of the fish, invertebrate, and algal assemblages sampled, and evaluates the relation of aquatic-assemblage structure to a suite of environmental parameters (that is, hydrology, water quality, physical habitat, and surrounding landscape features). Steps taken during the stratification and analytical process to limit variability associated with natural landscape factors such as climate, geology, longitude, latitude, elevation, and physiography are described. The relevance of these findings to urban systems and their implications for the protection of existing aquatic resources also are discussed.

## Study Area

The study area covers approximately 21,000 $km^2$ (8,100 $mi^2$) in New Jersey and southern New York (figs. 1 and 2). This area has a population of about 8.4 million and includes some of the most densely populated metropolitan areas in the United States (U.S. Bureau of Census, 2001, USA Statistics in Brief–Census, 2000 Resident Population of States and DC, accessed April 5, 2001, on the World Wide Web at URL http://www.census.gov/statab/www/part6.htrr). For more than 2 centuries, a prosperous commercial and industrial economy has centered on the seaports of New York City, N.Y., and Philadelphia, Pa. As of the early 1970’s, 22 percent of the study area was developed for urban residential, commercial, or industrial use

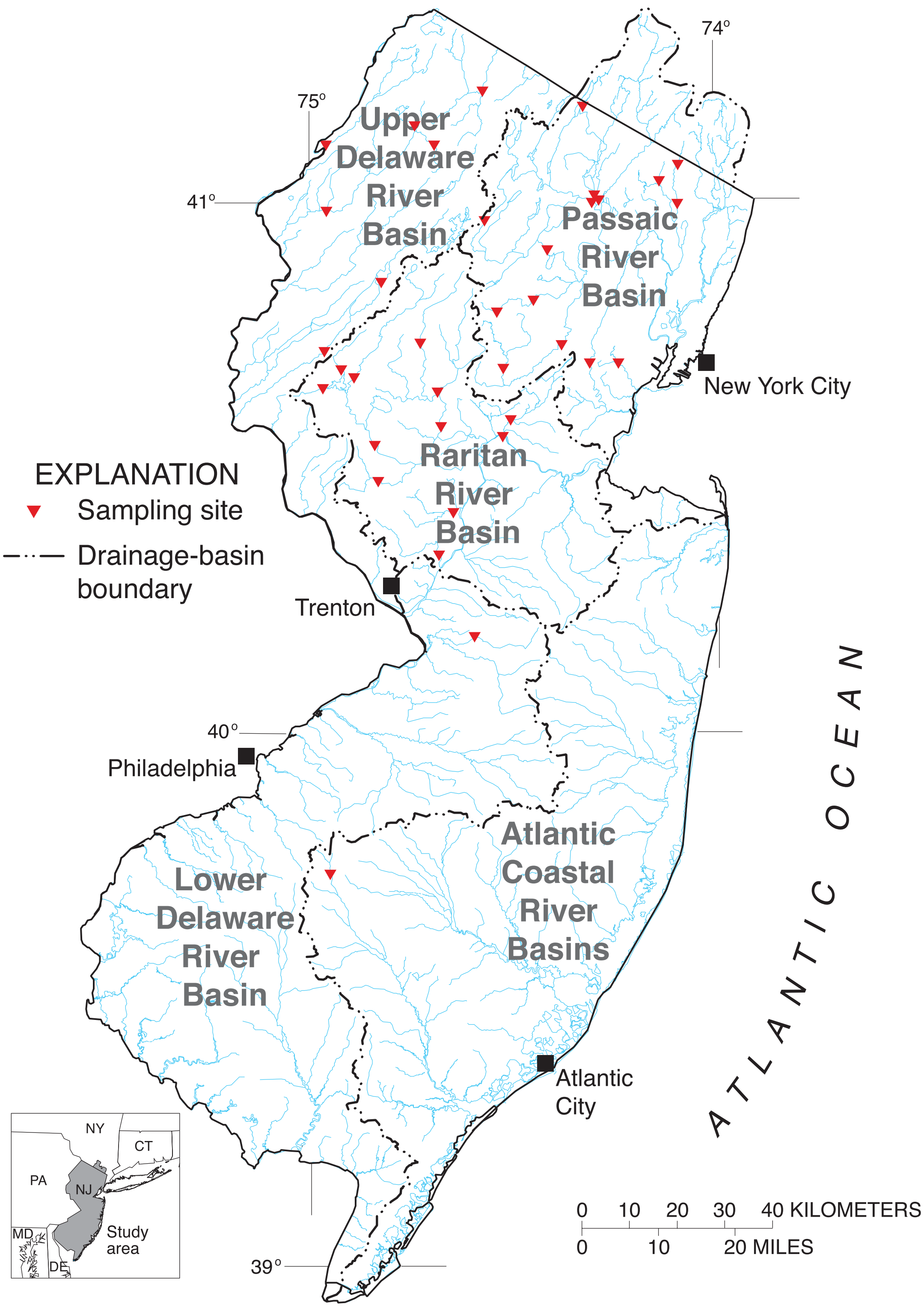


**Figure 1.** Location of fish, invertebrate, and algal sampling sites in New Jersey.

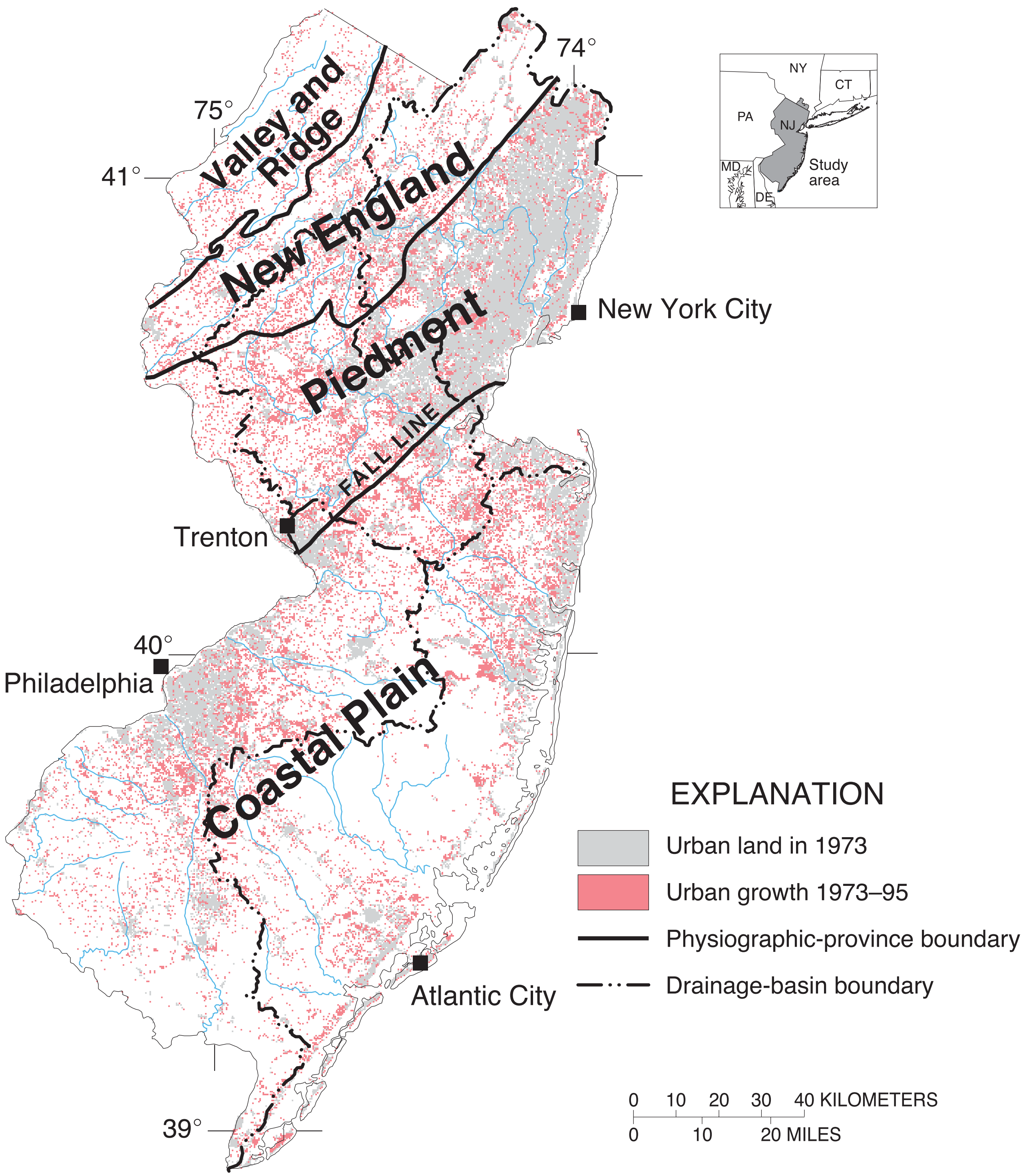


**Figure 2.** Change in extent of urban land from 1973 to 1995 and location of physiographic provinces in New Jersey and southern New York. (N.J. Department of Environmental Protection, 1996, modified with U.S. Geological Survey, 1986)

(fig. 2). By the mid-1990's, urban land use had increased to 33 percent, with a corresponding 11-percent decrease in forested and agricultural land. Overall population growth during the same period was about 8 percent. The cost of land and pressure for development have reduced greatly the amount of agricultural land, which accounted for nearly 60 percent of the land use in the study area in 1900, but only 14 percent by the mid-1990's.

New Jersey is composed of four physiographic provinces. From northwest to southeast, they are the Valley and Ridge, New England, Piedmont, and Coastal Plain provinces (fig. 2). The area occupied by each province increases to the southeast. The Valley and Ridge province is characterized by a series of parallel ridges and valleys trending northeast-southwest (O'Brien, 1997). The steep, mountainous topography of this region, commonly reaching elevations of greater than 480 m, has deterred the intensive urbanization that has occurred in most other regions of New Jersey. To the southeast, the New England province consists of broad, flat-topped highlands and long, narrow valleys that range in elevation from 150 to 460 m. Southeast of the New England province is the Piedmont province, which consists of northwestward-dipping sedimentary rocks that form broad, gently sloping lowlands and rolling valleys where elevations typically reach only 120 m. About 55 percent (11,550 $km^2$) of the study area is in the Coastal Plain (fig. 2) and is characterized by flat to gently rolling topography and unconsolidated sedimentary deposits (Wolfe, 1977).

Streamflow in the study area is dominated by surface runoff. Most of northeastern New Jersey and the corridor between New York City and Philadelphia consist of heavily urbanized, and thus relatively impervious areas, that yield runoff rapidly. Surface-water and ground-water withdrawals have reduced base flow in streams in some areas. Water-supply systems in the study area are highly connected, and transfer of water across drainage divides and among basins is common. Annual precipitation and runoff during the years of this study were slightly greater than the 40-year mean (Ayers and others, 2000). Although hydrology and surficial geology differ across the study area, analyses of available chemical and biological data indicate that land use is the primary differentiating factor that accounts for much of the variability in water quality and aquatic communities (Stackelberg, 1997; O'Brien, 1997; Kennen, 1999).

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## APPROACH

Landscape and local-scale (proximal) environmental variables relating to fish, invertebrate, and algal communities were measured at 36 sampling sites (fig. 1) in northern New Jersey. The drainage areas contributing to streamflow at each site ranged from 3 to 96 percent urban land use (fig. 3). A 100- to 300-m reach representative of the stream segment at the base of each basin was selected; if possible, reaches upstream from local

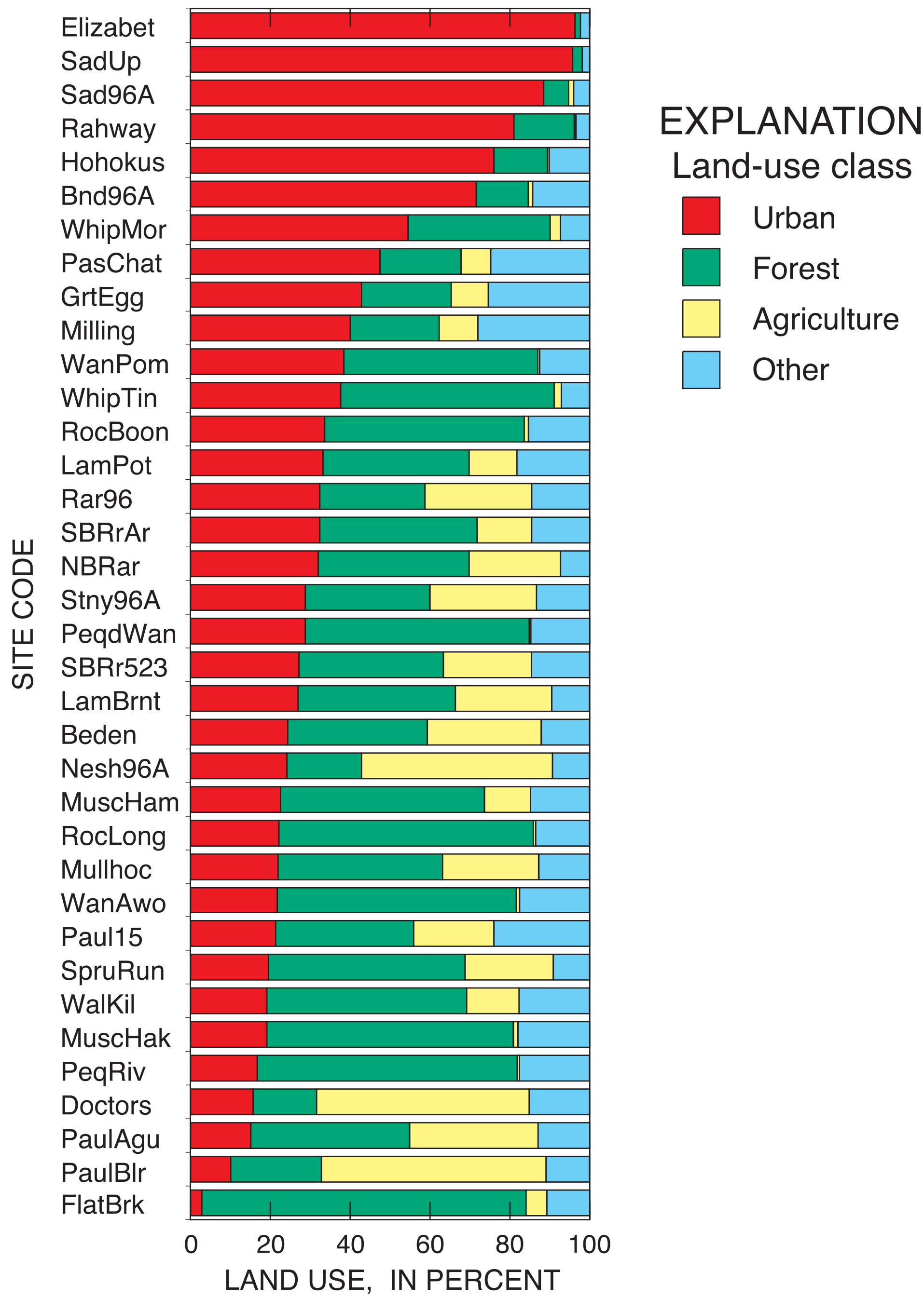


**Figure 3.** Land use in the drainage basins of the 36 streams sampled in New Jersey, 1995-97. (Data from N.J. Department of Environmental Protection, 1996)

effects such as wastewater outfalls, bridges, storm sewers, and major tributaries were selected (Meador and others, 1993a; Waite and Carpenter, 2000). One reach at each of five sites was sampled for 3 consecutive years (1996-98) to assess temporal variability. In addition, two additional stream reaches were sampled at each of two sites in 1997 to assess spatial variability (Meador and others, 1993a). For this study, a total of 48 fish, invertebrate, and algal samples were collected from the 36 streams.

## Design of Land-Use Gradient

The land-use-gradient design used in this assessment was based on an environmental framework that minimizes variability associated with natural landscape factors—factors that may confound understanding of chemical and biological responses along the ULUG (Cuffney and others, 1997; Maret and others, 1997; McMahon and Cuffney, 2001). Various steps were taken to reduce variability in the data attributed to natural factors such as climate, geology, longitude, latitude, elevation, and physiography. First, a group of prospective sites within the study area that fall within the uppermost level of the physiographic stratification (synonymous in New Jersey with level III ecoregions (Omernik, 1995)) (fig. 2) was screened. This type of stratification, which has been applied widely in other water-quality investigations throughout the United States (Hughes and others, 1994), was used to create relatively homogeneous biotic and abiotic sampling areas. Second, with the exception of one large river (integrator) site (2,082 $km^2$) (table 1), basin size was restricted to a range of 21 to 496 $km^2$ (primarily third- and fourth-order streams) to maintain consistency with other studies conducted as part of the NAWQA Program (Gilliom and others, 1995). Third, available data were used to aid in the study design and in site selection. For example, an ordination of benchmark (“minimally impaired”) aquatic macroinvertebrate communities (fig. 8; New Jersey Department of Environmental Protection, 1994b) from basins throughout New Jersey was used to assess similarity in biological integrity among physiographic strata. This assessment indicated a high degree of similarity among northern streams and a distinct separation between northern and southern streams. Therefore, sampling was concentrated in northern New Jersey and reduced much of the natural biological variability that would have resulted from inclusion of southern New Jersey streams. In addition, a historical analysis was conducted to assess whether specific basin characteristics were important predictors of aquatic-community impairment and whether the proximity of these characteristics to a sampling site was important (Kennen, 1999; Chang and others, 2000). Finally, partial constrained ordination was used to remove any remaining natural variability in these data (for example, variability associated with differences in latitude, longitude, altitude, drainage area, and year sampled). The latter technique is explained in detail in the Analytical Methods section.

## Aggregation of Anthropogenic Factors Used in Assessment

More than 400 basin, reach, and instream environmental factors were summarized to describe environmental characteristics at each sampling site. These characteristics include information on nutrients, pesticides, volatile organic compounds, and major ions in stream water; trace elements and organochlorine compounds in bed sediment; geomorphic, hydrologic, and instream and riparian habitat characteristics; field parameters such as water temperature, alkalinity, pH, dissolved-oxygen concentration, and specific conductance; and fish, invertebrate, and algal assemblages. Other factors assessed were directly related to the level of anthropogenic disturbance, urban land, impervious surface cover, and socioeconomic characteristics (for example, human population density and housing value). Results of previous analyses indicate that aquatic-community impairment is related to the proximity of anthropogenic factors (for example, urban land use, high population density, and point sources (Kennen, 1999)). To assess the relative importance of distance from a source to a sampling site, a distance-weighted negative exponential decay model was applied to many of the explanatory variables to improve the accuracy of the representation of hydrologic processes along land and stream pathways. In addition, newly available digital data on landscape characteristics such as topography, roads, and soils were used to generate summary statistics for each basin.

**Table 1.** Watershed characteristics of sampled streams, New Jersey.

[Stations are listed in order of decreasing percentage of urban land; multiple-reach and multiple-year stations are in bold; ds, downstream; us, upstream; Rte, Route; USGS, U.S. Geological Survey; m, meters; $km^2$, square kilometers; ASL, above sea level]

| Station name | Station code | USGS station number | Drainage area ($km^2$) | Elevation ASL (m) | Land use (in percent) Urban | Forest | Agriculture | Water | Wetland | Barren |
|---|---|---|---|---|---|---|---|---|---|---|
| Elizabeth River at Hillside, N.J. | Elizabet | 01393400 | 36 | 9 | 96.2 | 1.7 | 0.0 | 0.2 | 1.8 | 0.2 |
| Saddle River at Upper Saddle River, N.J. | SadUp | 01390450 | 28 | 57 | 95.6 | 2.7 | .0 | .0 | .7 | .1 |
| **Saddle River at Ridgewood, N.J.** | Sad96A | 01390500 | 57 | 22 | 88.4 | 6.4 | .8 | .4 | 3.9 | .1 |
| Rahway River near Springfield, N.J. | Rahway | 01394500 | 66 | 20 | 80.9 | 15.3 | .2 | .9 | 1.4 | 1.4 |
| Hohokus Brook ds W Crescent Ave at Allendale, N.J. | Hohokus | 01390815 | 25 | 82 | 75.8 | 13.7 | .3 | 1.8 | 7.9 | .5 |
| **Bound Brook at Middlesex, N.J**. | Bnd96 | 01403900 | 125 | 8 | 71.5 | 13.4 | .5 | .5 | 12.7 | 1.6 |
| Whippany River at Morristown, N.J. | WhipMor | 01381500 | 75 | 79 | 54.4 | 35.7 | 2.7 | 1.0 | 5.8 | .3 |
| Passaic River near Chatham, N.J. | PasChat | 01379500 | 259 | 59 | 47.4 | 20.5 | 6.4 | .8 | 23.3 | 1.6 |
| Great Egg Harbor River near Sicklerville, N.J. | GrtEgg | 01410784 | 39 | 35 | 42.8 | 22.6 | 8.5 | .7 | 23.6 | 1.8 |
| Passaic River near Millington, N.J. | Milling | 01379000 | 142 | 66 | 39.9 | 22.4 | 8.6 | 1.0 | 27.0 | 1.1 |
| Wanaque River at Pompton Lakes, N.J. | WanPom | 01387041 | 277 | 56 | 38.3 | 48.8 | .3 | 6.7 | 5.0 | .9 |
| Whippany River ds Tingley Road Near Brookside, N.J. | WhipTin | 01381295 | 21 | 105 | 37.5 | 53.6 | 1.6 | 1.4 | 5.8 | .1 |
| Rockaway River above reservoir at Boonton, N.J. | RocBoon | 01380500 | 301 | 111 | 33.5 | 50.2 | .8 | 4.5 | 9.8 | 1.2 |
| Lamington River near Pottersville, N.J. | LamPot | 01399500 | 85 | 86 | 33.2 | 36.7 | 12.0 | 1.8 | 14.7 | 1.7 |
| **Raritan River at Queens Bridge at Bound Brook, N.J.** | Rar96 | 01403300 | 2,082 | 5 | 32.3 | 26.6 | 26.6 | 1.7 | 11.2 | 1.6 |
| South Branch Raritan River at Arch St at High Bridge, N.J. | SBRrAr | 01396535 | 179 | 73 | 32.2 | 39.6 | 13.5 | 1.5 | 12.4 | .7 |
| North Branch Raritan River near Raritan, N.J. | NBRar | 01400000 | 492 | 15 | 31.9 | 38.0 | 21.8 | .8 | 6.6 | .9 |
| **Stony Brook at Princeton, N.J.** | Stny96A | 01401000 | 117 | 19 | 28.8 | 31.2 | 25.7 | .9 | 12.7 | .7 |
| Pequannock River at Pompton Lakes, N.J. | PeqdWan | 01387042 | 496 | 55 | 28.7 | 56.2 | .3 | 6.3 | 7.9 | .8 |
| South Branch Raritan River us Rte 523 at Darts Mills, N.J. | SBRr523 | 01397295 | 436 | 33 | 27.1 | 36.4 | 22.1 | 4.2 | 9.2 | 1.0 |
| Lamington River at Burnt Mills, N.J. | LamBrnt | 01399780 | 259 | 23 | 26.9 | 39.5 | 24.2 | .9 | 7.5 | .9 |
| Beden Brook near Rocky Hill, N.J. | Beden | 01401600 | 70 | 12 | 24.2 | 35.3 | 27.8 | .3 | 11.6 | .8 |
| **Neshanic River at Reaville, N.J.** | Nesh96A | 01398000 | 67 | 33 | 24.2 | 18.8 | 47.4 | .2 | 7.7 | 1.8 |
| Musconetcong River at Hampton, N.J. | MuscHam | 01456600 | 317 | 96 | 22.5 | 51.2 | 11.6 | 5.5 | 7.9 | 1.4 |
| Rockaway River at Longwood Valley, N.J. | RocLong | 01379680 | 57 | 210 | 22.1 | 63.9 | .3 | 3.0 | 9.7 | 1.0 |
| Mulhockaway Creek at Van Syckel, N.J. | Mullhoc | 01396660 | 31 | 85 | 21.8 | 41.5 | 23.5 | .3 | 10.8 | 2.1 |
| Wanaque River near Awosting, N.J. | WanAwo | 01383505 | 73 | 178 | 21.8 | 59.9 | .6 | 10.8 | 6.0 | .9 |
| Paulins Kill us Rte 15 at Lafayette, N.J. | Paul15 | 01443290 | 67 | 164 | 21.3 | 34.8 | 19.8 | 2.4 | 17.2 | 4.6 |
| Spruce Run near Glen Gardner, N.J. | SpruRun | 01396588 | 39 | 87 | 19.5 | 49.4 | 21.4 | .4 | 8.6 | .7 |
| Wallkill River near Sussex, N.J. | WalKil | 01367770 | 158 | 119 | 19.1 | 50.2 | 13.2 | 4.5 | 11.7 | 1.4 |
| Musconetcong River ds East Ave at Hackettstown, N.J. | MuscHak | 01456070 | 197 | 158 | 19.0 | 62.0 | 1.0 | 9.0 | 3.0 | 2.0 |
| Pequannock River at Riverdale, N.J. | PeqRiv | 01382800 | 218 | 57 | 16.7 | 65.4 | .3 | 5.7 | 11.4 | 0.6 |
| Doctors Creek at Allentown, N.J. | Doctors | 01464515 | 44 | 16 | 15.6 | 16.1 | 53.0 | 1.6 | 12.5 | 1.2 |
| Paulins Kill at Augusta, N.J. | PaulAgu | 01443310 | 104 | 152 | 15.0 | 40.0 | 32.0 | 2.0 | 6.0 | 2.0 |
| Paulins Kill ds Blair Creek at Blairstown, N.J. | PaulBlr | 01443515 | 366 | 102 | 10.0 | 23.0 | 56.0 | 1.0 | 8.0 | .0 |
| Flat Brook at Flatbrookville, N.J. | FlatBrk | 01440010 | 168 | 103 | 2.7 | 81.4 | 5.0 | 1.6 | 9.3 | .1 |

# DATA COLLECTION

## Stream Characteristics

At each sampling site, habitat was characterized at multiple spatial scales (reach, segment, and basin). This approach combines geographic information system (GIS) assessment, mapping, and field collection methods. Instream and riparian features as well as the physiography, hydrography, geology, and geomorphology of the stream corridor were characterized.

### Habitat

The length of each reach and the area and distribution of habitat types were measured at each sampling site. Measures of instream habitat were made along three to six transects perpendicular to streamflow and associated with specific geomorphic units (pools, riffles, runs) spaced throughout a stream reach (fig. 4). Along each transect, channel features such as wetted width, water-column velocity, elevation, discharge, substrate size, dominant substrate, substrate embeddedness, water depth, and instream cover were measured. Streambank angle, height, and stability as well as canopy opening and riparian width and stability were recorded. Embeddedness was estimated visually by means of a percentage class (that is, 0, 25, 50, 75, or 100 percent (Meador and others, 1993b)). Velocity (in centimeters per second (cm/s)) was determined (fig. 4) at a depth equivalent to 0.6 times the measured stream depth by using a velocity meter. Instantaneous stream discharge (in cubic feet per second ($ft^3/s$)) was either measured at the time of sample collection or acquired from USGS streamflow gaging-station records. Through use of a clinometer, canopy opening was derived from angular measurements made from mid-channel to the tallest objects to the left and right of each transect. The left and right angles were summed and subtracted from $180^0$ to acquire the canopy opening. The composition of the streambed substrate was assessed using the Wolman Pebble Count methods (Wolman, 1954), placed in Wentworth size classes that ranged from sand (<2 mm) to boulders (>2,048 mm) (Harrelson and others, 1994), and then converted to a percentage. Instream cover was estimated as a percentage of the total area along a transect (approximately 2 m to either side of the transect) available to fish as complex cover. Average values of instream measurements were calculated.

Segment-level assessment included such factors as sinuosity, slope, segment length, and channel gradient. Sinuosity was calculated as the ratio of river distance to the straight-line distance between the upstream and downstream ends of the stream segment. Channel gradient (percent slope) was determined as the change in elevation between the upstream and downstream ends of a segment of stream containing the sampling site divided by the river distance and multiplied by 100.

**Figure 4.** Water velocity being measured along one of six transects at Saddle River at Ridgewood, N.J. (01390500).

Basin-level assessment included level I Anderson classifications (Anderson and others, 1976) of land-use and land-cover data (percentage of urban land, agricultural land, forest, wetland, water, and barren land), impervious-surface-area percentage, point-source location and flow, percent forest in riparian buffer areas (15, 30, 100 and 300 m to either side of the stream channel), soil permeability, stream order, drainage-basin area, total stream length, human population density, housing density, and housing value. Stream order was determined from 1:24,000-scale topographic maps and is based on a measure of the position of a stream in a hierarchy of tributaries (Strahler, 1957). Values of stream sinuosity, gradient, and elevation were derived from 1:24,000-scale USGS topographic maps. Most other basin-level measurements were determined by using 1:24,000-scale digital coverages.

## Ichthyofauna

Fish samples following the protocols outlined by Meador and others (1993a) were collected from late July to early September 1996-98 during stable, low-flow periods. Stream reaches consisting of at least one meander wavelength and containing a minimum of one riffle-run-pool sequence were targeted. Pool-riffle sequences are approximately 5 to 7 bankfull widths in length in natural channels (Leopold and others, 1995); in chanelized streams or areas of low gradient where the target habitat was not present; however, a reach 15 to 20 times the bankfull width was sampled. Total distances sampled typically ranged from 100 to 400 m, depending on stream size (Lyons, 1992; Plafkin and others, 1989). Each stream reach was electrofished in an upstream direction by using pulsed direct current (DC) output (fig. 5). A concerted effort was made at all sampling sites to ensure that all habitats were thoroughly sampled throughout each stream reach. Electrofishing equipment used during sampling was selected on the basis of stream size. Voltage output typically ranged from 150 to 1,000 at 2 to 4 amperes and 60 cycles, depending on conductivity. Measurements of conductivity (capability of water to carry an electrical current) were made prior to electrofishing to determine the appropriate output voltage for effective fish capture (Reynolds, 1996). All streams sampled were wadable and were electrofished by using either a pulsed DC backpack or a tow-barge unit (fig. 5).

**Figure 5.** Electrofishing at Saddle River at Ridgwood, N.J. (01390500) through use of a pulsed direct-current tow-barge unit. (Photograph by Mark Ayers, U.S. Geological Survey)

During the sampling process, stunned fish were netted immediately (5.0-mm mesh) and placed in plastic holding containers strategically located along the stream reach. Sampling effort was measured as a function of total power applied to the electrodes and ranged from 1,400 to 3,800 shocking seconds, depending on stream size and reach length (Reynolds, 1996). After the stream reach was sampled, fish were identified to the species level, counted, examined externally for disease and anomalies, recorded, and released. Reference specimens of unknown or difficult-to-identify species (for example, Notropis spp.) were fixed in 10-percent buffered formalin and retained for validation (Walsh and Meador, 1998). Unidentified specimens were confirmed later by using regional taxonomic keys by Stiles (1978) and Smith (1985) and validated by a State ichthyologist. All fish data were converted into percent abundance (which ranged from 0 to 100 percent for each species) for all statistical analyses. Although two passes were made along each reach at 16 of the 36 sites, only data from the first pass were used in the analysis. Fish-community sampling by using only single-pass electrofishing is an established approach that is applied commonly in many bioassessments (Pusey and others, 1998). To prevent bias associated with sampling effort and over-weighting of highly susceptible species, abundance data were standardized by reach length prior to analysis.

## Macroinvertebrates

Benthic-macroinvertebrate samples were collected during the summers of 1996-98 during stable-flow periods following the protocols of Cuffney and others (1993). To maintain consistency among sites, a random sampling design in which areas of similar substrate composition, current velocity, water depth, and canopy cover was used. The faunistically rich cobble-riffle sections of the stream were the richest targeted habitat (RTH) in northern New Jersey. Five replicate RTH benthic-macroinvertebrate samples were collected with a Slack sampler (0.5 m wide by 0.25 m high, 425-µm mesh) in cobble-riffle sections of the stream channel (fig. 6). The sampler was placed firmly on the stream bottom and the substrate (0.25-$m^2$ area) was disturbed vigorously to dislodge aquatic organisms, which were swept by the current into the trailing net. Large rocks within the sampling area were removed gently, and brushed or rubbed to dislodge attached organisms (fig. 6). The remaining substrate in the sampling area was disturbed to a depth of 0.1 m with a three-pronged hand rake.

**Figure 6.** Slack sampler with 425-micron mesh being used for Richest Targeted Habitat (RTH) sampling in New Jersey (Photograph by Denis Sun, U.S.Geological Survey)

The five RTH samples were composited (Cuffney and others, 1993), for a total sample area of 1.25 $m^2$. Large organic and inorganic debris was inspected, rinsed, and removed. All materials in the samples, including small debris and loose material, were elutriated and sieved through a standard brass 425-µm mesh sieve. The remaining material was placed into a 1-L container, preserved with 10-percent buffered formalin, and shipped to the USGS National Water-Quality Laboratory (NWQL) in Arvada, Colorado, for analysis (Moulton and others, 2000). In the laboratory, samples were sieved and rinsed with tap water to remove fine sediments and excess preservative. A quantitative fixed-count processing method was used to identify and estimate the abundance of each taxon sorted in the samples. Samples were placed on a gridded subsampling frame (constructed of clear Plexiglas™ ) and distributed homogeneously throughout the frame. Multiple sections of the gridded frame (5.1 cm x 5.1 cm) were chosen randomly and a 300-organism subsample was removed systematically, sorted into gross taxonomic categories, and placed in polyseal screw-cap vials containing 70-percent ethanol. This method is similar to the methods described by Barbour and others (1999) and Plafkin and others (1989). Macroinvertebrates were identified later to the lowest possible taxonomic level by using either a 6-50x stereoscopic microscope or a 40-1,000x compound microscope for mounted specimens (for example, Chironomidae and Oligochaeta). Taxonomic specialists were consulted to identify problematic specimens or to validate identifications. A complete explanation of macroinvertebrate processing, identification, and quality-control methods is provided in Moulton and others (2000).

## Attached Algae (Periphyton)

Following the procedures outlined in Porter and others (1993), algal samples covering a discrete area of substrate were acquired through the use a modified syringe-type sampling device (fig. 7). The sampler consists of the cut-off barrel portion of a 30-mL syringe with a rubber gasket cemented to the flanged end (Porter and others, 1993). A relatively flat rock (typically colocated within the same riffle as the invertebrate samples) was removed gently from the stream bottom and placed upright in a white plastic pan. The sampling device then was placed with gasket end down against the upper surface of the rock to create a watertight seal. A small amount of water was added to the enclosed area (barrel of syringe), and a small Dremmel™ brush bit, which was cemented into the end of a plastic rod, was used to detach the

**Figure 7.** Algal sampling equipment used in New Jersey streams including (from left to right) a pipette, the cut-off barrel portion of a 30-milliliter syringe above a flat rock with three circular algal samples removed, and a hand-held cordless drill with a brush bit.

algae from the rock surface (fig. 7). The liquid containing the detached algae was extracted from the barrel of the syringe with a pipette. Five individual area collections (each approximately 15 cm$^2$) from five different riffles within a stream reach were composited to form a single sample (25 total collections) with a total surface area of approximately 75 cm$^2$. To preserve the quantitative algal sample properly, full-strength buffered formaldehyde was added to constitute a 3-percent concentration in the sample jar. Algal samples were shipped to the NWQL. Samples then were transferred to the Academy of Natural Sciences (ANS) in Philadelphia, Pa., for taxonomic identification, counts, and biovolume (total cell volume) estimation. Biovolumes were calculated for the most common taxa (that is, those taxa that made up more than 5 percent of the numerical abundance in at least one sample).

All algal samples received by the ANS were subsampled before complete assessment of soft algae and diatoms was performed. The contents of a sample jar were decanted into an appropriately sized graduated cylinder and the algal material was suspended by shaking or swirling. Known subsample volumes (usually 20 mL) derived from the original sample volume were transferred into glass vials for separate soft-algae and diatom identification and enumeration. All diatom samples were prepared by using the traditional nitric acid digestion approach prior to slide mounting and identification (Academy of Natural Sciences of Philadelphia, 1999a, b). This procedure is necessary to remove organic matter from the valves (frustules) of the diatoms as well as to remove any extraneous organic material that may be present in the sample. After acid digestion, the cleaned algal cells were mounted permanently on glass microscope slides by using a commercially available, toluene-based mounting medium. Soft algae were prepared for assessment by using the Palmer-Maloney counting cell technique (Palmer and Maloney, 1954; Academy of Natural Sciences of Philadelphia, 1999c). From 8 to 50 fields were enumerated from each Palmer-Maloney counting cell. Prepared samples were examined by using a compound microscope with magnification that ranged from 10x to 100x. All diatom and soft-algal species encountered were identified to the lowest possible taxonomic level and enumerated.

## Water Quality

Depth- and width-integrated streamwater samples were collected once at most of the 36 stream sites during June 9-18, 1997, following the methods specified in Shelton (1994) for chemical analysis; at six of the sites, however, samples were collected more frequently (112 stream samples from April 1996 through April 1997). The more frequently acquired samples were used to establish long-term median concentrations of measured constituents for comparison to the single-day concentrations. Results of this comparison were used to validate concentration variability and evaluate the suitability of a constituent for use in further analysis. Samples were analyzed at the NWQL for nutrients, major ions, suspended sediment, dissolved organic carbon (DOC), and suspended organic carbon (SOC). Samples analyzed for DOC and SOC were filtered by using a 0.45-µm silver filter. The filtrate was used to determine DOC and the filters were analyzed for SOC. All surface-water samples were shipped on ice to the NWQL for analysis. Stream temperature, dissolved-oxygen concentration, specific conductance, pH, and alkalinity were measured at the sampling site. Specific conductance and pH were measured with electronic meters and alkalinity was determined by titration.

**Volatile organic compounds (VOCs)**.--All water that contributes to streamflow is susceptible to contamination with VOCs through direct industrial and wastewater discharges, accidental spills of fuel products, improper disposal of industrial solvents, and urban runoff from streets and impervious surfaces (O'Brien and others, 1997). To investigate the effects of these toxic compounds on surface-water quality and aquatic-community health, samples to be analyzed for VOCs were collected at all 36 stream sites during January 27-30, 1997. This period was chosen for sample collection because VOC volatility is minimal during cold weather. A VOC sampler designed by the USGS (Shelton, 1997) was used to collect samples from the center of flow. Samples were acidified with two drops of 1:1 HCl and shipped on ice to the NWQL for analysis. Samples were analyzed for 86 VOCs by using both purge-and-trap isolation and concentration and capillary-column gas chromatograph/mass spectrometry (GC/MS) (Connor and others, 1998). The reporting level for reliable quantification of VOCs ranged from 0.05 to 5.0 µg/L. Only concentrations of the seven most frequently occurring VOCs (cis-1,2-dichloroethene, methyl tert-butyl ether, methylbenzene, naphthalene, tetrachloroethene, trichlorethene, and chloroform) were used in subsequent statistical analyses. Indices such as total VOC concentration and total number of detections also were included.

**Pesticides**.--Pesticides commonly are used to control weeds and insects in urban and agricultural areas throughout New Jersey. In general, their presence in streams is consistent with seasonal application patterns. To assess the effects of pesticide use on water quality, depth- and width-integrated water samples were collected at each of the 36 stream sites during June 9-18, 1997. Following the protocol of Shelton (1994), samples were collected in late spring at or near base-flow conditions. The samples were collected in Teflon bottles and split for multiple analysis through the use of a Teflon cone splitter. Each aliquot was filtered through a 142-mm glass-fiber filter with a nominal 0.7-µm pore opening to remove any suspended particulate matter and shipped on ice to the NWQL. All samples were analyzed for 47 pesticides (25 herbicides, 20 insecticides, and 2 degradation products).

A GC/MS technique was used to detect pesticide compounds accurately at concentrations ranging from 0.001 to 0.018 µg/L (Zaugg and others, 1995). Pesticides measured by using the GC/MS technique were isolated on a C-18 solid-phase extraction column. The sorbed pesticides were eluted from the solid-phase extraction column by using a 3:1 hexane:isopropanol solvent mixture. Results were censored when concentrations were not determined accurately as a result of analytical and (or) other limitations (Reiser and O'Brien, 1999). Quality-control (QC) samples were collected and analyzed to ensure the quality of the data. Field-equipment blanks were analyzed to determine the possibility of contamination during the collection and processing of samples. Other QC techniques, including split samples and spiked reagent water samples, were used to assess the variability in the results obtained by use of standardized analytical techniques and to evaluate potential bias in recovery of analytes from the water sample, respectively (Zaugg and others, 1995). Recoveries of spiked reagent water samples indicated that most of the analytical concentrations were acceptable. Of the 25 pesticides detected, only the concentrations of the five most common (atrazine, desethyl atrazine, metolochlor, prometon, and simazine), whose detection frequencies were 78 percent or higher, were used in the analysis. Indices, such as total-pesticide concentration and total number of detections, were calculated to assess the potential effects of the combination of pesticides in the surface water.

**Trace elements and organochlorine compounds in bed sediment**.--Sediment samples were collected from the upper 2 cm of the streambed in 10 depositional zones along each stream reach and composited for analysis according to guidelines established by Shelton and Capel (1994). Collection of sediments from multiple microhabitats (riffles, runs, and pools) provided a more complete assessment of the concentration of trace elements and organochlorine compounds in streambed sediment than collection from a single habitat. Samples were collected from the surface of the streambed with glass petri dishes and a Teflon spatula and were homogenized into a glass bowl. All material to be analyzed for organochlorines was passed through a 2.0-mm stainless-steel sieve into a 1,000-mL glass jar with the aid of a Teflon-coated spoon. Bed material to be analyzed for trace elements was passed through a 63-µm nylon sieve by flushing continuously with river water into a 500-mL polyethylene jar. Water included in the trace-element sample was decanted after settling of the fine-grained sediments. Sample jars were placed on ice and shipped to the NWQL for analysis. Prior to sampling, all equipment was washed and soaked in 0.2-percent phosphate-free detergent, and rinsed in tap and deionized water. Organochlorine and trace-element sampling equipment was given a final rinse with residue-grade methanol and 5-percent nitric acid solution, respectively. Sampling equipment then was air dried and placed in sealed plastic containers; organochlorine sampling equipment also was wrapped in aluminum foil (Shelton and Capel, 1994). These cleaning procedures are designed specifically to prevent possible equipment contamination.

Sediment samples were analyzed for trace elements and organochlorine compounds including pesticides, pesticide degradative products, and polychlorinated biphenyls (PCBs). Organochlorine compounds used in the analysis include dichlorodiphenyltrichloroethane (DDT) and its degradation products DDD and DDE, aldrin, dieldrin, endrin, PCBs, two metabolic by-products of chlordane (heptachlor and heptachlor epoxide), lindane, mirex, and toxaphene. Two additional indices derived for use in subsequent analyses are total concentration of organochlorines and number of organochlorines found in bed sediment. Organochlorine concentrations were normalized to total organic carbon (TOC) and expressed as micrograms per kilogram (µg/kg) dry weight. For this study, concentrations of DDT, DDE, and DDD were calculated by summing concentrations of the *o,p'*- and *p,p'*- congeners of these compounds. Only trace elements having a high frequency of detection or that were present in elevated concentrations above natural background levels in bottom sediments were used in the analysis. Frequently detected trace elements include arsenic, copper, lead, manganese, mercury, nickel, selenium, and zinc (Ayers and others, 2000). Total concentration of trace elements and number of trace elements in streambed sediments were used as separate indices in the subsequent analysis to assess the potential cumulative effects of trace elements in the environment.

## Watershed Characteristics

The watershed characteristics examined in this study were derived from digital data consisting of mapped polygons of land use and land cover (LU/C), impervious surface area, soils, and elevation.

### Interpretation of Digital Data

The LU/C data set used in this study was developed initially from an interpretation of a New Jersey statewide Integrated Terrain Unit (ITU) coverage (1:24,000 scale) of land use based on aerial photographs taken in 1986 (New Jersey Department of Environmental Protection, 1996). Areas of new development were added to the 1986 land-use data set to create the 1995 land-use data set (Kauffman and others, 2001). The digital data set of new development created by using ortho-rectified digital images of aerial photography flown in March 1995 and March 1997 was obtained from the New Jersey Office of State Planning (NJSOP), Trenton, New Jersey. All digital images meet National Map Accuracy Standards and contain complete metadata information. The impervious-surface information was estimated for all urban polygons on the basis of the combined 1986 and 1995 coverage. The 1973 Geographic Information Retrieval and Analysis System (GIRAS) land-use data (mapped at 1:250,000 scale) also were used for this evaluation; however, the coarser 1973 polygons with a larger mapping unit of 1.4 ha generally led to overestimates of urban land use, and underestimates of agricultural and undeveloped land uses. To adjust for the differences in scale, the GIRAS data set was used to modify the ITU data set to represent 1973 land use (Kauffman and others, 2001). Changes in land use from 1973 to 1995 and from 1986 to 1995 were calculated.

Population estimates were derived from U.S. Bureau of the Census (1992) data updated for 1995. Soil characteristics, including soil depth and permeability, were acquired by using the New Jersey soil-classification data (SSURGO) and supplemented with the State Soil Geographic Database (STATSGO) (U.S. Department of Agriculture, 1991) in areas where SSURGO data were unavailable. All data were converted to a grid or raster format (30 m x 30 m) for final compilation and calculation of statistics.

## Derivation of the Index of Impervious Area

The adverse effects of impervious cover on surface runoff and, in turn, on water quality, aquatic communities, and habitat are well-documented (Schueler, 1994). A spatially distributed index of impervious area for New Jersey was derived from available digital data (Ayers and others, 2000). Mapped impervious-surface-cover (ISC) data for three New Jersey Watershed Management Areas (aggregations of watersheds with similar natural hydrologic and physical features) were available in the 1995-97 LU/C. Mapped ISC data were used to develop and calibrate the impervious-surface index used in our analyses. All geographic analyses were done in raster format. Because road areas are more likely than non-road areas to reflect the area of effective imperviousness in a watershed, impervious area was separated into these two components so that the effects of these two types of impervious surfaces on water quality and aquatic-assemblage structure could be determine separately. A digital coverage of all major and most local roads in New Jersey, derived by the N.J. Department of Transportation from 1991 to 1993 digital ortho-rectified imagery, provided an estimate of road extent. Census data of roads in 1990 (U.S. Bureau of the Census, 1992) were used for the part of New York that drains into New Jersey. A simple assumption that two-lane roads in New Jersey are about 10 m wide was applied to the line coverage. Lines were converted to raster grids (30 m square) and an index of road impervious area for New Jersey was developed.

A regression approach then was used to develop an index of non-road impervious area for urban residential land in New Jersey. The available 1995 LU/C provided the estimate of total impervious area. Non-road impervious area was calculated for each grid cell by subtracting the road from the total impervious area. The 1990 census block information (U.S. Bureau of the Census, 1992) provided spatial estimates of population density and housing density. Equations were developed with the ARC-GRID linear regression command and the following regression equation was chosen for the index:

Non-road impervious area = 23.0 +
(5.72 * natural log of housing density).

The non-road impervious-area equation then was applied to all urban-residential grid cells. Other land-use categories (that is, commercial and industrial) simply were assigned the mean non-road impervious area, computed as ISC (1995-97 LU/C) minus road impervious area. For each grid cell, total impervious area was computed as the sum of road and non-road impervious area. This approach was validated later by using data from 193 New Jersey municipalities in seven counties. Given the total municipal developed area for residential, commercial, industrial, and other land uses, the model was used to determine total impervious-surface area for these land uses with an adjusted $r^2$ of 0.948.

## Flow-Path Length and Topography

The distance a constituent travels along a flow path, the angle of the hillslope, and travel time contribute to its overall effect on water quality and aquatic-assemblage structure at a given sampling site. Distance-weighted negative exponential decay factors applied to explanatory variables (White and others, 1992) on the basis of distance along a flow path from the source to the sampling point is a surrogate for degradation of contaminants as they travel downstream. Each variable was multiplied by a decay factor that ranged from -1 to 0, with -1 representing maximum net loss and 0 representing no net loss (conservative substance). Distance-weighted factors allow sources of influence near sampling sites to be distinguished from those farther upstream (Stackelberg, 1997). For example, if the applied decay factor is high (near -1), sources near a sampling site are given more weight than distant sources (contribute less to contamination at a sampling site), whereas low decay factors (near 0) indicate that distant and nearby sources may be equal contributors of a contaminant (O'Brien, 1997; Stackelberg, 1997).

In-stream distance was used as one measure and in-stream distance divided by Strahler stream-order number (Strahler, 1957) was another measure of the stream travel time for the relative contribution each 900-$m^2$ parcel of land has on the movement of contaminants to a downstream sampling site. A topographic wetness index also was used as a weighting factor because the topography of a basin affects the likelihood that a land parcel will contribute runoff to the stream. For the third- and fourth-order basins evaluated in this study, distance effects were far more important than hillslope effects on downstream sampling sites.

## ANALYTICAL METHODS

### Spatial and Temporal Assessment

Spatial and temporal (year-to-year) variability in aquatic-assemblage structure was evaluated by using two-way indicator species analysis (TWINSPAN), a classification technique (Hill, 1979). The objective of this divisive, hierarchical classification technique is to sort sampling sites into groups such that the degree of association is strong among members of the same group and weak among members of different groups. TWINSPAN analysis reveals patterns among sites and is commonly recommended for use with abundance data because it is a robust technique with a low associated incidence of misclassification. Relativized (proportional) abundance data were used to assess all 36 stream reaches (48 total samples when multiple-year and multiple-reach sites are included). For this analysis, default pseudospecies cut-levels were used with a maximum of four divisions and the data were untransformed. The creation of pseudospecies during the TWINSPAN classification is a means by which the program incorporates abundance data rather than relying only on presence-absence.

Sorenson's percent similarity index (Magurran, 1988) also was used to evaluate spatial and temporal variability in fish and invertebrate assemblages. This approach was used to establish similarities among sites sampled over multiple years and multiple reaches and to complement the TWINSPAN analysis.

### Univariate and Multivariate Approaches

A combination of univariate and multivariate approaches was used to statistically evaluate the relation of the environmental variables to aquatic-assemblage structure. First, univariate procedures were used to assess normality, skewness, and kurtosis of explanatory variables prior to all analyses. Assumptions commonly associated with multivariate procedures are that the samples have multivariate normal distributions and that within-group covariation is equal; however, observational data in community ecology infrequently comply with these statistical assumptions. Therefore, if the data deviated from model assumptions, appropriate transformations (for example, arcsine, square-root, or log) were applied to approximate normality and homoscedasticity.

#### Principal Components Analysis

Second, principal components analysis (PCA) (SAS Institute Inc., 1989) in combination with collinearity assessment was used to reduce the number of environmental variables. PCA is well suited for decreasing the dimensionality of complex data sets (Digby and Kempton,1987; Manly, 1994) and was used as the primary data reduction-technique. PCA is an ordination approach that extracts orthogonal components that are linear combinations of the input variables, with as many components derived as there are variables. The first component of the PCA accounts for the greatest amount of variance in the data set, and each succeeding component accounts for a progressively smaller proportion of the remaining variance. Individual explanatory variables are represented by loadings. These loadings commonly are interpreted to indicate which variables account for most of the variance within each principal component, and which variables are collinear and explain similar portions of the variance. Redundant variables are eliminated from subsequent analyses and surrogate variables that best represent the variance within a group of correlated variables are identified. Because the environmental-variables data set aggregated for this assessment was large (n=413), these data were divided into related-variable groups (for example, nutrients, habitat, and hydrology) prior to PCA. PCA was run separately on each variable group and the correlation matrix was specified. The broken-stick method (Jackson, 1993), rather than the heuristic Kaiser-Guttman procedure of interpreting components with eigenvalues greater than 1, was used to evaluate the number of components to be retained for interpretation. The Kaiser-Guttman approach has been shown to overestimate the number of significant dimensions (Jackson, 1993). Variables with the strongest loadings along significant primary components were retained for further analysis. A Spearman rank correlation matrix (SAS Institute Inc., 1989) of the environmental variables was examined to reduce further the number of collinear variables (that is, significantly correlated variables with an $r^2 > 0.80$).

## Detrended Correspondence Analysis

Third, detrended correspondence analysis (DCA) and multiple linear regression (MLR) were used to build a series of models that describe the relation between the environmental data and the distribution of sites along the ULUG and to reduce further the number of environmental variables. DCA is considered to be one of the most powerful multivariate tools available for assessing patterns in communities composed of species that vary along compositional gradients (Peet and others, 1988). The advantages of simultaneously ordering sites and species along multiple axes have made it an ordination technique favored by ecologists for non-linear data analysis of communities (for example, Wright and others, 1984; Chang and Gauch, 1986; Leland and others, 1986; Ben-Shahar and Skinner, 1988; Edds, 1993; Tate and Heiny, 1995; Galacatos and others, 1996; Zampella and Bunnell, 1998). This unimodal technique is scaled in units of average standard deviation or species turnover (Gauch, 1982). A species appears, rises to its mode, and disappears over a length of approximately four standard deviations, which is equivalent to one species turnover (or beta diversity). DCA arranges sites with similar taxonomic composition to cluster more closely together and sites with dissimilar taxonomic composition to cluster farther apart (Gauch, 1982). Site scores produced through use of DCA can be extracted and related to environmental variables. The site scores represent a coordinate along an ordination axis specifying the x-y location of the sample in ordination space. Ideally, site scores represent the position of communities along an important environmental gradient, such as the ULUG being assessed in this study. DCA is an eigenanalysis and the resulting eigenvalues are equal to the maximized dispersion of the species scores along the ordination axis, and are, thus, a measure of the relative strength of the ordination axis (Jongman and others, 1995).

## Multiple Linear Regression

The first through fourth axis scores extracted from the DCA analysis were used to build multiple linear regression (MLR) models. This step of the analysis was an indirect gradient analysis approach because the axes of variation were interpreted as environmental gradients and were indirectly related to the remaining 60 environmental variables. Two assumptions were made: (1) the first axis of the indirect gradient analysis represents the hypothesized urban gradient, and (2) multiple linear regressions result in best-fit models of ecologically significant explanatory variables.

MLR is a least-squares regression method (that is, it minimizes the sum of squares of the differences between the observed and the expected response), where the response variable (for example, DCA axis 1 scores) is expressed as a function of two or more explanatory variables (for example, human population density, percent urban land, percent impervious surface cover). By using three to five explanatory variables to estimate values of a response variable, we were able to limit errors in prediction and, at the same time, account for a large proportion of the variance in the response variable. Separate analysis of the response for each of the environmental variables cannot replace MLR, especially if some variables are correlated or if interaction exists (Jongman and others, 1995). In these situations, variables that are complementary in explanatory power (Whittaker, 1984) could provide erroneous conclusions if they are assessed separately. Therefore, it is imperative that the investigator select the best set of explanatory variables for use in a multiple regression equation by assessing and reducing sources of multicollinearity (when two or more explanatory variables in a sample are highly correlated). Screening variables prior to MLR modeling can improve predictive power; however, care must be taken not to eliminate variables that are important in providing an ecological description of the relations modeled. Prior to MLR analysis, redundant variables and variables to be used as covariables in partial canonical correspondence analysis were removed. This approach produced many highly significant MLR models and further reduced the number of significant environmental variables from 60 to 28.

## Indices of Aquatic Community Impairment

A separate analysis of aquatic-community-impairment ratings derived from bioassessment criteria were used to build MLR models for fish (Kurtenbach, 1993), invertebrates (New Jersey Department of Environmental Protection, 1994a; Kennen, 1999), and algae (Bahls and others, 1992). For this analysis, the aquatic community indices were assumed to be highly correlated with the inferred urban gradient. That is, as the amount of urban land increased, impairment of the communities increased. Increasing impairment was reflected by decreasing index scores for fish and macroinvertebrates, and an increasing index score for algae.

An index of biotic integrity (IBI) for assessing differences in fish communities in New Jersey streams (Kurtenbach, 1993) was used in this analysis. This approach is a modification of the IBI of Karr and others (1986), which uses various ecological attributes of fish communities to assess directly the integrity of aquatic systems (Fausch and others, 1984). The relevance of the IBI for New Jersey and for this analysis depends on its component metrics, most of which are based on assumptions of how fish communities respond to environmental degradation (Miller and others, 1988; Fausch and others, 1990; Scott and Hall, 1997). The New Jersey IBI consists of 10 biometrics: total number of fish species, number and identity of intolerant species, proportion of individuals as omnivores, proportion of individuals as insectivorous cyprinids, number of individuals in a sample, and proportion of individuals with disease and anomalies (for example, tumors, fin damage, and skeletal abnormalities), proportion of individuals as white suckers (<u>Catostomus</u> <u>commersoni</u>), and number and identity of benthic insectivores. This index has been shown to be a cost-effective and reliable screening tool for assessing stream condition in New Jersey (Kurtenbach, 1993; Chang and others, 2000).

A modification of the U.S. Environmental Protection Agency (USEPA) family-level rapid-bioassessment protocol (RBP II) (Plafkin and others, 1989) was developed and used to assess the condition of macroinvertebrate communities in New Jersey streams. Scoring criteria were based on percent similarity to regional reference sites (N.J. Department of Environmental Protection, 1994b) and five metrics were used to obtain the New Jersey Impairment Score (NJIS): total taxa richness; Ephemeroptera, Plecoptera, and Trichoptera (EPT) richness; percent dominance; modified family biotic index (Hilsenhoff, 1988); and percent EPT. These metrics have been found to be highly sensitive to human-related disturbance and typically decrease with only minimal increases in environmental degradation. Although some redundancy might be suggested between EPT richness and percent EPT, these two biometrics were evaluated and found to be statistically independent (Kennen, 1999). The resulting index was a reliable tool for assessing levels of macroinvertebrate community impairment in New Jersey streams (Kennen, 1999).

An autecological tolerant diatom index (TDI) for algae, which is based on the percent abundances of all species in the genera <u>Navicula</u>, <u>Nitzschia</u>, <u>Cylindrotheca</u>, and <u>Surirella</u>, was used to evaluate the condition of algal communities in New Jersey streams. These are common genera of predominantly motile taxa that can maintain their positions on the substrate surface in depositional environments (Bahls and others, 1992). Because these taxa are able to avoid being buried, they are considered to be more tolerant of sedimentation than other diatoms. In addition, many of these diatoms (for example, <u>Navicula</u>, <u>Nitzschia</u>) are indicative of nutrient and (or) organic enrichment (van Dam and others, 1994). These taxa generally increase in relative density with increasing siltation and other human-related disturbance.

Because the three indices used in this analysis (IBI, NJIS, and TDI) are responsive to various anthropogenic disturbances, they provided a level of complementary evaluation that aided in finding additional environmental factors (n=3) that were indirectly associated with the hypothesized urban gradient. The procedure REG (SAS Institute Inc., 1991) was used to summarize the relations between combinations of environmental variables and the community indices. Stepwise criteria were specified and collinearity, variance inflation factors (VIFs), and the effect of each observation on the estimates and predicted values were evaluated. The significance level to enter the MLR model was set at 0.10 and that to leave the model was 0.05. These options provided a statistically conservative MLR modeling approach. This same approach was used on the extracted DCA axis scores for the ULUG models. In addition, the three indices were used to identify disturbance thresholds (a point along an environmental axis where the response in community condition changes rapidly) by regressing their values on percent total impervious surface cover.

## Partial Canonical Correspondence Analysis

Finally, partial canonical correspondence analysis (pCCA), a direct gradient approach, was used to identify relations among the final set of environmental variables and fish-, invertebrate-, and algal–community structure along the ULUG by evaluating community and environmental data sets simultaneously (ter Braak, 1986b; Jongman and others, 1995). pCCA on relativized abundance data was used to restrict further the effects of variability along natural gradients not fully accounted for through our stratified environmental framework (for example, latitude, drainage area, and altitude). In effect, the residual variation associated with these covariables was "partialed out" (eliminated) from the constrained ordinations (ter Braak, 1986b; ter Braak and Smilauer, 1998). Axes in pCCA are selected as linear combinations of all environmental variables that maximize the dispersion of the species scores. Subsequent axes are acquired by using the same procedure, but have the added constraint of being uncorrelated with (orthogonal to) the covariables (ter Braak, 1986b). The environmental variables in pCCA are represented by arrows (vectors) in the ordination diagram whose lengths are proportional to their importance and whose directions indicate the maximum rate of change in that variable. The relative strength of association of an environmental variable with an ordination axis is reflected in the acuteness of the angle in relation to the axis. A biplot or triplot is produced with species and site points representing the dominant patterns in community composition as explained by the environmental variables (ter Braak, 1986a). As a result, a close arrangement of sites and species tends to reflect greater commonality with regard to an environmental gradient than sites and species that are distant from each other.

A Monte Carlo test of significance (299 random permutations), in which it is assumed that sites are spatially independent and are chosen randomly with respect to the environment, was used to evaluate each environmental variable before it was included in the final pCCA model (ter Braak and Smilauer, 1998). The goal of the Monte Carlo test is to isolate a subset of environmental variables that leads to a reasonable interpretation of important gradients in a few dimensions (Palmer, 1993). Variables whose p-values were less than 0.05 were retained. Variable inflation factors (VIFs) were examined to verify that all environmental variables included in the pCCA contributed unique information to the analysis. VIFs with values > 20, indicating a high degree of collinearity with other variables in the model (ter Braak and Smilauer, 1998), were removed. In addition, canonical correlations were assessed for significance against the first four pCCA axes. Canonical correlations are helpful in determining which environmental variables contribute most to an ecological description of the gradient (Smilauer, 1992). Only environmental variables with a significant relation ($p<0.05$) to an ordination axis are displayed in figures 11-13. An unrestricted Monte Carlo global permutation test (399 permutations) conducted by using the reduced model was used to assess the significance of the species and environment relation for the first canonical axis and the sum of all canonical axes (ter Braak and Smilauer, 1998).

This statistical test has maximum power against a type I error and is based on direct comparison with a simulated distribution of the environmental variables. This distribution is derived from random permutations of the environmental variables that are significant at the 0.05 level (ter Braak and Smilauer, 1998).

### Analysis of Benchmark Communities

Two complementary approaches based on reciprocal averaging (DCA and TWINSPAN) were used to evaluate benchmark communities. These analyses were used to assess community patterns and to investigate hierarchical relations among sites, respectively. The sensitivity of the TWINSPAN clusters was assessed by using a jackknifing procedure (for example, Ibarra and Stewart, 1989). This analysis requires a sequential deletion of sites and calculation of percent persistence of clusters at each node of the original dendrogram (fig. 8). Both analyses were performed on an octave-transformed data matrix (for example, Gauch, 1982, p. 52).

### Resolution of Taxonomic Ambiguities and Data Censoring

Percentage abundance was calculated for each taxon at each site. The potential for ambiguous taxa in the data matrices (invertebrates and algae only) was assessed and rectified where possible. Ambiguous taxa can occur when a species is present in the same data matrix as its parent (genus-level) taxon. This scenario commonly arises when immature or damaged specimens cannot be identified. These ambiguities were resolved where possible by dividing the abundance of the parent taxa among the known children (species) relative to their proportional abundance in the sample. This approach preserves the maximum amount of taxonomic information in the data matrices and provides a consistent data set that increases the validity of intersite comparisons (Cuffney and others, 1997). Some taxonomic ambiguities, however, could not be resolved and were left at the parent level.

Sample-by-species data matrices for fish, invertebrate, and algal taxa were censored by using a predetermined joint frequency and percentage abundance approach. Rare organisms that accounted for less than 0.01 percent of the overall abundance and that were present in fewer than 5 percent of the samples were excluded from the analyses. This approach reduced the number of invertebrate taxa from 267 to 170 and the number of algal taxa from 178 to 103. Only seven species were eliminated from the fish sample-by-species matrix, and all of these were single occurrences.

## VARIABILITY OF AQUATIC COMMUNITIES

### Species Abundance

Data matrices were reduced to 43, 170, and 103 species for analyses of fish-, invertebrate-, and algal-species/environment relations (tables 2, 3, and 4), respectively. Fifty fish species were collected over the 3 years of sampling, including nine non-native and three exotic species (table 2). The white sucker (Catostomus commersoni) was the most abundant fish species and accounted for greater than 20 percent of the mean abundance of fishes sampled. The tessellated darter (Etheostoma olmstedi) was the second most abundant species (greater than 17 percent; table 2). Six species (white sucker, tessellated darter, blacknose dace (Rhinichthys atratulus), spottail shiner (Notropis hudsonius), longnose dace (Rhinichthys cataractae), and American eel (Anguilla rostrata)) together accounted for greater than 70 percent of the mean abundance at the 36 sites.

Net spinning caddisflies of the family Hydropsychidae were the most common benthic invertebrates. These caddisflies accounted for greater than 34 percent of the benthic community and were found at all but 1 of the 36 sites (table 3). Riffle beetles (Elmidae), blackflies (Simuliidae), and baetid mayflies (Baetidae) accounted for 8.9, 6.7, and 3.7 percent of the mean abundance, respectively. Water mites (Hydrachnidia) also were fairly common (2.6 percent), as were the chironomids Polypedilum sp. (2.5 percent) and Rheotanytarsus sp. (4.1 percent). These taxa were found at

**Table 2.** Ecological characteristics and mean percent abundance of fish species collected in 36 New Jersey streams

[Mean percent abundance represents "first-pass" data only; BH, benthic herbivore; BI, benthic insectivore; E, exotic; F, filter feeder; I, insectivore; IS, intolerant species; N, native; NN, non-native (introduced); O, omnivore; P, piscivore; PL, planktivore; T, tolerant species (Halliwell and others, 1999; Kurtenbach, 1993); <, less than]

| ORDER<br>Family<br>*Genus species* | Common name | Taxa code | Trophic guild | Tolerance class | Historical presence | Mean percent abundance |
|---|---|---|---|---|---|---|
| **PETROMYZONTIFORMES** | | | | | | |
| **Petromyzontidae** | | | | | | |
| *Lampetra appendix* (DeKay) | American brook lamprey | ABL | F | IS | N | 0.04 |
| **ANGUILLIFORMES** | | | | | | |
| **Anguillidae** | | | | | | |
| *Anguilla rostrata* (Lesueur) | American eel | EEL | P | T | N | 6.61 |
| **CYPRINIFORMES** | | | | | | |
| **Cyprinidae** | | | | | | |
| *Carassius auratus* (Linnaeus)[1] | Goldfish | GF | O | T | E | < .01 |
| *Cyprinella analostana* Girard | Satinfin shiner | SFS | I | T | N | 1.84 |
| *Cyprinella spiloptera* (Cope) | Spotfin shiner | SPS | I | T | N | .04 |
| *Cyprinus carpio* Linnaeus | Common carp | CRP | O | T | E | .09 |
| *Exoglossum maxillingua* (Lesueur) | Cutlips minnow | CLM | BI | IS | N | 1.30 |
| *Hybognathus regius* Girard | Eastern silvery minnow | ESM | BH | T | N | 1.51 |
| *Luxilus cornutus* (Mitchill) | Common shiner | CMS | I | T | N | 4.27 |
| *Notemigonus crysoleucas* (Mitchill) | Golden shiner | GSH | O | T | N | .05 |
| *Notropis amoenus* (Abbott) | Comely shiner | CS | O | T | N | .14 |
| *Notropis hudsonius* (Clinton) | Spottail shiner | STS | I | T | N | 9.72 |
| *Notropis procne* (Cope) | Swallowtail shiner | SWS | I | T | N | .48 |
| *Pimephales promelas* Rafinesque | Fathead minnow | FHM | O | T | NN | .03 |
| *Rhinichthys atratulus* (Hermann) | Blacknose dace | BND | BI | T | N | 9.34 |
| *Rhinichthys cataractae (Valenciennes)* | Longnose dace | LND | BI | T | N | 7.26 |
| *Semotilus atromaculatus* (Mitchill) | Creek chub | CC | O | T | N | 1.35 |
| *Semotilus corporalis* (Mitchill) | Fallfish | FF | O | T | N | 1.34 |
| **Cobitidae** | | | | | | |
| *Misgurnus anguillicaudatus (Cantor)*[1] | Oriental weatherfish | OW | BI | T | E | < .01 |
| **Catostomidae** | | | | | | |
| *Catostomus commersoni* (Lacepède) | White sucker | WS | O | T | N | 20.61 |
| *Erimyzon oblongus* (Mitchill) | Creek chubsucker | CCS | O | IS | N | .17 |
| **SILURIFORMES** | | | | | | |
| **Ictaluridae** | | | | | | |
| *Ameiurus natalis* (Lesueur) | Yellow bullhead | YBH | O | T | N | .49 |
| *Ameiurus nebulosus* (Lesueur) | Brown bullhead | BBH | O | T | N | .11 |
| *Noturus insignis* (Richardson) | Margined madtom | MM | BI | IS | N | .62 |
| **SALMONIFORMES** | | | | | | |
| **Esocidae** | | | | | | |
| *Esox niger* Lesueur | Chain pickerel | CPR | P | T | N | .11 |
| *Esox americanus americanus* Gmelin | Redfin pickerel | RFP | P | T | N | .37 |

**Table 2.** Ecological characteristics and mean percent abundance of fish species collected in 36 New Jersey streams--Continued

| ORDER<br>Family<br>*Genus species* | Common name | Taxa code | Trophic guild | Tolerance class | Historical presence | Mean percent abundance |
|---|---|---|---|---|---|---|
| **Umbridae** | | | | | | |
| *Umbra pygmaea* (DeKay) | Eastern mudminnow | EMM | O | T | N | .24 |
| **Salmonidae** | | | | | | |
| *Salvelinus fontinalis* (Mitchill) | Brook trout | BRK | P | IS | N | .09 |
| *Salmo trutta* Linnaeus | Brown trout | BRN | P | IS | E | .68 |
| *Oncorhynchus mykiss* (Walbaum) | Rainbow trout | RBT | P | IS | NN | .15 |
| **PERCOPSIFORMES** | | | | | | |
| **Aphredoderidae** | | | | | | |
| *Aphredoderus sayanus* (Gilliams) | Pirate perch | PP | I | T | N | .02 |
| **ATHERINIFORMES** | | | | | | |
| **Cyprinodontidae** | | | | | | |
| *Fundulus diaphanus* (Lesueur) | Banded killifish | BK | I | T | N | 2.23 |
| *Fundulus heteroclitus* (Linnaeus) | Mummichog | MG | O | T | N | .46 |
| **Poeciliidae** | | | | | | |
| *Gambusia affnis* (Baird and Girard)[1] | Western mosquitofish | MF | I | T | E | < .01 |
| **SCORPAENIFORMES** | | | | | | |
| **Cottidae** | | | | | | |
| *Cottus cognatus* Richardson | Slimy sculpin | SS | BI | IS | N | .04 |
| **PERCIFORMES** | | | | | | |
| **Percichthyidae** | | | | | | |
| *Morone americana* (Gmelin)[1] | White perch | WP | P | T | N | < .01 |
| *Morone saxatilis* (Walbaum)[1] | Striped bass | STR | P | T | N | < .01 |
| **Centrarchidae** | | | | | | |
| *Pomoxis nigromaculatus* (Lesueur) | Black crappie | BCR | P | T | NN | .14 |
| *Lepomis macrochirus* Rafinesque | Bluegill sunfish | BG | O | T | NN | 1.29 |
| *Lepomis cyanellus* Rafinesque | Green sunfish | GSF | O | T | NN | 1.64 |
| *Lepomis gibbosus* (Linnaeus) | Pumpkinseed sunfish | PS | O | T | N | 1.40 |
| *Lepomis auritus* (Linnaeus) | Redbreast sunfish | RBS | O | T | N | 3.57 |
| *Enneacanthus gloriosus* (Holbrook) | Bluespotted sunfish | BSS | I | T | N | .03 |
| *Ambloplites rupestris* (Rafinesque) | Rock bass | RKB | P | T | NN | 1.42 |
| *Micropterus salmoides* (Lacepède) | Largemouth bass | LMB | P | T | NN | .88 |
| *Micropterus dolomieu* Lacepède | Smallmouth bass | SMB | P | T | NN | .50 |
| **Percidae** | | | | | | |
| *Percina peltata* (Stauffer) | Shield darter | SD | BI | IS | N | .25 |
| *Etheostoma olmstedi* (Storer) | Tessellated darter | TD | BI | T | N | 17.01 |
| *Stizostedion vitreum* (Mitchill)[1] | Walleye | WAL | P | T | NN | < .01 |
| *Perca flavescens* (Mitchill)[1] | Yellow perch | TP | P | T | N | < .01 |

**TOTAL NUMBER OF SPECIES : 50**

[1]Taxa did not meet selection criteria and were excluded from ordination analysis.

**Table 3.** Total number, mean percent abundance, and frequency of occurrence of macroinvertebrates collected in 36 New Jersey streams

[Taxa are listed by Phylum, CLASS and (or) ORDER, family, *genus*, and *species*. Other taxonomic levels (in parentheses) were used to further differentiate taxa displayed—for example, suborder, subfamily, and tribe; <, less than]

| **Phylum**<br>**CLASS** and (or) ORDER<br>Family<br>*Genus species* | **Taxa code** | **Number of individuals** | **Mean percent abundance** | **Frequency of occurrence** |
|---|---|---|---|---|
| Platyhelminthes | | | | |
| **TURBELLARIA** | TUR_CL | 11,669 | 1.21 | 15 |
| Nemertea | | | | |
| **ENOPLA** | | | | |
| HOPLONEMERTEA | | | | |
| Tetrastemmatidae | | | | |
| *Prostoma* sp. | PRS_SP | 2,134 | .22 | 19 |
| Nematoda | NEM_PH | 8,875 | .92 | 29 |
| Mollusca | | | | |
| **GASTROPODA** | GAS_CL | 587 | .06 | 6 |
| PROSOBRANCHIA[1] | PSB_SO | 85 | < .01 | 2 |
| MESOGASTROPODA | MES_OR | 1,110 | .12 | 1 |
| Viviparidae | | | | |
| Campeloma decisum (Say)[1] | CAM_DEC | 4 | < .01 | 3 |
| Hydrobiidae | HYB_FM | 1,454 | .15 | 4 |
| Pleuroceridae | | | | |
| Elimia virginica (Say)[1] | ELM_VIR | 88 | < .01 | 6 |
| *Pleurocera acuta* (Rafinesque)[1] | PLE_ACU | 6 | < .01 | 4 |
| Ancylidae | ACY_FM | 1,567 | .16 | 14 |
| *Ferrissia* sp. | FER_SP | 331 | .03 | 4 |
| Physidae | | | | |
| *Physella* sp. | PHY_SP | 3,081 | .32 | 12 |
| Planorbidae | PLA_FM | 520 | .05 | 6 |
| *Helisoma anceps* (Menke)[1] | HES_ANC | 57 | < .01 | 3 |
| *Planorbella trivolvis* (Say)[1] | PLA_TRI | 4 | < .01 | 1 |
| **BIVALVIA** | BIV_CL | 1,743 | .18 | 15 |
| Corbiculidae | | | | |
| *Corbicula fluminea* (Muller) | COR_FLU | 3,894 | .41 | 7 |
| Sphaeriidae | SPH_FM | 5,770 | .60 | 12 |
| *Musculium* sp. | MUS_SP | 316 | .03 | 3 |
| *Pisidium* sp.[1] | PIS_SP | 50 | < .01 | 1 |
| *Sphaerium* sp. | SPH_SP | 316 | .03 | 9 |
| Unionidae | | | | |
| *Elliptio complanata* (Lightfoot)[1] | ELL_COM | 4 | < .01 | 3 |
| Annelida | | | | |
| **OLIGOCHAETA** | OLIG_CL | 519 | .05 | 3 |
| LUMBRICULIDA | | | | |

**Table 3.** Total number, mean percent abundance, and frequency of occurrence of macroinvertebrates collected in 36 New Jersey streams--Continued

| **Phylum** **CLASS** and (or) ORDER Family *Genus species* | **Taxa code** | **Number of individuals** | **Mean percent abundance** | **Frequency of occurrence** |
|---|---|---|---|---|
| Lumbriculidae | | | | |
| *Lumbriculus variegates* (Muller) | LUM_VAR | 4,557 | .48 | 17 |
| TUBIFICIDA | | | | |
| Enchytraeidae | ENC_FM | 1,196 | .13 | 6 |
| Naididae | NAID_FM | 5,190 | .54 | 11 |
| Tubificidae | TUB_FM | 8,150 | .85 | 14 |
| Megadrile | MEG_PO | 661 | .07 | 24 |
| **HIRUDINEA** | HIR_CL | 134 | .01 | 2 |
| Erpobdellidae | ERP_FM | 189 | .02 | 10 |
| Arthropoda | | | | |
| **ARACHNIDA** | | | | |
| ACARI | | | | |
| Hydrachnidia | HYDRAC | 24,741 | 2.58 | 34 |
| **CRUSTACEA** | | | | |
| **MALACOSTRACA** | | | | |
| AMPHIPODA | AMP_OR | 10,115 | 1.05 | 25 |
| Crangonyctidae | | | | |
| *Crangonyx* sp.[1] | CRA_SP | 67 | < .01 | 1 |
| Gammaridae | | | | |
| *Gammarus fasciatus* Say | GAM_FAS | 14,539 | 1.52 | 28 |
| ISOPODA | ISP_OR | 9,336 | .97 | 2 |
| Asellidae | | | | |
| *Caecidotea* sp. | CEC_SP | 5,991 | .62 | 9 |
| *Lirceus* sp. | LIR_SP | 4,90 | .05 | 1 |
| DECAPODA | | | | |
| Cambaridae[1] | CMB_FM | 16 | < .01 | 10 |
| *Orconectes limosus* (Rafinesque) [1] | ORC_LIM | 9 | < .01 | 4 |
| **INSECTA** | | | | |
| COLLEMBOLA[1] | COL_OR | 27 | < .01 | 3 |
| EPHEMEROPTERA | EPH_OR | 3,036 | .32 | 14 |
| Baetidae | BAE_FM | 21,701 | 2.26 | 29 |
| *Acentrella* sp. | ACE_SP | 4,237 | .44 | 11 |
| *Baetis* sp. | BAE_SP | 5,016 | .52 | 24 |
| *Baetis flavistriga* McDunnough | BAE_FLA | 1,427 | .15 | 8 |
| *Baetis intercalaris* McDunnough | BAE_INT | 1,317 | .14 | 9 |
| *Baetis pluto* McDunnough[1] | BAE_PLU | 1 | < .01 | 1 |
| *Heterocloeon* sp. | HET_SP | 1,801 | .19 | 9 |
| Caenidae | | | | |
| *Caenis* sp. | CAE_SP | 2,170 | .23 | 10 |
| Ephemerellidae | EPH_FM | 31,829 | 3.33 | 20 |
| *Drunella cornutella* (McDunnough) [1] | DRU_COR | 1 | < .01 | 1 |

**Table 3.** Total number, mean percent abundance, and frequency of occurrence of macroinvertebrates collected in 36 New Jersey streams--Continued

| Phylum<br>**CLASS** and (or) ORDER<br>Family<br>*Genus species* | Taxa code | Number of individuals | Mean percent abundance | Frequency of occurrence |
|---|---|---|---|---|
| *Eurylophella aestiva* (McDunnough) [1] | EUR_AES | 10 | < .01 | 1 |
| *Eurylophella versimilis* (McDunnough) [1] | EUR_VER | 21 | < .01 | 1 |
| *Serratella* sp. | SER_SP | 1,541 | .16 | 4 |
| Heptageniidae | HEP_FM | 13,932 | 1.45 | 26 |
| *Leucrocuta* sp. | LEU_SP | 252 | .03 | 1 |
| *Stenacron interpunctatum* (Say) [1] | STE_INT | 62 | < .01 | 2 |
| *Stenacron* sp.[1] | STE_SP | 14 | < .01 | 1 |
| *Stenonema* sp. | STN_SP | 3,243 | .34 | 15 |
| *Stenonema ithaca* (Clemens and Leonard) [1] | STN_ITH | 1 | < .01 | 1 |
| *Stenonema mediopunctatum* (McDunnough) [1] | STN_MED | 4 | < .01 | 2 |
| *Stenonema modestum* (Banks) [1] | STN_MOD | 50 | < .01 | 3 |
| *Stenonema smithae* Traver[1] | STN_SMI | 7 | < .01 | 2 |
| Isonychiidae | | | | |
| *Isonychia* sp. | ISO_SP | 10,260 | 1.07 | 20 |
| Leptohyphidae | | | | |
| *Leptohyphes robacki* Allen | LTO_ROB | 118 | .01 | 2 |
| *Tricorythodes* sp. | TRI_SP | 583 | .06 | 5 |
| Leptophlebiidae[1] | LEPT_FM | 69 | < .01 | 2 |
| *Paraleptophlebia* sp.[1] | PAL_SP | 1 | < .01 | 1 |
| Polymitarcyidae | | | | |
| *Ephoron leukon* Williamson[1] | EPR_LEU | 3 | < .01 | 1 |
| Potamanthidae | | | | |
| *Anthopotamus* sp. | ANP_SP | 339 | .04 | 4 |
| ODONATA | | | | |
| Anisoptera (Suborder) | | | | |
| Aeshnidae | | | | |
| *Boyeria* sp.[1] | BOY_SP | 3 | < .01 | 1 |
| *Boyeria vinosa* (Say) [1] | BOY_VIN | 9 | < .01 | 1 |
| Cordulegastridae | | | | |
| *Cordulegaster obliqua* (Say) [1] | CRD_OBL | 1 | < .01 | 1 |
| Gomphidae[1] | GOM_FM | 60 | < .01 | 5 |
| *Lanthus parvulus* (Selys) [1] | LAN_PAR | 2 | < .01 | 2 |
| *Ophiogomphus* sp.[1] | OPH_SP | 2 | < .01 | 1 |
| *Stylogomphus albistylus* (Hagen) [1] | STY_ALB | 2 | < .01 | 2 |
| Macromiidae | | | | |

**Table 3.** Total number, mean percent abundance, and frequency of occurrence of macroinvertebrates collected in 36 New Jersey streams--Continued

| Phylum<br>CLASS and (or) ORDER<br>Family<br>*Genus species* | Taxa code | Number of individuals | Mean percent abundance | Frequency of occurrence |
|---|---|---|---|---|
| *Macromia illinoiensis* Walsh[1] | MCM_ILL | 1 | < .01 | 1 |
| Zygoptera (Suborder) | ZYG_SO | 568 | .06 | 1 |
| Calopterygidae | CAL_FM | 141 | .02 | 1 |
| *Calopteryx* sp.[1] | CAL_SP | 22 | < .01 | 1 |
| Coenagrionidae | COE_FM | 133 | .01 | 5 |
| *Argia* sp. | ARG_SP | 448 | .08 | 9 |
| PLECOPTERA | PLC_OR | 474 | .05 | 4 |
| Chloroperlidae | CHL_FM | 155 | .02 | 2 |
| *Sweltsa* sp.[1] | SWE_SP | 64 | < .01 | 2 |
| Leuctridae | | | | |
| *Leuctra* sp.[1] | LEC_SP | 15 | < .01 | 1 |
| Perlidae | PER_FM | 215 | .02 | 5 |
| *Acroneuria* sp. | ACR_SP | 725 | .08 | 14 |
| *Agnetina* sp.[1] | AGN_SP | 52 | < .01 | 4 |
| *Paragnetina immarginata* (Say) [1] | PAR_IMM | 30 | < .01 | 2 |
| *Paragnetina media* (Walker) | PAR_MED | 289 | .03 | 14 |
| *Perlesta placida* complex | PER_PLA | 114 | .01 | 2 |
| Peltoperlidae | | | | |
| *Tallaperla* sp.[1] | TAL_SP | 13 | < .01 | 4 |
| Pteronarcyidae | | | | |
| *Pteronarcys* sp.[1] | PTE_SP | 18 | < .01 | 3 |
| Taeniopterygidae | | | | |
| *Taeniopteryx* sp. | TAE_SP | 4,960 | .51 | 3 |
| HEMIPTERA | | | | |
| Corixidae | | | | |
| *Sigara* sp.[1] | SIG_SP | 9 | < .01 | 1 |
| Gerridae | | | | |
| *Aquarius remigis* (Say) [1] | AQU_REM | 2 | < .01 | 1 |
| *Metrobates* sp.[1] | MET_SP | 60 | < .01 | 1 |
| Notonectidae | | | | |
| *Notonecta* sp.[1] | NOT_SP | 4 | < .01 | 1 |
| Saldidae | | | | |
| *Saldula* sp.[1] | SAU_SP | 1 | < .01 | 1 |
| Veliidae | | | | |
| *Microvelia* sp.[1] | MCV_SP | 3 | < .01 | 3 |
| *Rhagovelia obesa* Uhler | RHA_OBE | 672 | .07 | 12 |
| MEGALOPTERA | | | | |
| Corydalidae | | | | |
| *Corydalus cornutus* (Linnaeus) | COR_COR | 1,051 | .11 | 26 |
| *Nigronia serricornis* (Say) | NIG_SER | 668 | .07 | 17 |

**Table 3.** Total number, mean percent abundance, and frequency of occurrence of macroinvertebrates collected in 36 New Jersey streams--Continued

| **Phylum** **CLASS** and (or) ORDER Family *Genus species* | **Taxa code** | **Number of individuals** | **Mean percent abundance** | **Frequency of occurrence** |
|---|---|---|---|---|
| Sialidae | | | | |
| *Sialis* sp. | SIA_SP | 229 | .02 | 9 |
| NEUROPTERA | | | | |
| Sisyridae | | | | |
| *Climacia* sp.[1] | CLM_SP | 84 | < .01 | 2 |
| TRICHOPTERA | TRC_FM | 1,082 | .11 | 10 |
| Annulipalpia (Suborder)[1] | ANN_SO | 59 | < .01 | 1 |
| Glossosomatidae | | | | |
| *Glossosoma lividum* (Hagen) | GLO_LIV | 108 | .01 | 2 |
| *Glossosoma nigrior* Banks | GLO_NIG | 188 | .02 | 4 |
| *Glossosoma* sp. | GLO_SP | 3,406 | .36 | 10 |
| *Protoptila* sp. | PRT_SP | 1,161 | .12 | 8 |
| Hydropsychidae | HYD_FM | 182,179 | 18.98 | 35 |
| *Ceratopsyche bronta* (Ross) | CER_BRO | 2,347 | .24 | 14 |
| *Ceratopsyche morosa* (Hagen) | CER_MOR | 2,684 | .28 | 12 |
| *Ceratopsyche slossonae* (Banks) | CER_SLO | 276 | .03 | 3 |
| *Ceratopsyche sparna* (Ross) | CER_SPA | 2,944 | .31 | 12 |
| *Ceratopsyche* sp. | CER_SP | 31,812 | 3.31 | 27 |
| *Ceratopsyche walkeri* (Betten and Mosely) | CER_WAL | 322 | .03 | 3 |
| *Cheumatopsyche pettiti* (Banks) [1] | CHE_PET | 23 | < .01 | 2 |
| *Cheumatopsyche* sp. | CHE_SP | 74,434 | 7.75 | 35 |
| *Diplectrona modesta* Banks | DIP_MOD | 145 | .02 | 1 |
| *Hydropsyche betteni* Ross | HYD_BET | 15,210 | 1.58 | 28 |
| *Hydropsyche demora* Ross | HYD_DEM | 342 | .04 | 3 |
| *Hydropsyche depravata* group | HYD_DEP | 6,168 | .64 | 1 |
| *Hydropsyche leonardi* Ross | HYD_LEO | 1,389 | .15 | 4 |
| *Hydropsyche scalaris* Hagen | HYD_SCA | 327 | .03 | 3 |
| *Hydropsyche* sp. | HYD_SP | 11,768 | 1.27 | 31 |
| *Macrostemum carolina* (Banks) | MAC_CAR | 319 | .03 | 4 |
| *Macrostemum zebratum* (Hagen) | MAC_ZEB | 103 | .01 | 4 |
| Hydroptilidae | | | | |
| *Hydroptila gunda* Milne[1] | HYT_GUN | 1 | < .01 | 1 |
| *Hydroptila perdita* Morton[1] | HYT_PER | 1 | < .01 | 1 |
| *Hydroptila spatulata* Morton[1] | HYT_SPA | 34 | < .01 | 1 |
| *Hydroptila* sp. | HYT_SP | 2,162 | .23 | 13 |
| *Leucotrichia pictipes* (Banks) | LUC_PIC | 1,395 | .15 | 9 |
| *Leucotrichia* sp.[1] | LUC_SP | 26 | < .01 | 2 |
| Philopotamidae | | | | |
| *Chimarra aterrima* Hagen[1] | CHI_ATE | 28 | < .01 | 1 |
| *Chimarra obscura* (Walker) [1] | CHI_OBS | 2 | < .01 | 2 |

**Table 3.** Total number, mean percent abundance, and frequency of occurrence of macroinvertebrates collected in 36 New Jersey streams--Continued

| **Phylum**<br>**CLASS** and (or) ORDER<br>Family<br>*Genus species* | **Taxa code** | **Number of individuals** | **Mean percent abundance** | **Frequency of occurrence** |
|---|---|---|---|---|
| *Chimarra* sp. | CHI_SP | 34,782 | 3.62 | 29 |
| *Dolophilodes distinctus* (Walker) [1] | DOL_DIS | 393 | .04 | 3 |
| *Wormaldia* sp.[1] | WOR_SP | 5 | < .01 | 1 |
| Polycentropodidae | POL_FM | 223 | .02 | 2 |
| *Polycentropus* sp.[1] | POL_SP | 5 | < .01 | 1 |
| *Neureclipsis* sp. | NUR_SP | 449 | .05 | 4 |
| Psychomyiidae | | | | |
| *Psychomyia nomada* (Ross) | PSY_NOM | 673 | .07 | 8 |
| Rhyacophlidae | | | | |
| *Rhyacophila fuscula* (Walker) | RHY_FUS | 139 | .01 | 6 |
| *Rhyacophila* sp. | RHY_SP | 101 | .01 | 3 |
| Integripalpia (Suborder) | | | | |
| Apataniidae | | | | |
| *Apatania incerta* (Banks) | APA_INC | 1,369 | .14 | 10 |
| Brachycentridae | | | | |
| *Brachycentrus appalachia* Flint | BRA_APP | 1,577 | .16 | 1 |
| *Brachycentrus nigrosoma* (Banks) | BRA_NIG | 179 | .02 | 1 |
| *Brachycentrus numerosus* (Say) | BRA_NUM | 272 | .03 | 3 |
| *Micrasema rusticum* (Hagen) [1] | MIC_RUS | 29 | < .01 | 1 |
| *Micrasema* sp. | MIC_SP | 1,595 | .17 | 11 |
| Helicopsychidae | | | | |
| *Helicopsyche borealis* Hagen | HEL_BOR | 10,187 | 1.06 | 9 |
| Lepidostomatidae | | | | |
| *Lepidostoma* sp. | LPD_SP | 3,088 | .32 | 8 |
| Leptoceridae | LPT_FM | 709 | .07 | 3 |
| *Ceraclea* sp.[1] | CRC_SP | 51 | < .01 | 2 |
| *Oecetis avara* (Banks) | OEC_AVA | 151 | .07 | 2 |
| *Oecetis inconspicua* complex[1] | OEC_INC | 67 | < .01 | 1 |
| *Oecetis persimilis* (Banks) | OEC_PER | 274 | .03 | 2 |
| *Oecetis* sp. | OEC_SP | 239 | .03 | 6 |
| Limnephilidae | | | | |
| *Hydatophylax* sp.[1] | HYP_SP | 65 | < .01 | 1 |
| Uenoidae | | | | |
| *Neophylax* sp. | NEO_SP | 192 | .02 | 7 |
| Odontoceridae | | | | |
| *Psilotreta frontalis* (Banks)[1] | PSI_FRO | 1 | < .01 | 1 |
| LEPIDOPTERA[1] | LEP_OR | 35 | < .01 | 2 |
| Pyralidae | | | | |
| *Petrophila* sp. | PET_SP | 5,390 | .56 | 5 |
| COLEOPTERA | | | | |

**Table 3.** Total number, mean percent abundance, and frequency of occurrence of macroinvertebrates collected in 36 New Jersey streams--Continued

| **Phylum**<br>**CLASS** and (or) ORDER<br>Family<br>*Genus species* | **Taxa code** | **Number of individuals** | **Mean percent abundance** | **Frequency of occurrence** |
|---|---|---|---|---|
| Adephaga (Suborder) | | | | |
| Gyrinidae | | | | |
| *Dineutus ciliatus* (Forsberg) [1] | DIN_CIL | 3 | < .01 | 1 |
| *Dineutus discolor* Aubé[1] | DIN_DIS | 7 | < .01 | 3 |
| *Dineutus* sp. | DIN_SP | 124 | .01 | 3 |
| Polyphaga (Suborder) | | | | |
| Curculionidae[1] | CUR_FM | 30 | < .01 | 1 |
| Elmidae | EMD_FM | 11,063 | 1.15 | 30 |
| *Ancyronyx variegata* (Germar) | ANY_VAR | 289 | .03 | 1 |
| *Dubiraphia* sp. | DUB_SP | 278 | .03 | 4 |
| *Macronychus glabratus* Say | MCR_GLA | 466 | .05 | 9 |
| *Microcylloepus pusillus* (LeConte) | MIO_PUL | 242 | .03 | 4 |
| *Microcylloepus* sp. | MIO_SP | 126 | .01 | 2 |
| *Optioservus fastiditus* (LeConte) [1] | OPT_FAS | 30 | < .01 | 2 |
| *Optioservus ovalis* (LeConte) | OPT_OVA | 244 | .03 | 4 |
| *Optioservus* sp. | OPT_SP | 12,151 | 1.27 | 24 |
| *Optioservus trivittatus* (Brown) | OPT_TRI | 2,400 | .25 | 18 |
| *Oulimnius latiusculus* (LeConte) | OUL_LAT | 3,221 | .34 | 15 |
| *Promoresia elegans* (LeConte) | PRO_ELE | 642 | .07 | 4 |
| *Promoresia tardella* (Fall) | PRO_TAR | 568 | .06 | 10 |
| *Promoresia* sp. | PRO_SP | 3,021 | .32 | 10 |
| *Stenelmis concinna* Sanderson | STL_CON | 165 | .02 | 5 |
| *Stenelmis crenata* (Say) | STL_CRE | 6,143 | .64 | 23 |
| *Stenelmis mera* Sanderson[1] | STL_MER | 1 | < .01 | 1 |
| *Stenelmis sandersoni* Musgrave | STL_SAN | 654 | .07 | 4 |
| *Stenelmis* sp. | STL_SP | 43,519 | 4.53 | 31 |
| Hydrophilidae | | | | |
| *Enochrus* sp.[1] | ENO_SP | 51 | < .01 | 1 |
| *Paracymus* sp.[1] | PAC_SP | 6 | < .01 | 1 |
| *Sperchopsis tessellata* (Ziegler) [1] | SPE_TES | 5 | < .01 | 1 |
| *Tropisternus lateralis* (Fabricius) [1] | TRO_LAT | 4 | < .01 | 1 |
| Psephenidae | | | | |
| *Ectopria* sp.[1] | ECT_SP | 35 | < .01 | 3 |
| *Psephenus herricki* (DeKay) | PSE_HER | 10,541 | 1.10 | 27 |
| Ptilodactylidae | | | | |
| *Anchytarsus bicolor* (Melsheimer) [1] | ANC_BIC | 1 | < .01 | 1 |
| Staphylinidae | | | | |
| *Stenus* sp.[1] | STS_SP | 3 | < .01 | 1 |
| DIPTERA | DIPT_OR | 285 | .03 | 6 |
| Brachycera (Suborder) [1] | BRA_SO | 15 | < .01 | 2 |

**Table 3.** Total number, mean percent abundance, and frequency of occurrence of macroinvertebrates collected in 36 New Jersey streams--Continued

| **Phylum**<br>**CLASS** and (or) ORDER<br>Family<br>*Genus species* | **Taxa code** | **Number of individuals** | **Mean percent abundance** | **Frequency of occurrence** |
|---|---|---|---|---|
| Athericidae | | | | |
| *Atherix lantha* Webb | ATH_LAN | 108 | .01 | 6 |
| Empididae | | | | |
| *Chelifera* sp. | CEL_SP | 1,242 | .13 | 6 |
| *Clinocera* sp.[1] | CLN_SP | 11 | < .01 | 1 |
| *Hemerodromia* sp. | HEM_SP | 13,939 | 1.45 | 30 |
| *Wiedemannia* sp.[1] | WIE_SP | 71 | < .01 | 1 |
| Tabanidae | | | | |
| *Chrysops* sp.[1] | CRY_SP | 1 | < .01 | 1 |
| *Tabanus* sp.[1] | TAB_SP | 4 | < .01 | 2 |
| Nematocera (Suborder) | | | | |
| Ceratopogonidae[1] | CRT_FM | 54 | < .01 | 2 |
| *Atrichopogon* sp.[1] | ATR_SP | 25 | < .01 | 1 |
| Chironomidae | CHR_FM | 1,197 | .13 | 34 |
| Diamesinae (Subfamily) | | | | |
| Diamesini (Tribe) | | | | |
| *Diamesa* sp. | DIA_SP | 1,232 | .13 | 7 |
| *Pagastia* sp.[1] | PAG_SP | 5 | < .01 | 1 |
| *Potthastia* sp.[1] | POT_SP | 29 | < .01 | 1 |
| Chironominae (Subfamily) [1] | CHR_SF | 3,864 | .40 | 22 |
| Chironomini (Tribe) | CHR_TR | 561 | .06 | 7 |
| *Cryptochironomus* sp. | CRP_SP | 105 | .01 | 2 |
| *Dicrotendipes* sp. | DIC_SP | 514 | .05 | 3 |
| *Microtendipes* sp. | MRT_SP | 2,023 | .21 | 4 |
| *Nilothauma* sp.[1] | NIO_SP | 50 | < .01 | 1 |
| *Phaenopsectra* sp. | PHA_SP | 124 | .01 | 3 |
| *Polypedilum* sp. | PLP_SP | 24,085 | 2.51 | 28 |
| *Saetheria* sp.[1] | SAE_SP | 42 | < .01 | 1 |
| *Stenochironomus* sp. | STC_SP | 406 | 0.04 | 5 |
| *Tribelos* sp.[1] | TRB_SP | 42 | < .01 | 1 |
| *Xenochironomus* sp. | XEN_SP | 171 | .02 | 5 |
| Tanytarsini (Tribe) | TAN_TB | 2,112 | .22 | 12 |
| *Cladotanytarsus* sp. | CLA_SP | 412 | .04 | 1 |
| *Micropsectra* sp. | MRO_SP | 1,867 | .19 | 2 |
| *Neozavrelia* sp. | NEZ_SP | 101 | .01 | 1 |
| *Paratanytarsus* sp. | PAT_SP | 433 | .05 | 2 |
| *Rheotanytarsus* sp. | RHT_SP | 39,256 | 4.09 | 29 |
| *Stempellinella* sp. | STP_SP | 653 | .07 | 4 |
| *Sublettea* sp. | SUB_SP | 697 | .07 | 3 |
| *Tanytarsus* sp. | TAN_SP | 3,413 | .36 | 15 |

**Table 3.** Total number, mean percent abundance, and frequency of occurrence of macroinvertebrates collected in 36 New Jersey streams--Continued

| **Phylum** **CLASS** and (or) ORDER Family *Genus species* | **Taxa code** | **Number of individuals** | **Mean percent abundance** | **Frequency of occurrence** |
|---|---|---|---|---|
| Orthocladiinae (Subfamily) | ORT_FM | 4,318 | .45 | 27 |
| *Brillia* sp. | BRI_SP | 268 | .03 | 2 |
| *Cardiocladius* sp. | CAR_SP | 3,092 | .32 | 12 |
| *Corynoneura* sp.[1] | COY_SP | 34 | < .01 | 2 |
| *Cricotopus bicinctus* group | CRI_BIC | 10,310 | 1.07 | 21 |
| *Cricotopus trifascia* group | CRI_TRIg | 1,157 | .12 | 9 |
| *Cricotopus* sp. | CRI_SP | 2,260 | .23 | 1 |
| *Eukiefferiella* sp. | EUK_SP | 638 | .07 | 9 |
| *Nanocladius* sp. | NAN_SP | 536 | .06 | 5 |
| *Orthocladius* sp. | ORT_SP | 9,909 | 1.03 | 22 |
| *Parakiefferiella* sp.[1] | PAK_SP | 25 | < .01 | 1 |
| *Parametriocnemus* sp. | PAE_SP | 1,050 | .11 | 9 |
| *Paraphaenocladius* sp.[1] | PAP_SP | 29 | < .01 | 1 |
| *Rheocricotopus* sp. | RHE_SP | 1,459 | .15 | 8 |
| *Synorthocladius* sp. | SYN_SP | 403 | .04 | 2 |
| *Thienemanniella* sp. | THI_SP | 497 | .05 | 9 |
| *Tvetenia* sp. | TVE_SP | 10,323 | 1.08 | 24 |
| *Xylotopus* sp.[1] | XYL_SP | 20 | < .01 | 1 |
| Tanypodinae (Subfamily) | TAN_SF | 365 | .04 | 5 |
| Pentaneurini (Tribe) | PEN_TB | 940 | .10 | 7 |
| *Ablabesmyia* sp. | ABL_SP | 203 | .02 | 5 |
| *Larsia* sp.[1] | LAR_SP | 40 | < .01 | 1 |
| *Paramerina* sp.[1] | PAM_SP | 40 | < .01 | 1 |
| *Thienemannimyia* group sp. | THE_SP | 9,318 | .97 | 20 |
| *Trissopelopia* sp.[1] | TRS_SP | 20 | < .01 | 1 |
| Dixidae | | | | |
| *Dixa* sp.[1] | DIX_SP | 11 | < .01 | 1 |
| Simuliidae | SIM_FM | 54,971 | 5.73 | 34 |
| *Simulium* sp. | SIM_SP | 8,860 | .92 | 29 |
| Tipulidae | | | | |
| *Antocha* sp. | ANT_SP | 5,980 | .62 | 17 |
| *Dicranota* sp.[1] | DCR_SP | 29 | < .01 | 1 |
| *Hexatoma* sp.[1] | HEX_SP | 50 | < .01 | 5 |
| *Tipula* sp. | TIP_SP | 210 | .02 | 16 |

[1] Taxa did not meet selection criteria and were excluded from ordination analysis.

**Table 4.** Total density, mean percent density, and frequency of occurrence of algal taxa collected in 36 New Jersey streams

[$cm^2$, square centimeters; ANS, Academy of Natural Sciences, Philadelphia, Pa.; FWA, Freshwater algae; <, less than]

| Group *Genus species* | Taxa code | Total density (cells/$cm^2$) | Mean percent density | Frequency |
|---|---|---|---|---|
| **Cyanochloronta (Blue-green algae)** | | | | |
| *Calothrix* sp.[1] | CALOT_BG | 44,150,302 | 28.78 | 35 |
| *Chamaesiphon* sp.[2] | CHLAM_BG | 59 | < .01 | 1 |
| *Entophysalis* sp. | ENTO_BG | 239,885 | .16 | 1 |
| *Hydrocoleum brebissonii* Kütz. | HYDRO_BG | 749,454 | .49 | 5 |
| *Lyngbya* sp. | LYNG_BG | 45,952 | .03 | 1 |
| *Oscillatoria* sp. | OCIL_BG | 2,579,890 | 1.68 | 18 |
| *Oscillatoria* sp. 1 ANS FWA | OSCIL1_BG | 801,195 | .52 | 3 |
| *Schizothrix friesii* (Ag.) Gom. | SCHIZ_BG | 393,377 | .26 | 5 |
| **Chlorophycophyta (Green algae)** | | | | |
| *Ankistrodesmus falcatus* (Corda) Ralfs | ANKFAL_G | 20,602 | .01 | 5 |
| *Chlamydomonas* sp. | CHLAMY_G | 18,570 | .01 | 2 |
| *Cladophora* sp. | CLADOP_G | 22,496 | .02 | 3 |
| *Coelastrum microporum* Naeg.[2] | COEMIC_G | 13,511 | < .01 | 1 |
| *Cosmarium* sp.[2] | COS_SP_G | 3,282 | < .01 | 1 |
| *Crucigenia* sp.[2] | CRU_SP_G | 6,755 | < .01 | 1 |
| *Gloeocystis* sp. | GLOEOC_G | 68,170 | .04 | 2 |
| *Kirchneriella* sp. | KIRCHN_G | 19,721 | .01 | 2 |
| *Lagerheimia* sp.[2] | LAG_SP_G | 1,203 | < .01 | 1 |
| *Oedogonium* sp. | OEDOG_G | 58,981 | .04 | 3 |
| *Pandorina morum* Bory[2] | PANMOR_G | 1,370 | < .01 | 1 |
| *Pediastrum tetras* (Ehr.) Ralfs | PEDTET_G | 56,818 | .04 | 2 |
| *Protoderma viride* Kütz.[1] | PROTOD_G | 57,692,471 | 37.61 | 22 |
| *Scenedesmus acuminatus* (Lagerh.) Chod. | SCE_AC_G | 71,272 | .05 | 1 |
| *Scenedesmus acutus* Meyen | SCEACU_G | 23,310 | .02 | 1 |
| *Scenedesmus arcuatus* Lemm. | SCE_AR_G | 17,302 | .01 | 1 |
| *Scenedesmus denticulatus* Kirch. | SCE_DE_G | 31,120 | .02 | 1 |
| *Scenedesmus ecornis* (Ralfs) Chod. | SCE_EC_G | 52,958 | .04 | 1 |
| *Scenedesmus quadricauda* (Turp.) Bréb. | SCE_QU_G | 14,897 | .01 | 2 |
| *Scenedesmus* sp. | SCENED_G | 27,500 | .02 | 2 |
| *Scenedesmus spinosus* Chod. | SCE_SP_G | 112,468 | .07 | 2 |
| *Treubaria crassipina* G.M. Smith[2] | TRECRA_G | 177 | < .01 | 1 |
| **Euglenophycophyta (Euglenophyte algae)** | | | | |
| *Trachelomonas hispida* (Perty) Stein[2] | TRAHIS_G | 2,338 | < .01 | 3 |
| *Trachelomonas* sp.[2] | TRA_SP_G | 540 | < .01 | 1 |
| *Trachelomonas volvocina* Ehr.[2] | TRAVOL_G | 2,766 | < .01 | 3 |

**Table 4.** Total density, mean percent density, and frequency of occurrence of algal taxa collected in 36 New Jersey streams--Continued

| Group<br>*Genus species* | Taxa code | Total density (cells/cm$^2$) | Mean percent density | Frequency |
|---|---|---|---|---|
| **Bacillariophyceae (Diatoms)** | | | | |
| *Achnanthes affinis* Grun. | ACAFFINI | 439,185 | .29 | 17 |
| *Achnanthes biasolettiana* (Kütz.) Grun.[2] | ACBIASOL | 55 | < .01 | 1 |
| *Achnanthes bioreti* Germain[2] | ACBIORET | 7,516 | < .01 | 1 |
| *Achnanthes clevei* Grun. | ACCLEVEI | 12,335 | .01 | 6 |
| *Achnanthes deflexa* Reim. | ACDEFLEX | 6,751,591 | 4.40 | 28 |
| *Achnanthes detha* Hohn & Hellerm.[2] | ACDETHA | 6,035 | < .01 | 1 |
| *Achnanthes exigua* Grun. | ACEXIGUA | 44,665 | .03 | 11 |
| *Achnanthes helvetica* (Hust.) Lange-Bert.[2] | ACHELVET | 609 | < .01 | 1 |
| *Achnanthes hungarica* (Grun.) Grun.[2] | ACHUNGAR | 831 | < .01 | 1 |
| *Achnanthes laevis* Schimanski[2] | ACLAEVIS | 111 | < .01 | 1 |
| *Achnanthes lanceolata* (Bréb. in Kütz.) Grun. | ACLANCEO | 1,231,016 | .80 | 32 |
| *Achnanthes lanceolata apiculata* Patr.[2] | ACLANAPI | 554 | < .01 | 1 |
| *Achnanthes lanceolata* dubia Grun.[2] | ACLANDUB | 469 | < .01 | 1 |
| *Achnanthes lewisiana* Patr. | ACLEWISI | 1,897,733 | 1.24 | 8 |
| *Achnanthes linearis* (W. Sm.) Grun.[2] | ACLINEAR | 55 | < .01 | 1 |
| *Achnanthes minutissima* Kütz. | ACMINUTI | 3,135,383 | 2.04 | 20 |
| *Achnanthes obliqua* MARR Hust.[2] | ACCFOBLQ | 221 | < .01 | 1 |
| *Achnanthes oblongella* Østr.[2] | ACOBLONG | 1,461 | < .01 | 1 |
| *Achnanthes petersonii* Hust. | ACPETERS | 22,535 | .02 | 2 |
| *Achnanthes pinnata* Hust. | ACPINNAT | 38,495 | .03 | 9 |
| *Amphora ovalis* (Kütz.) Kütz. | AMOVALIS | 22,840 | .02 | 8 |
| *Amphora perpusilla* (Grun.) Grun. | AMPERPUS | 2,144,236 | 1.40 | 28 |
| *Amphora submontana* Hust.[2] | AMSUBMON | 7,374 | < .01 | 2 |
| *Amphora veneta* Kütz.[2] | AMVENETA | 769 | < .01 | 1 |
| *Aulacosira ambigua* (Grun.) Simonsen[2] | AUAMBIG | 267 | < .01 | 1 |
| *Aulacosira granulata* (Ehr.) Simonsen[2] | AUGRANLT | 9,938 | < .01 | 1 |
| *Aulacosira granulata angustissima* (O. Müll.) Simonsen | AUGRNANG | 40,010 | .03 | 7 |
| *Bacillaria paradoxa* Gmelin in L. | BAPARDXA | 25,709 | .02 | 9 |
| *Biddulphia laevis* Ehr.[2] | BILAEVIS | 4,435 | < .01 | 2 |
| *Caloneis bacillum* (Grun.) Cl.[2] | CABACILL | 3,186 | < .01 | 3 |
| *Caloneis hyalina* Hust. | CAHYALIN | 96,369 | .06 | 17 |
| *Cocconeis diminuta* Pant.[2] | CCDIMINU | 896 | < .01 | 2 |
| *Cocconeis pediculus* Ehr. | CCPEDCLS | 99,000 | .07 | 15 |
| *Cocconeis placentula* Ehr. | CCPLACEN | 226,432 | .15 | 19 |
| *Cocconeis placentula euglypta* (Ehr.) Cl.[2] | CCPLAEUG | 7,810 | < .01 | 3 |
| *Cocconeis placentula lineata* (Ehr.) V. H. | CCPLALIN | 1,462,371 | .95 | 33 |
| *Cocconeis placentula pseudolineata* Geitler[2] | CCPLAPSE | 5,195 | < .01 | 2 |
| *Cyclotella atomus* Hust.[2] | CYATOMUS | 3,849 | < .01 | 4 |

**Table 4.** Total density, mean percent density, and frequency of occurrence of algal taxa collected in 36 New Jersey streams--Continued

| Group<br>*Genus species* | Taxa code | Total density (cells/cm$^2$) | Mean percent density | Frequency |
|---|---|---|---|---|
| *Cyclotella bodanica* Eulenst.[2] | CYBODANI | 2,393 | < .01 | 4 |
| *Cyclotella meneghiniana* Kütz. | CYMENEGH | 27,941 | .02 | 10 |
| *Cyclotella stelligera* (Cl. & Grun.) V. H.[2] | CYSTELLI | 7,176 | < .01 | 4 |
| *Cymbella affinis* Kütz. | CMAFFINS | 168,458 | .11 | 4 |
| *Cymbella cistula* (Ehr.) Kirchn.[2] | CMCISTUL | 667 | < .01 | 1 |
| *Cymbella delicatula* Kütz.[2] | CMDELCAT | 6,423 | < .01 | 2 |
| *Cymbella laevis* Naeg. ex Kütz.[2] | CMLAEVIS | 726 | < .01 | 1 |
| *Cymbella minuta* Hilse ex Rabh. | CMMINUTA | 463,023 | .30 | 22 |
| **Cymbella minuta pseudogracilis** (Choln.) Reim.[2] | CMMINPSE | 2,411 | < .01 | 1 |
| *Cymbella minuta silesiaca* (Bleisch ex Rabh.) Reim.[2] | CMMINSIL | 4,156 | < .01 | 1 |
| *Cymbella muelleri* Hust. | CMMUELLE | 1,408,802 | .92 | 1 |
| *Cymbella naviculiformis* Auersw. ex Héib.[2] | CMNAVICU | 4,155 | < .01 | 1 |
| *Cymbella prostrata* (Berk.) Cl.[2] | CMPROSTR | 769 | < .01 | 1 |
| *Cymbella sinuata* Greg. | CMSINUAT | 1,635,522 | 1.07 | 21 |
| *Cymbella tumida* (Bréb. ex Kütz.) V. H. | CMTUMIDA | 102,129 | .07 | 3 |
| *Diatoma vulgare* Bory | DAVULGAR | 53,463 | .04 | 6 |
| *Diploneis smithii dilatata* (M. Perag.) Boyer[2] | DPSMIDIL | 271 | < .01 | 1 |
| *Eunotia arcus* Ehr.[2] | EUARCUS | 2,829 | < .01 | 1 |
| *Eunotia exigua* (Bréb. ex Kütz.) Rabh.[2] | EUEXIGUA | 772 | < .01 | 1 |
| *Eunotia formica* Ehr.[2] | EUFORMIC | 9,747 | < .01 | 2 |
| *Eunotia implicata* Nörpel, Lange-Bert. & Alles | EUIMPLIC | 44,457 | .03 | 7 |
| *Eunotia incisa* W. Sm. ex Greg.[2] | EUINCISA | 7,619 | < .01 | 1 |
| *Eunotia paludosa* Grun.[2] | EUPALUDO | 111 | < .01 | 1 |
| *Eunotia pectinalis* (O. F. Müll.) Rabh.[2] | EUPECTIN | 498 | < .01 | 1 |
| *Eunotia pectinalis minor* (Kütz.) Rabh.[2] | EUPECMIN | 6,123 | < .01 | 2 |
| *Eunotia pectinalis undulata* (Ralfs) Rabh.[2] | EUPECUND | 4,137 | < .01 | 2 |
| *Fragilaria bicapitata* A. Mayer[1] | FRBICAPI | 1,078 | < .01 | 1 |
| *Fragilaria brevistriata inflata* (Pant.) Hust. | FRBREINF | 9,339 | .01 | 5 |
| *Fragilaria brevistriata* Lange-Bert. | FRBREVIS | 42,868 | .03 | 12 |
| *Fragilaria constricta* Ehr.[2] | FRCTASTR | 1,550 | < .01 | 1 |
| *Fragilaria construens* (Ehr.) Grun. | FRCONSTU | 36,660 | .02 | 13 |
| *Fragilaria construens pumila* Grun. | FRCONPUM | 28,299 | .01 | 8 |
| *Fragilaria construens venter* (Ehr). Grun. | FRCONVEN | 80,473 | .05 | 12 |
| *Fragilaria hungarica tumida* Cl.-Eul.[2] | FRHUNTUM | 166 | < .01 | 1 |
| *Fragilaria leptostauron* (Ehr.) Hust. | FRLEPTOS | 25,587 | .02 | 7 |
| *Fragilaria pinnata* Ehr. | FRPINNAT | 125,233 | .08 | 20 |
| *Fragilaria tenera* (W. Sm.) Lange-Bert[2] | FRTENERA | 20 | < .01 | 1 |
| *Fragilaria vaucheriae* (Kütz.) Peters. | FRVAUCHE | 87,862 | .06 | 18 |
| *Fragilaria virescens exigua* Grun. in Cl. et Möller[2] | FRVIREXI | 166 | < .01 | 1 |

**Table 4.** Total density, mean percent density, and frequency of occurrence of algal taxa collected in 36 New Jersey streams--Continued

| Group<br>*Genus species* | Taxa code | Total density (cells/cm$^2$) | Mean percent density | Frequency |
|---|---|---|---|---|
| *Fragilaria virescens* Ralfs[2] | FRVIRESC | 2,924 | < .01 | 1 |
| *Frustulia rhomboides* (Ehr.) DeT. | FSRHOMBO | 8,715 | .01 | 8 |
| *Frustulia* sp.1[2] | FS1? | 127 | < .01 | 1 |
| *Frustulia* spp.[2] | FSSPP | 132 | < .01 | 1 |
| *Frustulia vulgaris* (Thwaites) DeT. | FSVULGAR | 38,326 | .03 | 4 |
| *Gomphonema affine* Kütz.[2] | GOAFFINE | 769 | < .01 | 1 |
| *Gomphonema augur* Ehr.[2] | GOAUGER | 55 | < .01 | 1 |
| *Gomphonema clevei* Fricke | GOCLEVEI | 105,255 | .07 | 7 |
| *Gomphonema gracile* Ehr. emend. V. H.[2] | GOGRACIL | 8,433 | < .01 | 1 |
| *Gomphonema olivaceoides* Hust. | GOOLIVAC | 14,402 | .01 | 6 |
| *Gomphonema olivaceum* (Lyngb.) Kütz. | GOOLIVCM | 50,138 | .03 | 7 |
| *Gomphonema parvulum* (Kütz.) Kütz. | GOPARVUL | 715,404 | .47 | 33 |
| *Gomphonema pumilum* (Grun.) Reich. & Lange-Bert. | GOPUMILU | 757,153 | .49 | 29 |
| *Gomphonema truncatum capitatum* (Ehr.) Patr.[2] | GOTRUCAP | 2,049 | < .01 | 1 |
| *Gyrosigma attenuatum* (Kütz.) Rabh. | GYATTENU | 17,537 | .01 | 10 |
| *Gyrosigma scalproides* (Rabh.) Cl.[2] | GYSCALPD | 2,121 | < .01 | 2 |
| *Gyrosigma spencerii* (Quek.) Griff. & Henfr.[2] | GYSPENCE | 1,580 | < .01 | 3 |
| *Gyrosigma* spp.[2] | GYSPP | 396 | < .01 | 1 |
| *Gyrosigma wormleyi* (Sulliv.) Boyer[2] | GYWORMLY | 1,916 | < .01 | 1 |
| *Hannaea arcus amphioxys* (Rabh.) Patr. | HNARCAMP | 47,344 | .03 | 1 |
| *Hantzschia amphioxys* (Ehr.) Grun.[2] | HAAMPHIO | 2,094 | < .01 | 1 |
| *Melosira varians* Ag. | MEVARIAN | 97,990 | .06 | 10 |
| *Meridion circulare* (Grev.) Ag. | MDCIRCUL | 19,379 | .01 | 13 |
| *Navicula capitata* Ehr. | NACAPITA | 25,167 | .02 | 11 |
| *Navicula cincta* (Ehr.) Ralfs | NACINCTA | 26,573 | .02 | 1 |
| *Navicula cocconeiformis* Greg. ex Grev.[2] | NACOCCON | 2,934 | < .01 | 1 |
| *Navicula confervacea* (Kütz.) Grun. | NACONFER | 16,308 | .01 | 4 |
| *Navicula contenta* Grun. ex V. H. | NACONTEN | 13,319 | .01 | 5 |
| *Navicula cryptocephala* Kütz. | NACRYPTO | 257,876 | .17 | 20 |
| *Navicula cryptocephala veneta* (Kütz.) Rabh. [2] | NACRYVEN | 221 | < .01 | 1 |
| *Navicula cryptolyra* Brockman[2] | NACRYLY | 1,395 | < .01 | 1 |
| *Navicula decussis* Østr. | NADECUSS | 15,835 | .01 | 7 |
| *Navicula elginensis* (Greg.) Ralfs[2] | NAELGINS | 825 | < .01 | 2 |
| *Navicula gastrum* (Ehr.) Kütz.[2] | NAGASTRU | 1,539 | < .01 | 1 |
| *Navicula halophila* (Grun.) Cl.[2] | NAHALOPH | 10 | < .01 | 1 |
| *Navicula ignota palustris* (Hust.) Lund[2] | NAIGNPAL | 1,357 | < .01 | 1 |
| *Navicula ingenua* Hust.[2] | NAINGNUA | 13,706 | < .01 | 2 |
| *Navicula integra* (W. Sm.) Ralfs[2] | NAINTGRA | 3,896 | < .01 | 1 |
| *Navicula luzonensis* Hust. | NALUZONS | 42,477 | .03 | 10 |

**Table 4.** Total density, mean percent density, and frequency of occurrence of algal taxa collected in 36 New Jersey streams--Continued

| Group<br>*Genus species* | Taxa code | Total density (cells/cm$^2$) | Mean percent density | Frequency |
|---|---|---|---|---|
| *Navicula menisculus upsaliensis* (Grun.) Grun. [2] | NAMENUPS | 1,956 | < .01 | 1 |
| *Navicula minima* Grun. | NAMINIMA | 1,427,812 | .93 | 26 |
| *Navicula mutica* Kütz. | NAMUTICA | 647,884 | .42 | 4 |
| *Navicula mutica ventricosa* (Kütz.) Cl. & Grun.[2] | NAMUTVEN | 871 | < .01 | 1 |
| *Navicula perminuta* Grun.[2] | NAPERMNT | 609 | < .01 | 1 |
| *Navicula protracta* (Grun.) Cl.[2] | NAPROTRA | 111 | < .01 | 1 |
| *Navicula pseudoscutiformis* Hust.[2] | NAPSESCU | 55 | < .01 | 1 |
| *Navicula pupula capitata* Skv. & Meyer | NAPUPCAP | 139,551 | .09 | 8 |
| *Navicula pupula* Kütz. | NAPUPULA | 93,954 | .06 | 7 |
| *Navicula pupula mutata* (Krass.) Hust.[2] | NAPUPMUT | 8,238 | < .01 | 1 |
| *Navicula radiosa* Kütz. | NARADIOS | 59,819 | .04 | 9 |
| *Navicula radiosa tenella* (Bréb. ex Kütz.) Grun. | NARADTEN | 1,933,539 | 1.26 | 21 |
| *Navicula rhynchocephala amphiceros* (Kütz.) Grun. | NARHYAMP | 15,397 | .01 | 4 |
| *Navicula rhynchocephala germainii* (Wallace) Patr. | NARHYGER | 17,337 | .01 | 4 |
| *Navicula rhynchocephala* Kütz. | NARHYNCH | 45,008 | .03 | 11 |
| *Navicula salinarum* Grun. | NASALINM | 595,150 | .39 | 10 |
| *Navicula salinarum intermedia* (Grun.) Cl.[2] | NASALINT | 2,699 | < .01 | 3 |
| *Navicula schoenfeldii* Hust. | NASCHOEF | 17,276 | .01 | 4 |
| *Navicula secreta apiculata* Patr.[2] | NASECAPI | 11,541 | < .01 | 1 |
| *Navicula seminulum* Grun. | NASEMLUM | 563,437 | .37 | 17 |
| *Navicula subminuscula* Mang. | NASUBMIN | 49,396 | .03 | 1 |
| *Navicula submuralis* Hust.[2] | NASUBMUR | 11,836 | < .01 | 1 |
| *Navicula subtilissima* Cl.[2] | NASUBTIL | 8,025 | < .01 | 2 |
| *Navicula symmetrica* Patr.[2] | NASYMTRC | 707 | < .01 | 1 |
| *Navicula tenera* Hust.[2] | NATENERA | 12,503 | < .01 | 2 |
| *Navicula tripunctata* (O. F. Müll.) Bory | NATRIPUN | 300,293 | .20 | 9 |
| *Navicula viridula* (Kütz.) Kütz. emend. V. H. | NAVIRDLA | 178,215 | .12 | 18 |
| *Navicula viridula avenacea* (Breb. ex Grun.) V.H. | NAVIRAVE | 248,246 | .16 | 17 |
| *Neidium affine* (Ehr.) Pfitz.[2] | NEAFFINE | 707 | < .01 | 1 |
| *Nitzschia amphibia* Grun. | NIAMPHIB | 2,304,920 | 1.50 | 31 |
| *Nitzschia commutata* Grun. [2] | NICOMTAT | 769 | < .01 | 1 |
| *Nitzschia constricta* (Kütz.) Ralfs[2] | NICONSTR | 7,514 | < .01 | 2 |
| *Nitzschia dissipata* (Kütz.) Grun. | NIDISSIP | 894,334 | .58 | 13 |
| *Nitzschia fonticola* Grun. | NIFONTIC | 69,746 | .05 | 7 |
| *Nitzschia frustulum* (Kütz.) Grun. | NIFRUSTU | 25,127 | .02 | 6 |
| *Nitzschia hungarica* Grun.[2] | NIHUNGRC | 2,906 | < .01 | 1 |
| *Nitzschia inconspicua* Grun. | NIINCONS | 3,038,614 | 1.98 | 30 |
| *Nitzschia linearis* (Ag. ex W. Sm.) W. Sm. | NILINEAR | 22,920 | .02 | 7 |
| *Nitzschia palea* (Kütz.) W. Sm. | NIPALEA | 1,273,398 | .83 | 20 |

**Table 4.** Total density, mean percent density, and frequency of occurrence of algal taxa collected in 36 New Jersey streams--Continued

| **Group**<br>*Genus species* | **Taxa code** | **Total density (cells/cm²)** | **Mean percent density** | **Frequency** |
|---|---|---|---|---|
| *Nitzschia parvula* W. Sm.[2] | NIPARVUL | 119 | < .01 | 1 |
| *Nitzschia recta* Hantz. ex Rabh.[2] | NIRECTA | 332 | < .01 | 1 |
| *Nitzschia sigmoidea* (Nitz.) W. Sm.[2] | NISIGDEA | 2,906 | < .01 | 1 |
| *Nitzschia sinuata delognei* (Grun.) Lange-Bert.[2] | NI SINDE | 769 | < .01 | 1 |
| *Nitzschia tryblionella debilis* (Arnott) Hust.[2] | NITRYDEB | 3,831 | < .01 | 1 |
| *Nitzschia tryblionella levidensis* (W. Sm.) Grun. in Cl. et Grun.[2] | NITRYLEV | 1,916 | < .01 | 1 |
| *Opephora olsenii* M Moller[2] | OPOLSENI | 2,308 | < .01 | 1 |
| *Pinnularia microstauron* (Ehr.) Cl.[2] | PIMICROS | 7,255 | < .01 | 1 |
| *Pinnularia obscura* Krass.[2] | PIOBSCUR | 20 | < .01 | 1 |
| *Pinnularia subcapitata* Greg. | PISUBCAP | 3,407 | .01 | 5 |
| *Pinnularia sudetica* (Hilse) M. Perag.[2] | PISUDETI | 707 | < .01 | 1 |
| *Pinnularia viridis* (Nitz.) Ehr.[2] | PIVIRIDI | 4,633 | < .01 | 2 |
| *Reimeria sinuata* (Greg.) Kociolek & Stoermer | RESINUTA | 431,361 | .28 | 12 |
| *Rhoicosphenia curvata* (Kütz.) Grun. ex Rabh. | ROCURVAT | 2,689,622 | 1.75 | 33 |
| *Stauroneis anceps* Ehr.[2] | SSANCEPS | 859 | < .01 | 1 |
| *Stauroneis kriegeri* Patr.[2] | SSKRIEGE | 5,814 | < .01 | 1 |
| *Stauroneis smithii* Grun.[2] | SSSMITHI | 845 | < .01 | 3 |
| *Stauroneis* spp.[2] | SSSPP | 260 | < .01 | 1 |
| *Stephanodiscus niagarae* Ehr.[2] | STNIAGAR | 1,395 | < .01 | 1 |
| *Surirella amphioxys* W. Sm.[2] | SUAMPOXY | 2,215 | < .01 | 3 |
| *Surirella angusta* Kütz. | SUANGUST | 29,791 | .02 | 2 |
| *Surirella ovalis* Bréb.[2] | SUOVALIS | 3,933 | < .01 | 3 |
| *Synedra incisa* Boyer[2] | SYINCISA | 520 | < .01 | 1 |
| *Synedra pulchella* Ralfs ex Kütz.[2] | SYPULCHE | 8,010 | < .01 | 2 |
| *Synedra rumpens* Kütz. | SYRUMPEN | 113,070 | .07 | 10 |
| *Synedra ulna* (Nitz.) Ehr. | SYULNA | 107,776 | .07 | 20 |
| *Synedra ulna contracta* Østr.[2] | SYULNCNT | 12,751 | < .01 | 3 |
| *Tabellaria flocculosa* (Roth) Kütz.[2] | TAFLOCCU | 5,992 | < .01 | 1 |
| *Thalassiosira pseudonana* Hasle & Heimdal[2] | THPSNANA | 1,539 | < .01 | 1 |
| *Thalassiosira weissflogii* (Grun.) Fryxell & Hasle[2] | THWEISS | 55 | < .01 | 1 |
| **Rhodophycophyta (Red algae)** | | | | |
| *Audouinella violacea* Kütz. | AUDVIO_R | 1,684,745 | 1.10 | 12 |

[1] Taxa were downweighted during ordination analysis.

[2] Taxa did not meet selection criteria and were excluded from ordination analysis.

34, 28, and 29 of the 36 sites sampled, respectively. Chimarra sp. (Family: Philopotamidae) accounted for 3.6 percent of the benthic community. These taxa together accounted for more than 66 percent of the mean abundance.

The green alga Protoderma viride (37.6 percent) and the blue-green alga Calothrix sp. (28.8 percent) were the most abundant algal taxa (table 4). These two taxa accounted for more than 66 percent of the mean numerical density of all algae sampled. Because of their ubiquitous distribution and high abundance (44-57 million cells/cm$^2$), however, these two species were downweighted or made passive in all ordination analyses. Although blue-green and green algae were numerically the most abundant taxa, greater than 81 percent of the taxa reported were diatoms. Achnanthes deflexa, the most abundant diatom, accounted for 4.4 percent of the mean abundance and was found at 28 of the 36 sites.

## Spatial and Temporal Variability in Aquatic Assemblages

Similarity among fish communities ranged from 71 to 91 percent, indicating very low spatial (reach-to-reach) and temporal (year-to-year) variability (table 5). Percent similarity among reaches and years was high and averaged 82 and 83 percent, respectively. Similarity for invertebrate-species composition was lower and ranged from 60 to 74 percent (table 6). Average percent similarity was slightly higher among multiple reaches (66 percent) than among multiple years (63 percent) (table 8). Sorenson's percent similarity index was not calculated for algal taxa because data for the third year of algal sampling were unavailable at the time of analysis; however, results of TWINSPAN classification analysis (not shown) verified a high level of similarity among multiple-reach and multiple-year sites for both invertebrate and algal communities. Close clustering of multiple reaches and multiple years for invertebrate and algal communities indicated that variability among reaches and years was significantly less than variability among the 36 stream sites.

Results of these analyses indicate a high degree of similarity among communities in multiple reaches sampled during 1998 and among communities in the same reach sampled for 3 consecutive years (1996-98), indicating that the population in a single reach in an individual year likely is representative of stream populations. These results indicate that comparisons among sites likely were valid for individual reaches. As a result, data derived from samples collected from the 36 stream reaches in 1996 were assumed to be representative of those derived from samples collected over the 3-year period and, consequently, were used in the remainder of the analyses. Bilger and Brightbill (1998), Frenzel and Swanson (1996), and Waite and Carpenter (2000) used an analogous approach and came to a similar conclusion.

## Distinctions Among Benchmark Community Groups

Indirect ordination and classification analysis of the 43 least impaired sites in New Jersey indicated that the northern New Jersey and Coastal Plain invertebrate communities differ substantially (fig. 8). Naturally lower pH and concentrations of dissolved solids and nutrients and a lack of rock and cobble substrate in the Coastal Plain streams are the primary environmental and physical factors responsible for these differences. In addition, these results indicate that two of the sites designated as benchmark sites are highly dissimilar from the others and are separated from the northern New Jersey and Coastal Plain groups in ordination space (fig. 8). For example, AN0025 (Paulins Kill at Blairstown, N.J.) and AN0224 (Passaic River near Millington, N.J.) are moderately impaired and are dominated by Hydropsychidae and Amphipoda, which are considered to be indicators of an anthropogenic disturbance. The northern New Jersey group (fig. 8) is characterized by many disturbance-intolerant invertebrate taxa such as Ephemerellidae, Amphinemouridae, Philopotamidae, Simuliidae, and Diamesinae. The Coastal Plain group is characterized by Leuctridae, Lepidostomatidae, Chironomidae (for example, Tribelos, Conchapelopia), Simuliidae, Limnodrilus, and Bivalvia. TWINSPAN analysis was used to verify our ordination results. Results of this analysis also identified two major groups of sites, northern New Jersey and Coastal Plain. Jackknifing analysis of

**Table 5.** Sorenson's percent similarity index of fish-species composition among New Jersey stream reaches (A, B, C) during a single year (1997) and among years (1996-98) for a single reach
[—, no multiple-reach samples collected; comparison is derived from single-pass data of fish abundance; station names are listed in table 1]

| Station code | Reaches | | | Sampling period | | |
|---|---|---|---|---|---|---|
| | A–B | A–C | B–C | 1996–97 | 1996–98 | 1997–98 |
| Nesh96A | 82.9 | 80.0 | 85.7 | 88.9 | 75.0 | 78.9 |
| Bnd96 | — | — | — | 71.4 | 81.8 | 77.8 |
| Sad96A | 73.3 | 88.2 | 81.3 | 85.7 | 75.9 | 84.8 |
| Stny96A | — | — | — | 90.5 | 91.3 | 87.0 |

**Table 6.** Sorenson's percent similarity index of invertebrate-species composition among New Jersey stream reaches (A, B, C) during a single year (1997) and among years (1996-98) for a single reach
[—, no multiple-reach samples collected; station names are listed in table 1]

| Station code | Reaches | | | Sampling period | | |
|---|---|---|---|---|---|---|
| | A–B | A–C | B–C | 1996–97 | 1996–98 | 1997–98 |
| Nesh96A | 74.1 | 64.4 | 58.5 | 67.3 | 60.4 | 61.2 |
| Bnd96 | — | — | — | 63.6 | 64.4 | 60.8 |
| Sad96A | 61.7 | 63.3 | 72.7 | 61.7 | 63.3 | 61.2 |
| Stny96A | — | — | — | 67.3 | 63.4 | 63.0 |
| Rar96 | — | — | — | 62.7 | 65.5 | 63.4 |

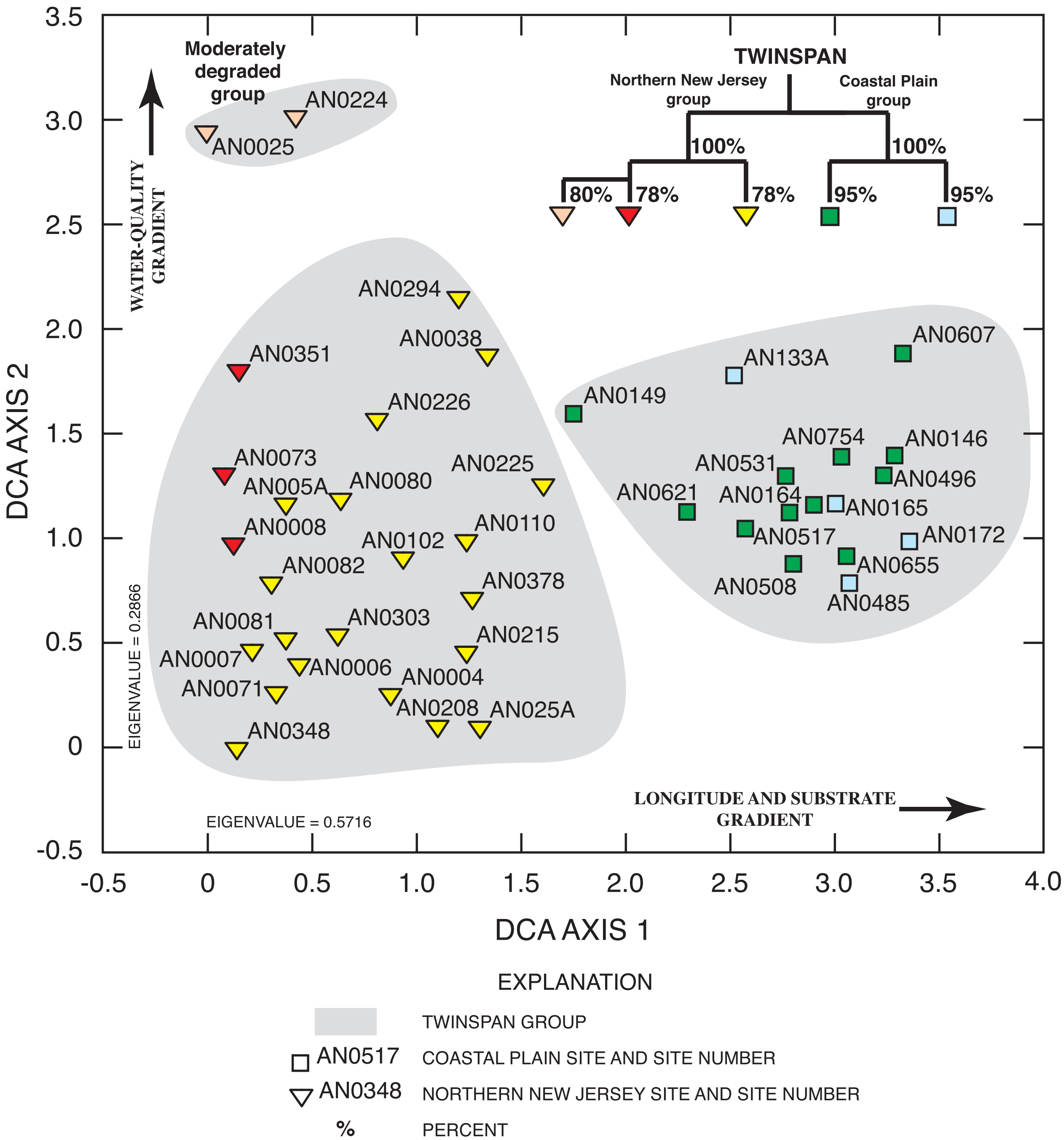


**Figure 8.** Detrended correspondence analysis (DCA) ordination of 43 invertebrate-community benchmark sampling sites in the New Jersey Department of Environmental Protection Ambient Biomonitoring Network (AMNET). (Sites described in New Jersey Department of Environmental Protection (1994b). Axis 1 expresses a longitudinal and substrate gradient that separates the sites into northern New Jersey and Coastal Plain groups. Axis 2 expresses a water-quality gradient. Dendrogram at top right shows classification of 43 New Jersey AMNET sites into community types on the basis of results of Two-Way INdicator SPecies ANalysis (TWINSPAN); numbers at each node are percent persistence of clusters based on results of a jackknifing (sensitivity) analysis.)

the TWINSPAN clusters (that is, a sensitivity analysis) indicated a 100-percent persistence at the northern New Jersey and Coastal Plain divisions (fig. 8). These results verify a clear separation between invertebrate communities in the northern and southern parts of the State. Differences in these community types are represented in the DCA by gradients of substrate, pH, conductivity, and longitude (first axis) and a moderate water-quality gradient (second axis), indicating that natural variability in community composition needs to be considered when assessing communities across a large study area. As a result of these findings, subsequent data-collection efforts were focused on only one of the areas—the northern New Jersey biological communities (fig. 1). One Coastal Plain site (GrtEgg) was retained because of its importance as an urban indicator site for National data aggregation conducted as part of the NAWQA Program; however, it was made passive in all ordination analyses and, therefore, had no effect on the distribution of sites in ordination space.

## Identification of Important Environmental Variables

PCA and correlation analysis were used to reduce the number of environmental variables in the data set from 413 to 71. Only those variables that loaded highest on significant principal components for each variable group were retained for use in multiple linear regression analysis; these included three hydrologic; eight socioeconomic; six distance, slope, and soil; three forested buffer; eight nutrient; four bed-sediment organic; five bed-sediment trace element; three pesticide and VOC; eight habitat; four water and wetland land use; five point source and impervious surface area; eight urban land use; three forest and undeveloped; and three agricultural land-use variables. Spearman rank correlation analysis was used to help reduce multicollinearity among the environmental variables. The advantage of multivariate analysis is its capability to analyze data matrices and determine multiple patterns of variation in the data set, reduce redundancy among similar variables, and iteratively reevaluate composited data to clarify patterns (Jongman and others, 1995). PCA was used in combination with MLR to help determine the restricted group of environmental variables that were responsible for the greatest amount of variability in the fish-, invertebrate-, and algal-community data sets.

## Indirect Gradient Assessment of Aquatic Assemblages

DCA of fish, invertebrate, and algal communities reflected strong first-axis gradients (fig. 9). The urban gradient (axis 1) indicates increasing biological integrity with decreasing urbanization and is reflected in large eigenvalues. Eigenvalues are a measure of the relative strength of an ordination axis (Jongman and others, 1995) and the first-axis eigenvalues for all aquatic communities were 0.539 or greater (table 7). Gradient length was 4.66 for the fish community, 4.70 for the invertebrate community, and 3.46 for the algal community (table 7). Eigenvalues for the first through fourth axes ranged from 0.611 to 0.085. The first through fourth axes explained 23.2 percent of the variance in fish community structure. Similarly, 26.9 and 34.9 percent of the variance in the invertebrate and algal communities, respectively, was explained by axes 1 through 4. For each of the aquatic communities, more than one-third of the total variance in the community data was accounted for by the first DCA axis (the hypothesized urban gradient).

The three aquatic community DCA's reflect similar patterns. Elizabet, the most densely urban and most biologically impaired site evaluated in this study, was farthest to the left for the fish and invertebrate assemblages and farthest to the right for the algal assemblage (fig. 9). FlatBrk is farthest to the right along the first axis of the fish DCA and is recognized as one of New Jersey's benchmark sites (N.J. Department of Environmental Protection, 1994b). The FlatBrk drainage basin also has the least urban land use (2.75 percent; table 1). PaulBlr (10 percent urban land use) and MuscHak (19 percent urban land use) are extremes on both the invertebrate and algal DCA's. These two sites are relatively unimpaired and have minimal urban development, and their drainage basins contain less than 10 percent total impervious surface cover. Elizabet appears to be an outlier and could be removed from the analysis; if Elizabet were removed, however, Rahway, another highly impaired site, stands apart from the group and the

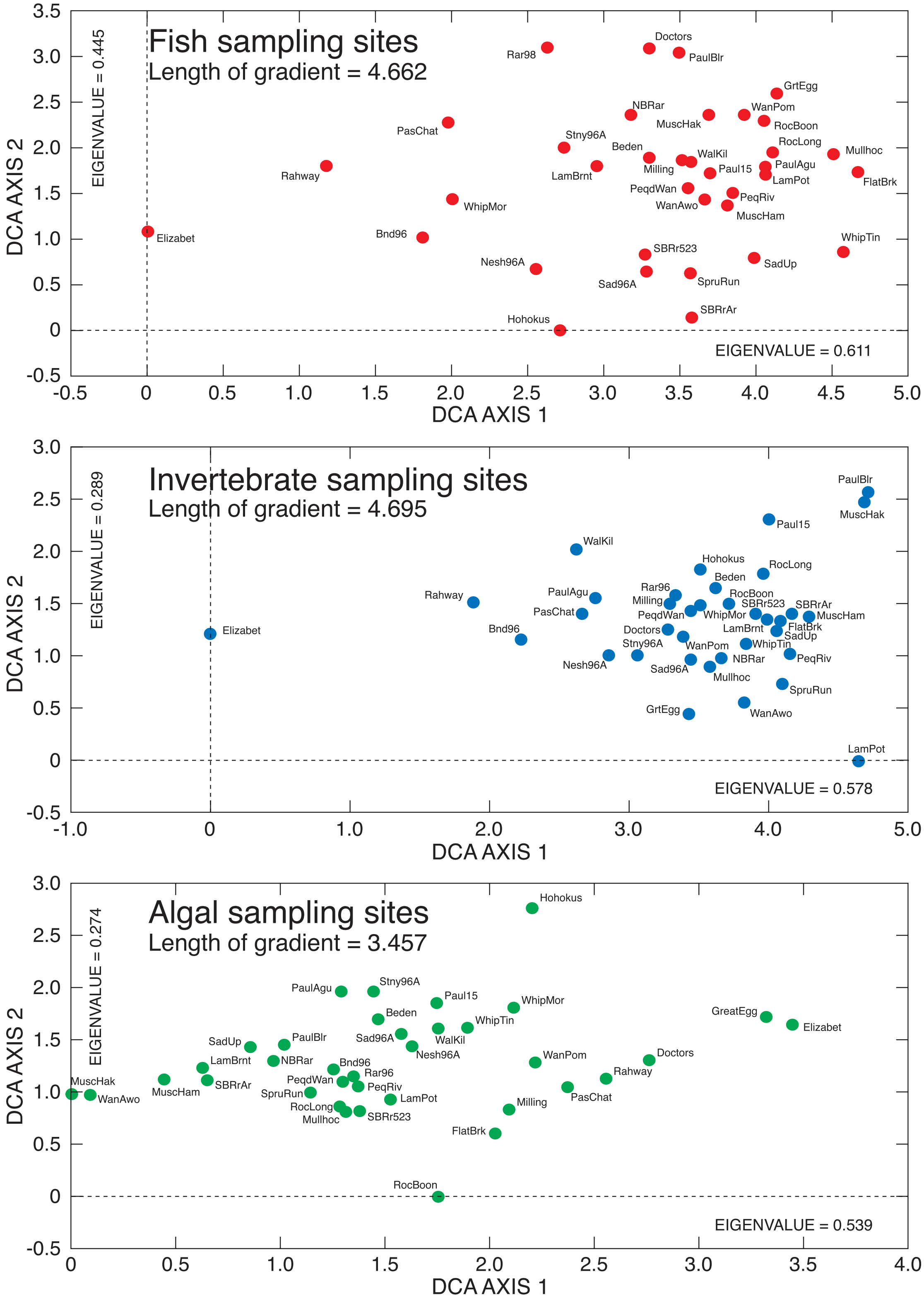


**Figure 9.** Detrended correspondence analysis (DCA) of relativized abundances of fish, invertebrate, and algal communities in New Jersey streams. (The distribution of sites along axis 1 represents the hypothesized urban land-use gradient (ULUG); extracted axis site scores were used to build multiple linear regression models representing the relation between the ULUG and important environmental variables. The ULUG is from left (most disturbed) to right (least disturbed) for the fish and invertebrate communities and from right to left for the algal community.)

**Table 7.** Gradient length and first- through fourth-axis eigenvalues for fish, invertebrate, and algae detrended correspondence analysis (DCA) ordinations for New Jersey streams

| Statistic | Fish | Invertebrate | Algae |
|---|---|---|---|
| Length of gradient | 4.662 | 4.695 | 3.457 |
| Axis 1 eigenvalue | .611 | .578 | .539 |
| Axis 2 eigenvalue | .445 | .289 | .274 |
| Axis 3 eigenvalue | .320 | .215 | .168 |
| Axis 4 eigenvalue | .264 | .141 | .085 |

**Table 8.** Results of multiple linear regression models relating first- through fourth-axis scores extracted from detrended correspondence analysis (DCA) and aquatic community indices to environmental variables that best describe the variation in assemblage structure for New Jersey streams

[IBI, index of biotic integrity; NJIS, New Jersey impairment score; TDI, tolerant diatom index; all models describing these relations were significant at the p < 0.02 level]

| Statistic | | Fish | | | | |
|---|---|---|---|---|---|---|
| DCA axis or index | | 1 | 2 | 3 | 4 | IBI |
| | Model $r^2$ | 0.91 | 0.53 | 0.31 | 0.17 | 0.84 |
| | p-value | .0001 | .0001 | .0023 | .0131 | .0001 |
| | | Invertebrate | | | | |
| DCA axis or index | | 1 | 2 | 3 | 4 | NJIS |
| | Model $r^2$ | .75 | .50 | .74 | .46 | .75 |
| | p-value | .0001 | .0001 | .0001 | .0001 | .0001 |
| | | **Algae** | | | | |
| DCA axis or index | | 1 | 2 | 3 | 4 | TDI |
| | Model $r^2$ | .49 | .65 | .42 | .33 | .62 |
| | p-value | .0001 | .0001 | .0002 | .0014 | .0001 |

long gradient remains. Although sequential removal of the most impaired site along the first DCA axis was repeated multiple times, the long gradient was maintained after each deletion. This result indicates that an outlying group or site represents a pattern that likely is caused by a distinct anthropogenic process.

## Significant Environmental Factors

MLR models that describe the relation between first-axis scores (that is, the hypothesized urban gradient) and the environmental variables modeled were highly significant (table 8). Twenty-eight environmental variables were found to be significantly related to extracted DCA axis scores for fish, invertebrate, and algal communities and their associated indices (table 9). Increasing fish-, invertebrate-, and algal-community impairment was statistically related to components of the urban gradient—in particular, to population density, impervious surfaces and wastewater discharges, and changes in flow and nutrients resulting from anthropogenic activity, respectively. Environmental factors, such as the periodicity of flooding (2YRPK), amount of road impervious area (IR), and population density (POPDEN, PD, and PDD3), were significantly related to impairment in all three aquatic communities (table 9). Some environmental factors, such as point-source flow (PTOTSPS), area of urban land in a basin in 1995 (U95), and non-road impervious area (IND1), were related to impairment in the aquatic invertebrate community only. In addition, total urban area in 1986 (U86D3) appeared to be important only for the longer lived fish community. The presence of cobble substrate (PERCO) was significantly related to non-impaired fish and aquatic-invertebrate communities, and the area of forest and wetland located near a sampling site (FWD1) was positively related to the non-impaired aquatic-invertebrate community (table 9).

MLR models were developed for each of the first four DCA ordination axes for fish, invertebrates, and algae. All models were highly significant ($p<0.002$) and accounted for 17 to 91 percent of the variation in species data (table 8). Seven environmental variables were highly correlated with the first-axis scores (the hypothesized urban gradient) of the fish-community data and accounted for 91 percent of the variance. Three variables accounted for 75 and 49 percent of the variance along the first axes for the invertebrate- and algal-community data, respectively (table 8). Variables significantly related to first-axis scores of the fish community were 2YRPK, IR, percent dominant substrate (PERDS), median total phosphorus concentration (MTP), index of stream flashiness (RT2575), area of water in the basin (WATD3), and median potassium concentration (MK). Strongest relations for the first-axis scores of the invertebrate community were population density near the site (PDD3), PERCO, and median dissolved sulfate concentration (MS04). For the algal community, 2YRPK, IR, and median dissolved organic carbon concentration (MDOC) were significantly related to the first-axis scores.

The second axis of the fish community was highly correlated with the area of urban land in a basin in 1986 (U86D3), mean dominant bank substrate (MBDS), housing value (HV), and MK, which is a surrogate for chemical use. For invertebrates, U95 and median specific conductance (MSC) were driving the second axis. The second axis of the algal community was correlated significantly with total 15-m stream buffer area in a watershed (S15AREA), population density (PD), area of commercial plus industrial urban land in a basin in 1986 (U12D1), MSO4, and suspended organic carbon concentration (SOC) (table 9).

The third-axis MLR model of the fish community was correlated highly with DOC concentration and mean stream slope (SSLP). Invertebrate third-axis scores were significantly correlated with U12D1, 2YRPK, PTOTSPS, HV, and the frequency of pesticides found in the water column (NHITSP). The algal third-axis scores were related to mean canopy opening (MCO) and MTP. The MLR model of the fourth-axis scores of the fish community was significantly related ($p< 0.013$) to a single environmental variable (2YRPK). The fourth-axis model of the invertebrate data was related significantly to U95, mean soil depth (SD), and non-road impervious area (IND1) near the site. The fourth algal axis was related to PERDS and MBDS. In general, the strength of the model decreases from the first- to

**Table 9.** Description of environmental variables significantly related to detrended correspondence analysis (DCA) axis scores or community indices in New Jersey

[dw, variable is distance-weighted to account for proximity of a source to the stream channel. The lower the number, the closer a source is to the stream—for example, dw1 is the distance weighting for a source that is nearest to the stream; sw, variable is slope-weighted to account for landform and typology; Wolman count, a count of 100 substrate particles in a cross section of stream; GIS, variable determined by use of geographic information system; SSURGO, New Jersey soil-classification data (U.S. Department of Agriculture, 1995); cm, centimeters; m, meters; km, kilometers; $km^2$, square kilometers; $mi^2$, square miles; $ft^3/s$, cubic feet per second; µg/L, micrograms per liter; mg/L, milligrams per liter; mS/cm, microsiemens per centimeter at 25 degrees Celsius; Mgal/yr, million gallons per year]

| Variable name | Explanatory environmental variable (Description, units of measurement, and source of data) | Minimum | Median | Maximum |
|---|---|---|---|---|
| | **Land-use characteristics** | | | |
| S15F | Total forest in the 15-m stream buffer area ($km^2$): GIS [8] | 15.21 | 64.05 | 83.60 |
| IR | Impervious area–roads ($km^2$): GIS [7, 8] | 137.71 | 284.47 | 1,350.13 |
| IND1 | Impervious area–non-roads ($km^2$)–dw1: GIS and regression [5] | .12 | 7.69 | 153.02 |
| WATD3 | Area of water in a basin ($km^2$)–dw3: GIS [4] | .28 | 4.66 | 102.40 |
| U86D3 | Area of urban land in a basin in 1996 ($km^2$)–dw3: GIS [4] | .25 | 6.61 | 55.75 |
| U95[1] | Area of urban land in a basin in 1995 ($km^2$): GIS [5] | 1.62 | 5.03 | 9.33 |
| U12D1 | Area of commercial plus industrial urban land in a basin in 1986 ($km^2$)–dw1: GIS [5, 6] | .00 | .04 | 1.51 |
| FWD1 | Area of forest and wetlands in a basin ($km^2$)–dw1: GIS [5, 8] | .01 | .38 | 6.25 |
| S15AREA | Total 15-m stream buffer area in a watershed ($km^2$): GIS [6] | .59 | 3.33 | 64.14 |
| | **Socioeconomic characteristics** | | | |
| POPDEN | Population density (persons/$km^2$): GIS [7, 8] | 310.80 | 1143.48 | 37,593.85 |
| PD | Population density (persons/900 $mi^2$): GIS [6] | .44 | 1.31 | 8.06 |
| PDD3 | Population density (persons/900 $mi^2$)–dw3: GIS [5] | .06 | .29 | 4.94 |
| HV[2] | Mean housing value (thousands of dollars): GIS [4, 5] | 4.52 | 5.37 | 5.97 |
| | **Physical characteristics** | | | |
| PERCO | Cobble substrate in the sampling area (percent): Wolman count [5, 8] | .00 | 51.00 | 87.00 |
| PERDS | Dominant substrate in the sampling area (percent): Wolman count [4, 6] | 50.00 | 68.00 | 99.00 |
| MBDS | Mean bank dominant substrate (percent): average score for a minimum of 12 observations [4, 6] | 1.00 | 4.50 | 8.00 |
| SSLP | Mean stream slope (m/km): GIS [4] | 3.00 | 18.30 | 32.58 |
| SD | Mean soil depth (cm): GIS–SSURGO [5] | 67.54 | 96.80 | 152.40 |
| SDD3[2] | Mean soil depth (cm)–dw3: GIS–SSURGO [8] | 1.50 | 6.00 | 9.17 |
| MCO | Median canopy opening (degrees): angular measurements made with a clinometer [6] | 42.50 | 152.50 | 180.00 |
| | **Hydrologic characteristics** | | | |
| RT2575[2] | Ratio of discharge ($ft^3/s$) exceeded for the indicated percentage of time: (Reed and others, 1997) [4] | .60 | 1.26 | 2.81 |
| 2YRPK[2] | 2-year peak flow ($ft^3/s$): (Reed and others, 1997) [7] | 4.61 | 6.98 | 9.10 |
| 10YRLOW[2] | 7-day 10-year low flow ($ft^3/s$): (Reed and others, 1997) [8] | -2.04 | 1.34 | 4.36 |

| Variable name | Explanatory environmental variable (Description, units of measurement, and source of data) | Minimum | Median | Maximum |
|---|---|---|---|---|
| | **Water-column chemistry** | | | |
| MTP | Median total phosphorus concentration (mg/L as P): (Reed and others, 1997) [4, 6, 8] | .01 | .04 | .29 |
| MSOC | Median suspended organic carbon concentration (mg/L as C): (Reed and others, 1997) [6, 9] | .10 | .40 | 1.10 |
| MDOC | Median dissolved organic carbon concentration (mg/L as C): (Reed and others, 1997) [4, 6] | 1.50 | 2.85 | 9.40 |
| MK | Median potassium concentration (mg/L as K): (Reed and others, 1997) [4] | .54 | 1.50 | 3.10 |
| MSC | Median specific conductance (mS/cm): instantaneous field measurements at sampling site [5] | 73.00 | 257.25 | 678.00 |
| MSO4 | Median dissolved sulfate concentration (mg/L): (Reed and others, 1997) [5, 6, 8] | 6.80 | 16.00 | 40.00 |
| PTOTSPS[3] | Total point-source flow (Mgal/yr) dw * sw [5] | .30 | 1.34 | 3.50 |
| NHITSP | Total number of pesticide detections in a sample: (Reiser and O'Brien, 1999) [5, 9] | 2.00 | 7.00 | 14.00 |
| ATRA | Concentration of atrazine (mg/L) detected in a sample: (Reiser and O'Brien, 1999) [9] | .00 | .01 | 1.44 |

**Table 9.** Description of environmental variables significantly related to detrended correspondence analysis (DCA) axis scores or community indices in New Jersey--Continued

1 Square-root transformation
2 $Log_e$ transformation
3 $Log_{10}$ (x+2) transformation
4 Significantly related to fish-community structure
5 Significantly related to invertebrate-community structure
6 Significantly related to algal-community structure
7 Significantly related to the structure of all three communities
8 Significantly related to community indices
9 Significantly related to median potassium concentration

the fourth-axis model. This outcome is expected because the extracted DCA axis scores reflect successively weaker environmental gradients from the first to the fourth axis.

MLR models of the biotic indices IBI, NJIS, and TDI were highly significant ($p<0.0001$) and accounted for 84, 75, and 62 percent of the variation in metric data, respectively (table 8). The index of biotic integrity for the fish community was negatively related to POPDEN, MTP, and MS04, and positively related to 15-m buffer (S15F), PERCO, MSC, and RT2575. NJIS was negatively related to IR and positively related to 10YRLOW (equates to presence of base flow), PERCO, and FWD1. TDI was positively correlated with chemical use (MSO4, MTP) and mean soil depth near the sampling site (SDD3). Only three environmental variables (10YRLOW, SDD3, and S15F), in addition to those identified during the MLR analyses of the extracted first through fourth DCA axis scores, were related significantly only to biotic indices (table 3). In addition to the 31 significant environmental variables identified in the MLR models, the concentration of atrazine detected in a sample (ATRA) was found to be related to MK and was retained for use in the pCCA assessment.

Disturbance thresholds for fish, invertebrate, and algal communities were evaluated by regressing individual community indices on percent ISC. A change in slope occurred between 20 and 25 percent ISC for the invertebrate community (fig. 10). Although somewhat less steep, a change in slope also occurred for the fish community, but between 18 and 24 percent (fig. 10). In both cases, as ISC increased to near 20 percent, IBI and NJIS declined, indicating an increase in community impairment. The TDI for the algal community did not appear to provide as good a correlation with ISC, and the degree of scatter in the data was greater than that for the IBI or NJIS. Interpretation of the relation between TDI and impervious surface was accomplished by excluding some sites with a much higher TDI value than would be determined from ISC. With these exclusions, a disturbance threshold appeared to occur between 16 and 24 percent ISC (fig. 10).

## Environmental Associations Across Multiple pCCA Axes

Partial canonical correspondence analysis (pCCA) was used to simultaneously evaluate species/environment relations and eliminate the effects of those covariables that express natural variability. Results of pCCA of fish, invertebrates, and algal species composition and environmental variables are presented in figures 11, 12, and 13. Results of the Monte Carlo global permutation tests of significance (399 permutations under the reduced model) for the first canonical axis ($p<0.021$) and the sum of all canonical eigenvalues ($p<0.006$) were significant for all communities assessed (table 10). The forward-selection procedure resulted in the retention of 9, 10, and 9 significant environmental variables for fish, invertebrates, and algal communities, respectively (tables 11, 12, and 13). The variance associated with drainage area, altitude, and longitude was partialed out of the fish and invertebrate direct-gradient analyses. In addition, the variance associated with sampling year was partialed out of the fish analysis. Only the variances associated with drainage area and altitude were partialed out of the algal community analysis. Longitude did not appear to have an appreciable effect on the distribution of algal taxa and sampling sites, and, subsequently, was eliminated as a covariable in the algal community direct-gradient analysis.

The first axis of the fish pCCA appears to indicate a strong relation with urban effects (population density and road impervious surface), substrate composition, and stream slope and density (table 11, fig. 11). This axis clearly separates the most urban sites (for example, Elizabet) on the far right from the less urbanized sites on the left (for example, Mullhoc and FlatBrk). The second axis is most strongly related to nutrient concentration and non-road impervious surface. Sites such as Rahway and Paul Agu appear to have high levels of nutrients and impervious surfaces, whereas Sad96A has less impervious surface area because it is located in a less densely developed suburban setting. The third axis is related to a combination of urban effects such as population density and changes in substrate composition. The fourth axis also shows some urban effect, such as high median

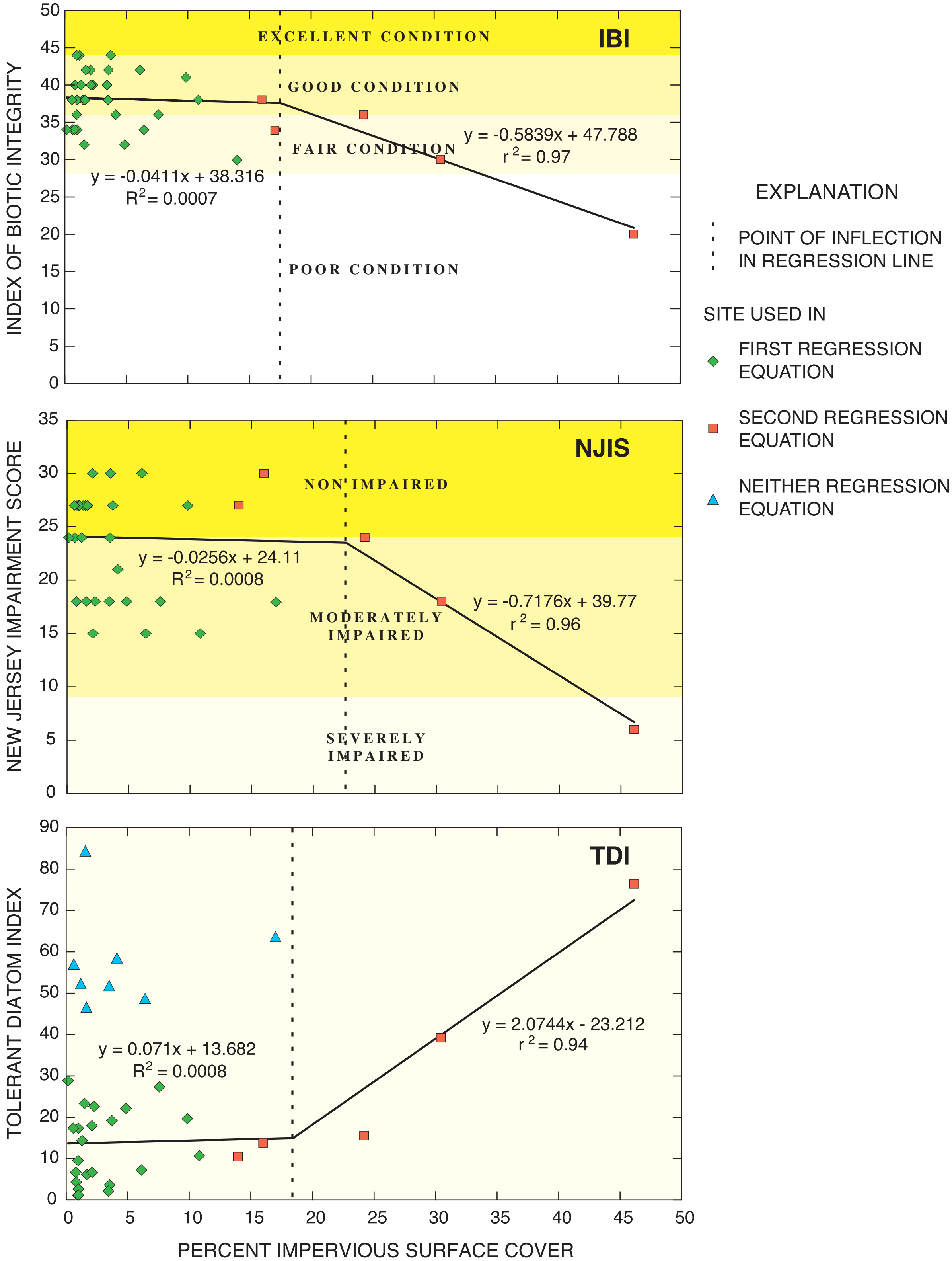


**Figure 10.** Regression relations of the index of biotic integrity (IBI), New Jersey impairment score (NJIS), and tolerant diatom index (TDI) with percent impervious surface cover. (Lower values of IBI and NJIS reflect poorer biotic conditions, whereas higher TDI values reflect poorer conditions. The dotted vertical lines represent the approximate inflection points (thresholds) beyond which streams in New Jersey become imparied. The horizontal color shades for IBI and NJIS reflect levels of condition and impairment, respectively, previously derived for New Jersey streams (Kurtenbach, 1993; N.J. Department of Environmental Protection, 1994a; Plafkin and others, 1989)

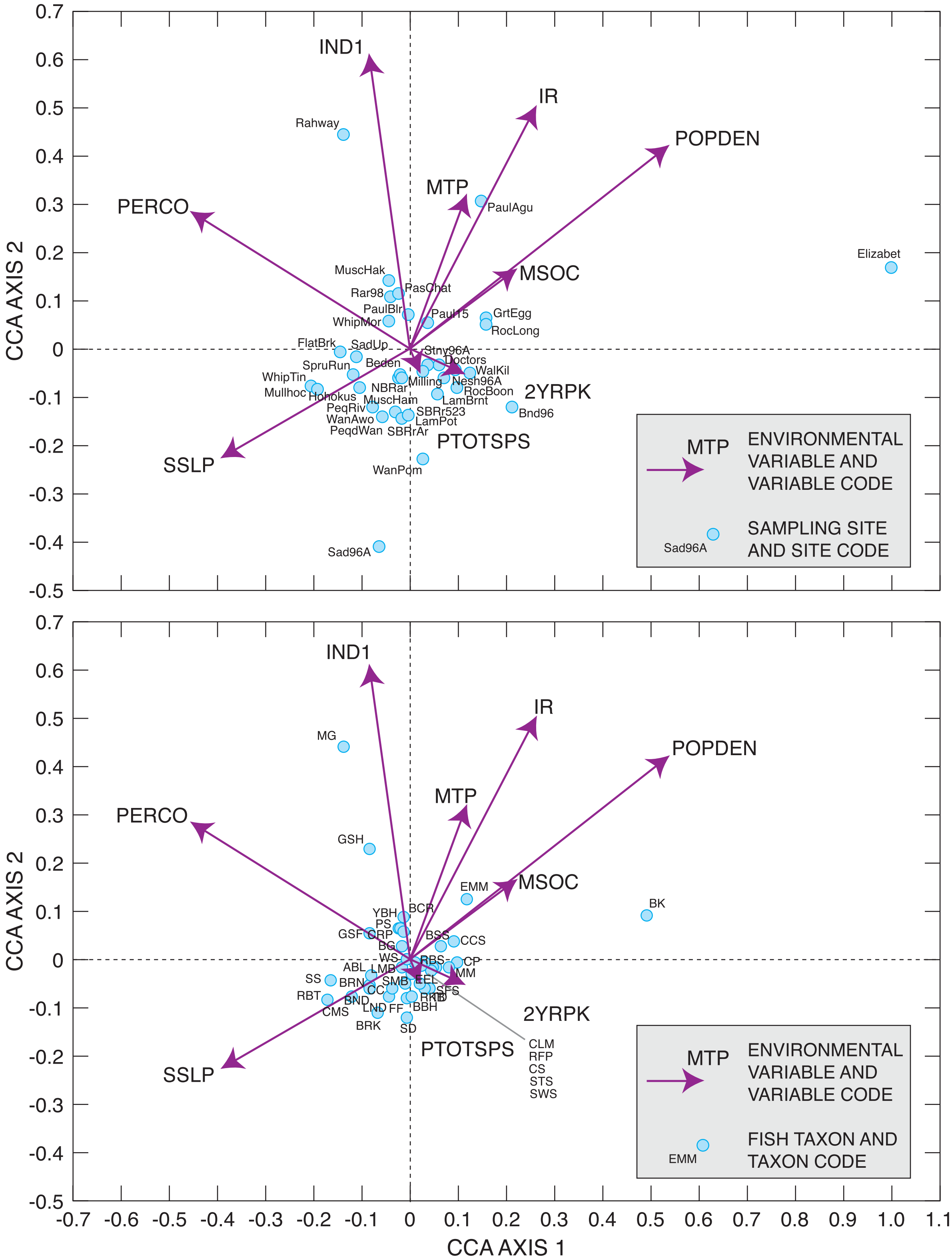


**Figure 11.** Partial canonical correspondence analysis (pCCA) biplots of the relation of environmental variables to fish sampling sites and species in New Jersey. (Site codes, taxa codes, and environmental variable codes are listed in tables 1, 2, and 9, respectively. The environmental variables shown are related significantly to one or more of the pCCA axes (table 11). Arrows (vectors) represent the importance of an environmental variable and the direction of influence. Only species with a minimum weight of 30 (39 of 43 species) are displayed. The larger a species weight, the greater its influence on the ordination. Peripheral species have little influence on the ordination and, therefore, are not displayed.)

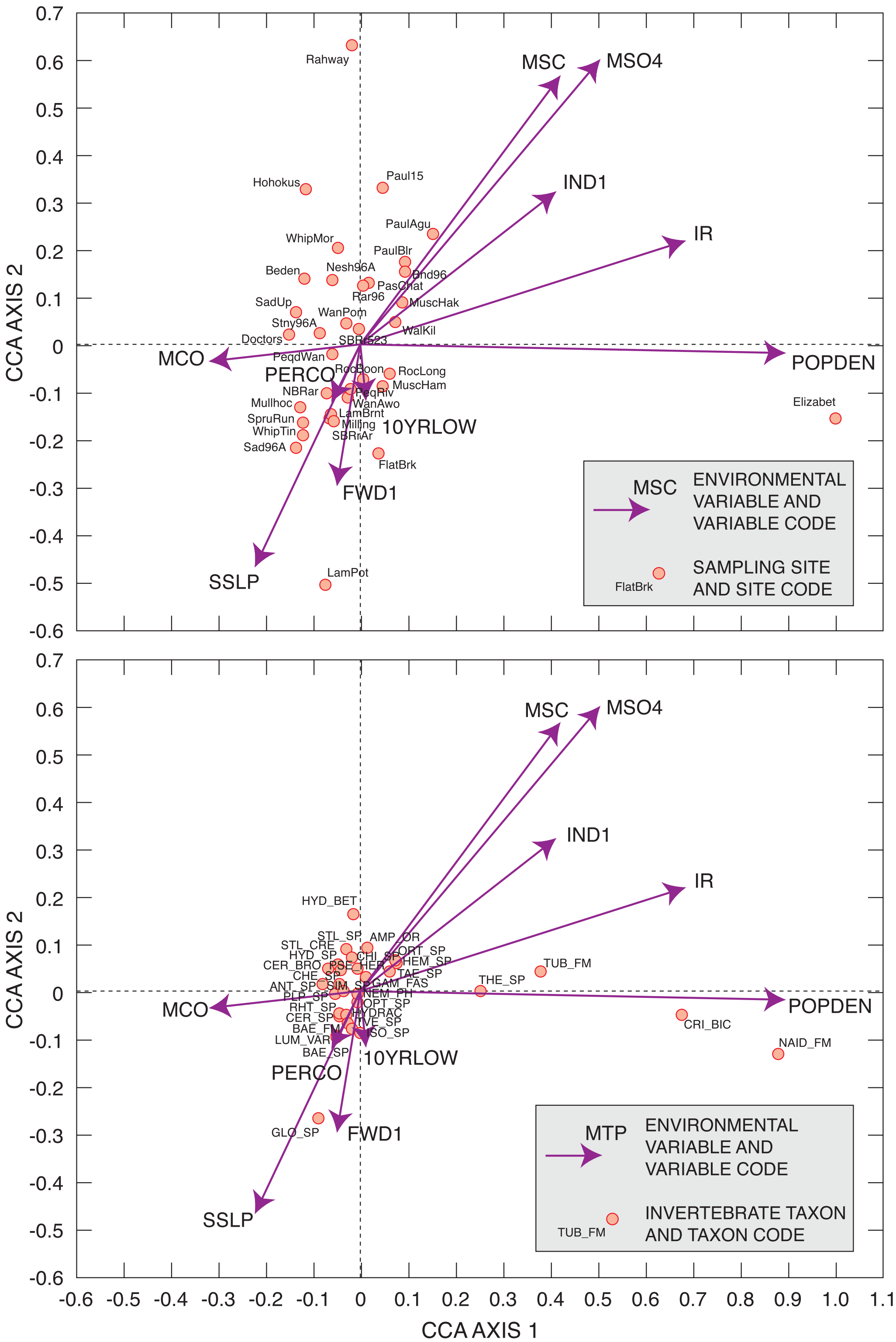


**Figure 12.** Partial canonical correspondence analysis (pCCA) biplots of the relations of environmental variables to invertebrate sampling sites and species in New Jersey. (Site codes, taxa codes, and environmental variable codes are listed in tables 1, 3, and 9, respectively. The environmental variables shown are related significantly to one or more of the pCCA axes (table 12). Arrows (vectors) represent the importance of an environmental variable and direction of influence. Only species with a minimum weight of 5 (30 of 170 species) are displayed. The larger a species weight, the greater its influence on the ordination. Peripheral species have little influence on the ordination and, therefore, are not displayed.)

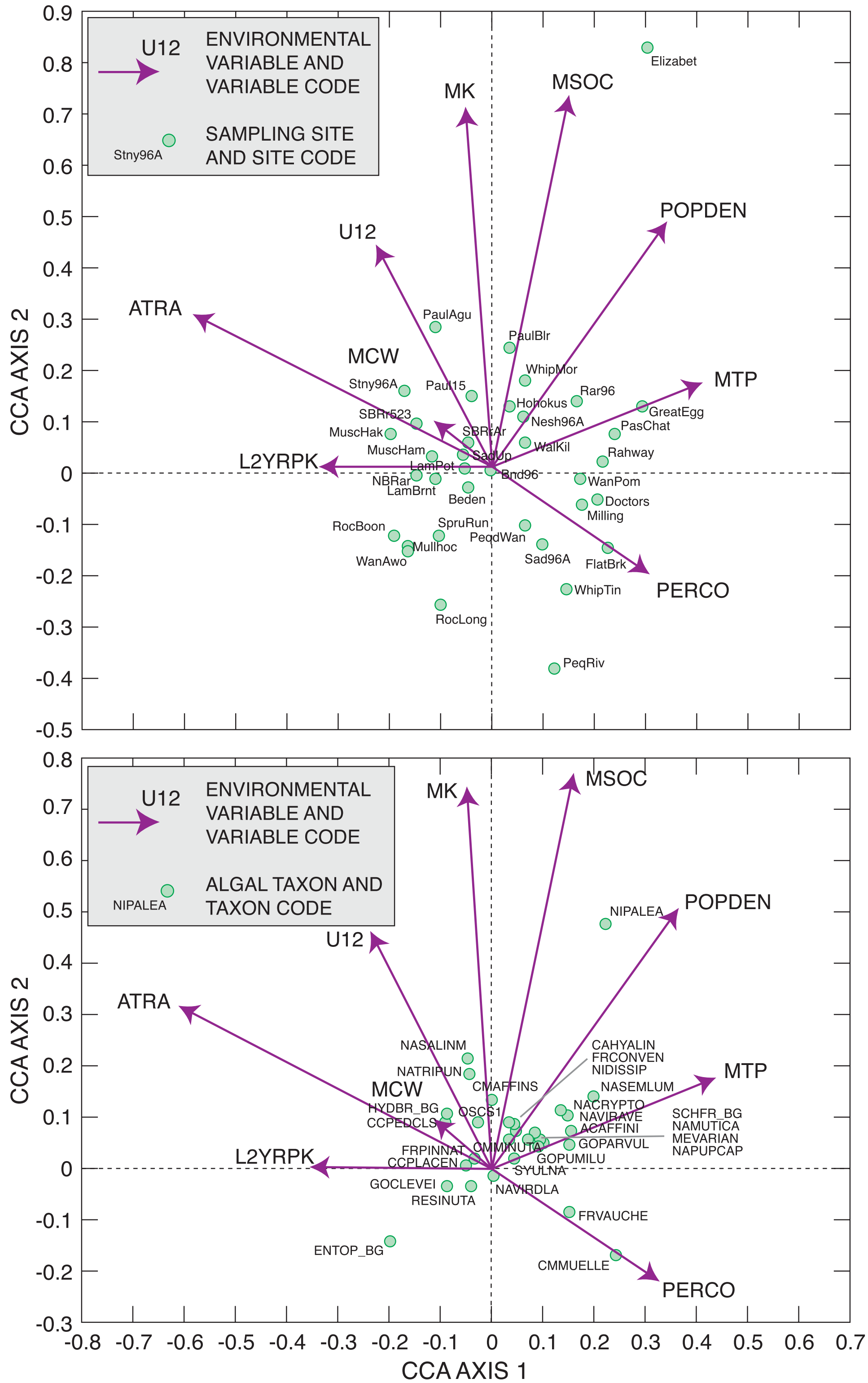


**Figure 13.** Partial canonical correspondence analysis (pCCA) biplots of the relations of environmental variables to algal sampling sites and species in New Jersey. (Site codes, taxa codes, and environmental variable codes are listed in tables 1, 4, and 9, respectively. The environmental variables shown are related significantly to one or more of the pCCA axes (table 13). Arrows vectors represent the importance of an environmental variable and the direction of influence. Only species with a minimum weight of 6 (30 of 103 species) are displayed. The larger a species weight, the greater its influence on the ordination. Peripheral species have little influence on the ordination and therefore are not displayed.)

**Table 10.** Results of Monte Carlo global permutation tests of significance for the first canonical axis and the sum of all canonical axes for partial canonical correspondence analysis (pCCA) of fish, invertebrate, and algal assemblage data in New Jersey

[Test performed by using 399 permutations under the reduced model]

| | p-value | |
|---|---|---|
| **Community** | **First axis** | **Sum of all axes** |
| Fish | 0.0200 | 0.0050 |
| Invertebrate | .0050 | .0050 |
| Algae | .0025 | .0025 |

**Table 11.** Canonical correlation coefficients of environmental variables with the first four partial canonical correspondence analysis (pCCA) axes for the fish community in New Jersey streams

[Environmental variables with a significant relation to at least one of the first four ordination axes are displayed in bold. Variables were evaluated by using t-values, which, when greater than 2.1, indicate a significant relation to the axis; numbers in parentheses are eigenvalues; POPDEN, population density; MTP, median total phosphorus concentration; PERCO, percent cobble; IR, road impervious surface area; 2YRPK, 2-year peak flow; SSLP, mean stream slope; PTOTSPS, total point-source flow; IND1, non-road impervious surface area; MSOC, median suspended organic carbon concentration]

| Environmental variable | Axis 1 (0.47) | Axis 2 (0.44) | Axis 3 (0.40) | Axis 4 (0.38) |
|---|---|---|---|---|
| POPDEN | **0.58** | 0.45 | **0.47** | **- 0.14** |
| MTP | **.13** | **.33** | - .18 | **- .26** |
| PERCO | **- .46** | .29 | **.40** | - .15 |
| IR | **.31** | **.59** | .20 | **- .38** |
| 2YRPK | **.16** | - .07 | .18 | **- .23** |
| SSLP | **- .42** | **- .25** | .54 | **- .03** |
| PTOTSPS | .01 | - .03 | - .36 | **- .25** |
| IND1 | **- .12** | **.79** | .21 | - .08 |
| MSOC | .21 | **.16** | - .28 | **- .34** |
| Cumulative percent of species variance explained | 8.2 | 15.8 | 22.8 | 29.5 |
| Cumulative percent of species-environment relation explained | 14.1 | 27.2 | 39.4 | 50.8 |

**Table 12.** Canonical correlation coefficients of environmental variables with the first four partial canonical correspondence analysis (pCCA) axes for the invertebrate community in New Jersey streams

[Environmental variables with a significant relation to at least one of the first four ordination axes are displayed in bold. Variables were evaluated by using t-values, which, when greater than 2.1, indicate a significant relation to the axis; numbers in parentheses are eigenvalues; POPDEN, population density; MSC, median specific conductance; PERCO, percent cobble; MSO4, median dissolved sulfate concentration; IR, road impervious surface area; FWD1; area of forest and wetlands in a basin; SSLP, mean stream slope; IND1, non-road impervious surface area; 10YRLOW, 7-day, 10-year low flow; MCO, median canopy opening]

| Environmental variable | Axis 1 (0.52) | Axis 2 (0.23) | Axis 3 (0.20) | Axis 4 (0.18) |
|---|---|---|---|---|
| POPDEN | **0.97** | **0.03** | 0.12 | 0.05 |
| MSC | **.41** | .51 | **- .11** | .22 |
| PERCO | **- .05** | **.11** | **.64** | **.50** |
| MSO4 | **.57** | **.64** | - .01 | **.04** |
| IR | **.80** | .23 | .12 | **.09** |
| FWD1 | .08 | - .45 | - .04 | **.17** |
| SSLP | .22 | **- .44** | - .08 | .31 |
| IND1 | .54 | **.40** | **- .37** | **.35** |
| 10YRLOW | - .01 | - .13 | - .14 | **.45** |
| MCO | **.43** | - .04 | - .07 | .04 |
| Cumulative percent of species variance explained | 13.5 | 19.6 | 24.7 | 29.4 |
| Cumulative percent of species-environment relation explained | 28.1 | 40.7 | 51.3 | 61.0 |

**Table 13.** Canonical correlation coefficients of environmental variables with the first four partial canonical correspondence analysis (pCCA) axes for the algal community in New Jersey streams

[Environmental variables with a significant relation to at least one of the first four ordination axes are displayed in bold. Variables were evaluated by using t-values, which, when greater than 2.1, indicate a significant relation to the axis; numbers in parentheses are eigenvalues; POPDEN, population density; MTP, median total phosphorus concentration; PERCO, percent cobble; 2YRPK, 2-year peak flow; MK, median potassium concentration; MSOC, median suspended organic carbon concentration; U12D1, area of commercial plus industrial land in a basin in 1986; ATRA, concentration of atrazine detected in a sample; MCW, median channel width]

| Environmental variable | Axis 1 (0.35) | Axis 2 (0.24) | Axis 3 (0.23) | Axis 4 (0.15) |
|---|---|---|---|---|
| POPDEN | 0.36 | **0.51** | **0.56** | -0.06 |
| MTP | **.47** | **.19** | **.07** | - .09 |
| PERCO | **.30** | **- .20** | .10 | - .53 |
| 2YRPK | **- .58** | .00 | **.09** | .04 |
| MK | **- .05** | **.75** | **- .32** | .08 |
| MSOC | .14 | **.71** | - .03 | .22 |
| U12D1 | **- .23** | **.47** | **.05** | .29 |
| ATRA | **- .57** | .30 | - .14 | - .04 |
| MCW | **- .15** | **.12** | .21 | **- .82** |
| Cumulative percent of species variance explained | 12.8 | 12.5 | 30.0 | 35.4 |
| Cumulative percent of species-environment relation explained | 26.4 | 44.2 | 61.8 | 73.0 |

suspended organic carbon concentration (organic enrichment), high point-source flows, and a negative response to flood frequency.

As with the fish data, population density and impervious surfaces dominate the first axis of the invertebrate pCCA (fig. 12). MSC and MSO4 also are highly correlated with the first axis (table 12). Again, the first axis separated the most anthropogenically modified stream (Elizabet) from sites falling on the lower end of the urban gradient (for example, Mullhoc and SpruRun). Substrate composition, median dissolved sulfate concentration, and impervious surfaces account for most of the variability in the second axis. Less impaired sites with higher proportions of habitable cobble substrate and less impervious surface area are found below the origin in the lower left and right quadrants of the pCCA (for example, LamPot, WhipTin, and SBRrAr), whereas sites with high concentrations of dissolved sulfate and greater road and non-road impervious area (for example, Rahway and Hohokus) are found toward the top of the pCCA. Non-road impervious surface (r=-0.37) and substrate composition (r=0.64) were significantly related to the third axis (table 12). The presence of stream base flow and the amount of forest and wetland area near a sampling site (FWD1) are significantly related to the fourth axis.

Impervious surfaces, atrazine concentration (represents an agricultural chemical signature), nutrient concentration, substrate composition, and periodicity of flood events dominated the first axis of the algal community pCCA (table 13; fig. 13). Unlike the first axis of the fish and invertebrate analyses, the first axis of the algal pCCA did not appear to separate the most disturbed from the least disturbed sites. Rather, the first axis separated streams with nutrient concentrations and wider channels on the right (for example, PasChat and Rahway) from streams in urbanizing areas with a residual agricultural signature, such as Stny96A and SBRr523, on the left of the pCCA diagram along the atrazine vector (fig. 13). Median suspended organic carbon concentration (an indicator of organic enrichment), median total phosphorus concentration (nutrients), urban commercial plus industrial LU, and population density describe the second axis. The second axis more readily conforms to the urban-gradient concept with the heavily disturbed Elizabet site along the top of the pCCA diagram, strongly separating from less disturbed sites such as RocLong and PeqRiv, which are near the bottom of the pCCA biplot. Median potassium concentration (r=0.32) and population density (r=0.56) are highly correlated with the third axis, and median channel width (r=-0.82) is the only variable that describes the fourth axis (table 13). Channel width likely is related to autochthonous production of periphyton in that periphyton production increases in response to increased light availability in wider stream channels (for example, more photosynthesis and metabolism). The high concentration of nutrients and greatly reduced canopy and riparian cover in many of these urban basins appears to provide ideal conditions for increasing algal biomass.

## RELATION OF ENVIRONMENTAL CHARACTERISTICS TO AQUATIC ASSEMBLAGES

The findings of this study indicate that variation in the structure of fish, invertebrate, and algal assemblages is directly related to a variety of anthropogenic activities, and that this variation is expressed along an urban land-use gradient. The environmental characteristics that were found to be most important in determining aquatic-assemblage structure include human population density, amount and type of impervious surface cover, nutrient concentrations, organic enrichment, hydrologic instability, the percentage of land in a basin occupied by forest and wetlands, and substrate quality. On the basis of the results of many of the urban-gradient models evaluated, however, changes in the landscape associated with increasing impervious cover and population density clearly are good predictors of community impairment and a decrease in biological integrity.

Variation in the fish and invertebrate communities was greatest along the ULUG, with axis 1 of the MLR and pCCA models indicating significant relations with population density and percent impervious surface area (tables 9, 11, and 12). The variation in algal-community structure also was related to changes in environmental conditions along the ULUG, but nutrient enrichment, hydro logic instability, and agricultural activities were most important. Variables related to changes in nutrient levels (MTP), hydrologic instability (2YRPK), and an agricultural chemical signature (ATRA) were significantly related to the first axis of the algal pCCA model, whereas population and urban industrial and commercial land were found to be more highly correlated with the second axis. These differences indicate that anthropogenic factors do not affect all aquatic communities equally. Hence, to understand fully the overall effects of environmental degradation associated with urban landscapes, communities at multiple trophic levels and environmental characteristics at multiple spatial scales need to be evaluated.

Obviously, not all environmental characteristics evaluated in the urban-gradient models affected the aquatic communities equally. Although aquatic communities are known to respond to the combined effects of multiple environmental stressors (chemical, physical, and biological), some individual variables in the MLR analysis were found to account for a greater portion of first-axis community variation across the ULUG models than other, less influential variables. For example, partial $r^2$ values for population density accounted for 79 and 62 percent of the first-axis variability in aquatic invertebrate and fish communities, respectively. In addition, mean annual peak flow accounted for 69 percent of the variability in the first axis of the algal community. These partial relations support earlier findings that specific anthropogenic factors such as population density and streamflow alone can affect significantly the biotic integrity of fish, invertebrate, and algal communities. Although highly significant in many of the models presented in this study, population density incorporates many socioeconomic and anthropogenic characteristics (Grimm and others, 2000) that relate to other contributing factors and further substantiate the need for integrated investigations of urban environments.

Results of various studies have indicated that LU/C at the basin scale is the primary anthropogenic effect on biotic integrity (Roth and others, 1996; Allan, 1995; Allan and Johnson, 1997). Others, however, have indicated that reach-scale properties such as substrate, depth and velocity, and canopy cover are more important than LU/C in explaining aquatic-community variability (Hawkins and others, 1982; Richards and others, 1997). This difference likely is explained as an artifact of the scale of study. In reality, basins affect reach-scale variables which, in turn, affect aquatic assemblage structure (Frissell and others, 1986; Lammert and Allan, 1999). In this study, environmental variability was evaluated at the basin, segment, and reach scales. The design identified geomorphic processes and major structural features of the stream corridor that, when modified, affect aquatic-assemblage structure. Landscape- and basin-scale variables such as impervious surface cover, population density, nutrient loading, and hydrologic instability all were found to affect aquatic-assemblage structure in this study; however, these are landscape factors that directly affect reach-scale variables (Lammert and Allan, 1999) and ultimately modify aquatic-assemblage structure. Reach-scale variables such as the amount of high-quality cobble substrate at the sampling site (PERCO), mean channel width (MCW), mean canopy opening (MCO), and stream slope (SSLP) were found to be significantly related to the structure of one or more of the aquatic assemblages (tables 10, 11, and 12). These findings further support the direct link between proximate (local-scale) environmental characteristics and aquatic-assemblage structure.

## Impervious Surfaces

Many investigators have documented a direct link between increases in impervious surface cover and increases in factors such as nonpoint-source runoff (Wilber and Hunter, 1977; Schueler, 1994; Ometo and others, 2000); surface runoff (Garie and McIntosh, 1986; Hollis, 1975); stormflow and flood frequency (Sauer and others, 1983; Brown, 1988; Lazaro, 1990); channel erosion, downcutting, and sediment yield (Trimble, 1997; Pizzuto and others, 2000); changes in water quality and temperature (Klein, 1979; Booth and Jackson, 1997); reductions in base flow (Brun and Band, 2000); and changes in aquatic-assemblage structure (Karr and Schlosser, 1978; Schlosser, 1991, Booth and Jackson, 1997; Wang and others, 1997; Klauda and others, 1998). The outcomes of these studies and the current study further substantiate the need to address the ecological consequences of ISC in watersheds at all planning and development stages. In addition, the type as well as the total amount of ISC in a watershed affects aquatic communities. For example, in this study non-road ISC often was significantly related to impairment of the aquatic-invertebrate community (tables 9 and 12, fig. 12), showing that in addition to the adverse effects of road impervious surfaces on all three aquatic communities (table 9), other types of impervious surfaces in suburban areas, such as sidewalks, patios, rooftops, and compacted soils, lawns, and fields that have lost water-retaining functionality, also can affect adversely the aquatic-invertebrate community. Separating road from non-road impervious area provided evidence that ISC at all levels of urban development, not just the most densely urban areas, affect aquatic-assemblage structure.

Evidence in the literature indicates a link between ISC and stream-ecosystem impairment. Some researchers have suggested that significant impairment begins at approximately 10 percent ISC (Schueler, 1994; Arnold and Gibbons, 1996; May and others, 1997). A second threshold appears to be present at about 25 to 30 percent ISC, where most indicators of stream quality consistently shift to poor condition (for example, aquatic diversity, water quality, and habitat scores). Some lines of evidence indicate that ISC may be linked to the quality of other watershed resources, such as lakes, reservoirs, estuaries, and aquifers (Caraco and others, 1998). Klein (1979) reported that aquatic-invertebrate and fish impairment is first observed when watershed imperviousness reaches about 12 percent, but does not become severe until it reaches about 30 percent. Booth and Jackson (1997) suggest that fish-habitat degradation occurs between 8 and 10 percent ISC, and Galli (1991), Schueler (1994), Klein (1979), and many others have demonstrated that the loss of sensitive fish species is directly related to increasing ISC. Increased ISC resulting from increasing levels of human disturbance in a watershed consistently has been linked to lower aquatic-insect diversity, a decline in channel stability, and a decline in fish-habitat quality (Caraco and others, 1998). In addition, Leidy and Fieldler (1985) found that the number of exotic species increased at sites subject to moderate human disturbance.

The threshold observed in this study differed among the three aquatic communities studied (fig. 10). For example, a substantial change occurred in fish-community condition between 18 and 24 percent ISC and in aquatic-invertebrate community impairment between 20 and 25 percent ISC. These results do not conform directly to findings of previous studies, which indicate that impairment begins at approximately 10 percent ISC. On average, an inflection point in the relation between ISC and community impairment in New Jersey streams did not appear until ISC was about 18 percent (fig. 10). This finding may indicate that (1) this study did not include a sufficient number of sites at the lower end of the ULUG to discern minor changes in community impairment; (2) many sites, including the least impaired sites in New Jersey, already are affected by low levels of anthropogenic disturbance, thereby dampening the effects of increasing ISC on community impairment; or (3) changes in community composition, including invasion of non-endemic species and increases in disturbance-tolerant organisms, already have occurred historically in many New Jersey watersheds, reducing the discriminatory ability of the community indices. Many studies (for example, Caraco and others, 1998), however, are of streams with drainage areas that typically are less than 53 $km^2$ (20 $mi^2$), whereas the watersheds in this study (third and fourth order) averaged about 152 $km^2$ (59 $mi^2$). The larger basin size may explain some of the differences seen between ISC thresholds in the literature and that found in this study. The more likely explanation, however, is that the high level of variability in community impairment at the lower end of the ULUG precludes appropriate discrimination, and that the inflexion point at approximately 18 percent ISC seen for communities in streams in urban and urbanizing New Jersey drainage basins may be the critical point at which moderate community impairment becomes discernible. This explanation is consistent with the premise that a second threshold is present at higher levels of ISC (at or near 20 percent), where indicators of stream quality shift to poor condition (Schueler, 1994; Arnold and Gibbons, 1996; May and others, 1997). Additional analyses relating community indices to other indicators of urbanization and the evaluation of additional sites at the higher end of the ULUG may improve our ability to define aquatic-community disturbance thresholds in New Jersey streams.

## Hydrologic Instability

A large proportion of the rain that falls on naturally vegetated landscapes typically infiltrates the soil, while the remaining water runs off into the streams. When landscapes are modified and the hydrologic cycle and natural runoff process and pathways are disturbed, however, less water is able to infiltrate the soil, and surface runoff increases. This process induces dramatic changes in stream hydrology that ultimately affect fish-, invertebrate-, and algal-assemblage structure. Evidence indicates that physical changes associated with hydrologic instability in watersheds degrade the habitat for aquatic communities (Lenat and Crawford, 1994; Richards and others, 1996). Schroeder and Savonen (1997) suggest that changes in the hydrologic regime can increase flood frequency and magnitude and alter the response of stream channels to floods as a result of changes in stream structure and morphology. Richter and others (1996) and Ward and Stanford (1989) indicate that modification of the hydrologic regime can alter the composition, structure, and function of aquatic ecosystems through their effects on environmental and habitat characteristics. In this study, changes in hydrologic factors (for example, decreases in base flow (10YRLOW), periodicity in peak discharge (2YRPK), and the flashiness of streamflow (RT2575) substantially affected the types and condition of invertebrate, fish, and algal communities in the study streams (table 9; figs. 11, 12, and 13), in part by their direct effects on the stream channel. Changes in mean annual peak flow (2YRPK) were significantly related to all communities in the MLR models (table 9); however, 2YRPK was found to be significantly related only to the fish- and algal-community structure in the pCCA (tables 11 and 13; figs. 11 and 13). Changes in fish-community structure also were significantly related to RT2575, an index of stream flashiness (table 9). These models appear to indicate that all three stream communities in urban areas of New Jersey are stressed as a result of increased fluctuation in discharge, and rarely reach stable population levels. In addition, these modified hydrologic processes promote the retention of highly tolerant aquatic organisms as species with life histories sensitive to hydrologic instability become much less prevalent. These findings are consistent with the results of many recent studies that indicate that streamflow variability is a significant determinant of the biological integrity of aquatic systems (for example, Poff and Allan, 1995; Clausen and Biggs, 1997; Pusey and others, 2000).

In addition, reductions in base flow resulting from changes in runoff patterns and from water-use and wastewater-distribution practices affect the suitability of a stream for many types of organisms (Klein, 1979). Brun and Band (2000) found that base flow declined as much as 20 percent from pre-urbanized times in a basin in urban Baltimore, Md. The capability of a stream to maintain base flow (10YRLOW; table 3) was significantly related to invertebrate-community structure (table 12, fig. 12). Blinn and others (1995) found that fluctuations in base flow caused repeated exposure of areas along stream margins to extended dewatering, which resulted in slow recovery and decreased production of macroinvertebrate communities. As the periodicity of dewatering events increases, the size of the habitat available to aquatic-invertebrate communities decreases. These processes inevitably result in reduced richness and diversity of aquatic macroinvertebrates, especially those that depend on margin habitat for survival.

## Effects of Runoff Quality

Aquatic-assemblage structure downstream from urban landscapes can be modified as a result of stormwater runoff that carries with it many terrestrial-borne constituents (fig. 14). Urban runoff commonly contains a combination of chemicals and chemical mixtures that include industrial and fuel-related compounds (for example, VOC's) (O'Brien and others, 1997); trace elements (explained in greater detail in the Trace-Elements section) derived from exhaust, tire derivatives, and asphalt wear; and pesticides (Reiser and O'Brien, 1999). The toxicity of these chemicals and chemical mixtures and their long-term (years) biological effects on aquatic communities is poorly understood, as are the potential for synergistic effects of

**Figure 14.** Unregulated impervious-area runoff exemplified by a storm-sewer pipe that drains directly into the Saddle River at Ridgewood, N.J. (01390500).

multiple constituents as they travel along diverse physiological pathways.

Although no trace element in either streamwater or the sediment was found to be significantly correlated with characteristics of the aquatic assemblages, MK in the water column was significantly correlated with characteristics of the fish community (table 9) and the algal community (table 13, fig. 13). In this study, however, MK was determined to be a surrogate for chemical use and represents the combined effects of various chemical and land-use variables including MSOC, concentration of methyl tert-butyl ether (a frequently detected VOC), agricultural land use, ATRA, and the number of pesticides detected (NHITSP). These five variables accounted for 84 percent of the variability in MK ($p<0.001$). This result indicates that MK represents a mixed agricultural and urban (chemical use) signature, and its significance in the MLR models is evidence of the cumulative effects of both these land uses on aquatic communities. The negative relations seen at opposite ends of the trophic structure—that is, fish and algae—exemplify the widespread effects of these types of anthropogenic disturbance on aquatic-assemblage structure.

Runoff from surrounding agricultural and urban landscapes is likely to contain many pesticides (Reiser and O'Brien, 1999), but the effects of these compounds on stream communities are poorly understood (Baker and Johnson, 1983) and, to date (2002), no comprehensive monitoring approaches designed to link pesticide effects to changes in aquatic communities in New Jersey streams have been developed. In this study, atrazine was detected at all sites. Moreover, although atrazine concentrations never exceeded the USEPA maximum contaminant level of 3 µg/L (U.S. Environmental Protection Agency, 1996), the concentration of this compound was found to have a strong negative relation to algal-community structure (table 13, fig. 13). Atrazine is an herbicide commonly applied for weed control in agricultural areas and for professional lawn care in urban and suburban areas throughout New Jersey, as are other herbicides. Atrazine could be acting as a surrogate for the suite of other herbicides used and detected in these watersheds (Reiser and O'Brien, 1999). Although the breakdown of these herbicides is known to be rapid, excess material applied to lawns and agricultural fields can be carried in runoff to streams and rivers. Under this scenario, herbicides in the water column could inhibit directly the growth and production of periphyton. The connection of herbicides in runoff to biological effects at higher trophic levels is less obvious, but may be related to "bottom-up" control of herbivore populations (Elwood and others, 1981; Hart and Robinson, 1990). Several investigators have shown that invertebrate grazer abundance, growth, and distribution are directly related to periphyton abundance (Jacoby, 1985; Lamberti and others, 1987; Welch and others, 1992; Feminella and Hawkins, 1995; Kjeldsen, 1996). Therefore, aquatic systems with diminished periphyton abundance resulting from herbicide use could indirectly affect the abundance of invertebrate grazers. Ultimately, this trophic alteration could affect adversely the growth and survival of highly sought-after game fish such as salmonids that rely on aquatic invertebrates for food.

## Erosion, Sedimentation, and Stream-Channel Modification

Increases in sedimentation can affect aquatic-assemblage structure directly by altering physical and biochemical conditions in a stream. These changes can include reduction in diversity, density, and richness of aquatic communities resulting from modified interstitial space, respiratory diffusion gradients, and food resources in sediments (Williams and Feltmate, 1992). Many investigators have reported a general decline in invertebrate density with increasing sedimentation (Chutter, 1969; McClelland and Brusven, 1980; Lenat and others, 1981; Bourassa and Morin, 1995; Zweig and Rabeni, 2001). Wood and Armitage (1997) suggest that sedimentation is a key concern in streams threatened by anthropogenic disturbance. In contrast, it is well established that high invertebrate richness and density are related to medium to large substrate heterogeneity, which provides stability; interstitial spacing for refuge and colonization; oxygen exchange; attachment sites for filter feeders; and a microbial, algal, and detrital food supply (Minshall, 1984; Allan, 1995; Wood and Armitage, 1997). The results of multivariate analyses in this study emphasize the association between substrate quality and the observed patterns in fish-, invertebrate-, and algal-assemblage structure in New Jersey streams. For example, the amount of cobble substrate in a stream was found to have a strong positive relation with aquatic macroinvertebrate community structure in the MLR models (table 9), and was found to be a significant variable along multiple axes for all aquatic communities in the pCCA (tables 11, 12, and 13). These findings indicate that a reduction in substrate quality and other stream habitat is associated with urban streams and likely results from channel erosion, sedimentation, and increases in flow, common to systems with minimally controlled urban stormwater runoff (fig. 14). Inevitably, this reduction in substrate quality affects the health and condition of all three types of New Jersey stream communities.

Increased watershed erosion and streambank degradation results in increases in downstream sedimentation and the loss of natural instream habitats such as rock ledges, deep pools, and substrate heterogeneity, which can affect directly aquatic community assemblage structure. Greater bank stability reduces streambank degradation and mitigates the extent to which many of these instream habitats are modified as the result of watershed erosion. Mean bank dominant substrate (MBDS), a surrogate for bank stability in this study, was significantly related to the changes in fish and algal communities in the MLR models (table 9). These relations may indicate that specific ecological characteristics of these communities are affected by changes in habitat conditions within a sampling reach. For example, fish reproduction may be restricted if bank degradation results in increases in silt and sand that cover prime spawning gravel. Salmonids (for example, Salmo trutta) and cyprinids (for example, Rhinichthys atratulus) require cobble and gravel substrates for spawning and reproduction. In addition, silt deposition on cobble and gravel substrates can inhibit the growth of sensitive algal species, whereas silt-tolerant species will flourish. Tolerant algae such as those in the genera Navicula, Nitzschia, and Surirella made up from 86 to 98 percent of the overall community abundance at sites along the urban gradient where urban land use exceeded 47 percent (Ayers and others, 2000).

Sediments also can carry contaminants such as pesticides, trace elements, and nutrients that not only affect communities at the source, but later can be released into the water column and become problematic downstream. Nutrient-related variables, such as median total phosphorus concentration (MTP), accounted for a significant proportion of the community variability in the algal and fish MLR models (table 9), and were strongly (and significantly) related to the first and second pCCA axes of the algal and fish communities, respectively (tables 11 and 13). Sampling reaches positively associated with high concentrations of nutrients (for example, Nesh96 and PasChat) typically contained a high proportion of algal taxa indicative of nutrient enrichment, such as Navicula seminulum and Nitzschia palea (fig. 13). Highly tolerant omnivorous fish such as the eastern mudminnow (Umbra pygmaea) and the mummichog (Fundulus heteroclitus) take advantage of increases in aquatic invertebrates that flourish when algal production increases. These fish species were common at urban sites and typically were located along vectors in the pCCA plot (for example, POPDEN and IND1) that represented substantial anthropogenic disturbance.

## Trace Elements

Geologic weathering accounts for natural releases of trace elements to aquatic environments. Human activities, however, have accelerated the release through point- and nonpoint-source contamination. Historically, industrial and other point sources were important, as were releases from fossil-fuel burning and use of trace-element-based pesticides. More commonly now, trace elements from atmospheric deposition, vehicular traffic, and other activities accumulate on urban surfaces and subsequently are carried in runoff to streams. When introduced into aquatic environments, trace elements sorb to fine-grained sediments (Forstner and Wittman, 1983). Trace elements can accumulate in sediments and affect the health of benthic organisms and higher trophic-level species that rely on benthic organisms for food. Some trace elements such as copper, iron, manganese, selenium, and zinc are vital to the metabolic processes of aquatic organisms (Sorenson, 1991); however, they can be toxic at elevated concentrations above natural background levels. Although urban streams typically are characterized by high trace-element concentrations (Lenat and Crawford, 1994; DeVivo and others, 1997) and although trace elements were detected at many of the sampling sites assessed in this study (O'Brien, 1997), none was significantly related to the aquatic communities in the MLR or pCCA models evaluated. This result does not necessarily indicate that the presence of trace elements in the water column or bed sediment had no effect on the aquatic communities studied; rather, it indicates that the effect may be less discernible than that seen for stronger anthropogenic influences such as impervious surface cover, flow characteristics, or population density, which accounted for a significant proportion of the variability in aquatic-assemblage structure.

## Riparian Conditions and Stream Buffers

Watershed development has been found to affect riparian conditions along the stream corridor, both directly and indirectly. May and others (1997) report that stream buffer width, vegetative condition, and longitudinal connectedness of buffer vegetation are altered as the level of development in a watershed increases. The width of these buffers is inversely related to ISC. These processes inevitably cause fragmentation of the riparian corridor and greatly reduce the amount of mature vegetation. In addition, soil erosion during and after construction activities affects vegetative cover by promoting the movement of invasive species, increasing pathogens, inhibiting vegetative regeneration, and increasing foliar damage (Reid, 1993; Schlosser and Karr, 1981). Reduction in canopy cover and the resultant changes in the stream-temperature profile can modify the structure and function of the aquatic communities. The total amount of 15-m forest buffer in a watershed (S15F) was found to be significantly related to the biotic integrity of the fish community in the MLR analysis (table 9). Schlosser and Karr (1981) indicate that stream buffers reduce the amount of sediment entering the stream channel by stabilizing stream banks and mitigating the effects of erosion. Intolerant fish species without mechanisms to survive in streams where substrate quality has been diminished by silt deposition are less likely to be present in disturbed than undisturbed streams. For example, benthic insectivorous fish such as the margined madtom (Noturus insignis), shield darter (Percina peltata), and slimy sculpin (Cottus cognatus) were absent in impaired streams (for example, Elizabet and Rahway) where the benthic-community structure has been modified by siltation and other anthropogenic processes. In contrast, species of motile algal taxa such as those in the genera Navicula, Nitzschia, and Gomphonema appear to have flourished at the more heavily disturbed sites. Many of these species also have mucilaginous stalks or filaments that help prevent burial by sediments.

## Forest and Wetlands

Results of previous investigations have indicated that aquatic-community health is affected negatively by the conversion of forested and agricultural lands to urban uses (Benke and others, 1981; Garie and McIntosh, 1986; Jones and Clark, 1987; Kennen, 1999). In New Jersey, new urban areas are rapidly displacing forest and agricultural lands. Investigations by Richards and Host (1994) and Richards and others (1996) have indicated a strong correlation between forested land and benthic-community structure. Roth and others (1996) emphasized that the amount of forested land in a basin was directly related to fish-community IBI scores. Similarly, wetlands protect water quality by providing a natural filter and by reducing concentrations of nitrate in surface waters (Lowrance, 1998; Johnston, 1991; Osborne and Kovacic, 1993). In this study, the area of forest and wetlands in a basin was related positively to aquatic-invertebrate community structure (table 12; fig. 12). Forests and wetlands play a major role in maintaining a high-quality supply of water, food, and habitat for disturbance-intolerant and highly desired species (Ayers and others, 2000). In this capacity, forest and wetlands help mitigate the adverse effects of many human-induced landscape alterations and changes in stream chemistry.

Forest and wetlands near the site (FWD1) was one of the significant distance-weighted variables in the algal community MLR models (table 9). These results indicate both the presence of these land uses and their close proximity to the stream corridor are important determinants of algal-community structure. This finding differs from that found in an earlier study in which the amount of forested land was found to be an important determinant of the level of aquatic invertebrate impairment regardless of its location in the basin (Kennen, 1999). This difference supports the finding that anthropogenic disturbances affect trophically distinct aquatic communities along different ecological pathways.

## Relation of Land-Use Changes to Assemblage Structure of Long-Lived Species

Human activities that modify the landscape, including commercial and residential development, bridge and road construction, deforestation, channel modification, removal of riparian vegetation, and point- and nonpoint-source contamination, historically have been linked to the degradation of the stream community. Although many of these changes occur rapidly (days to months), some occur over a long span of time (years to decades). Changes in land use over time have been linked to changes in aquatic-assemblage structure, especially of long-lived communities such as fish (Allan, 1995). Findings of this study further substantiate the link between long-lived species and historical changes in a watershed. For example, the percent of urban land use in a basin in 1986 was significantly related only to the fish-assemblage structure in the MLR analysis (table 9). Because fish are longer lived than either benthic invertebrates or algae, with species such as the white sucker (Catostomus commersoni) living 15 or more years, many of the changes in the watershed that occurred prior to the current study already have affected fish-assemblage structure. Therefore, the composition of the fish assemblage evaluated in this study already has been greatly modified by anthropogenic activities in the watershed prior to 1986, and its strong relation to the aggregated historical land use further substantiates this association.

## Community Indices

Aquatic community indices were significantly correlated with many of the anthropogenic factors prevalent along the ULUG and appeared to form good predictive models with few highly significant environmental variables (model $r^2$ values for IBI, NJIS, and TDI were 0.84, 0.75, and 0.62, respectively (table 8)). These data indicate that many of the metrics used to derive the indices for assessing biotic integrity in New Jersey streams can be used effectively to differentiate impaired streams from unimpaired streams. Specifically, the IBI and NJIS were related negatively to variables that indicate urban disturbance, such as population density and road ISC, but were positively related to environmental characteristics that typically indicate healthy, unimpaired communities, such as the amount of high-quality cobble substrate at a sampling site, percentage of forest and wetland area in a basin, and the amount of 15-m forested buffer area (table 9). The TDI, however, was a more one-dimensional index and was positively related to variables indicating nutrient enrichment (MTP and MSO4) and the depth of soil near the stream. Although one-dimensional, the TDI did perform the function for which it was designed—that is, to serve as an indicator of sedimentation and nutrient enrichment.

## Indicator Taxa

The results of the multivariate analysis indicate that many taxa are limited by anthropogenic disturbance along the ULUG. Sites along undisturbed vectors on the aquatic invertebrate pCCA, such as FWD1, PERCO, and SSLP (fig. 12) contained many more contaminant-sensitive taxa such as mayflies in the family Baetidae, caddisflies such as Glossosoma sp., and the chironomid Tvetenia sp. Intolerant fish species such as the slimy sculpin (Cottus cognatus) and American brook lamprey (Lampetra appendix) (fig. 11) and sensitive algal taxa such as Cymbella muelleri, Navicula viridula, and Fragilaria vaucheriae (fig. 13) were present along vectors of high gradient or high-quality habitat (SSLP and PERCO, respectively) in the pCCA plots. Most algal taxa belonging to the genus Cymbella are indicative of undisturbed streams with high dissolved oxygen concentration and little organic enrichment (van Dam and others, 1994), and were most abundant in streams at the low end of the ULUG, such as FlatBrk and PeqRiv (table 1). N. viridula, however, can tolerate elevated nitrogen concentrations above natural background levels, but are not nitrogen heterotrophs (van Dam and others, 1994). This species, which plotted along the PERCO environmental vector of the algae pCCA (fig. 13) and opposite the ATRA and U12 vectors, is an oligohalobe known to prefer neutral pH conditions and low conductivity (Patrick and Reimer, 1966). In this analysis, N. viridula did appear to be an indicator of relatively unimpaired stream conditions. F. vaucheriae, however, is an early successional species often described as an indicator of recent scouring. This species also is known to be highly susceptible to grazing and does not compete well in mature communities (Stephen Porter, U.S. Geological Survey, oral commun., 2001). Its presence along the less disturbed vectors of the algal pCCA is likely related to a combination of these two autecological traits.

Changes in the trophic and taxonomic structure of aquatic communities occur in association with anthropogenic disturbance in a watershed (for example, Garie and McIntosh, 1986; Jones and Clark, 1987; Weaver and Garman, 1994). Changes in aquatic-assemblage structure in the study streams are consistent with those found in many earlier investigations. For example, species distributions in ordination space indicated that tolerant species were more common in streams at the high end of the ULUG and that sensitive taxa were more closely associated with less disturbed sites. Highly tolerant invertebrate taxa such as Tubificidae, Naididae, and the chironomids Thienemannimyia group sp. and Cricotopus bicinctus plotted along the POPDEN axis of the pCCA near the most urban site in the assessment (Elizabet). Similarly, highly tolerant fish species such as the banded killifish (Fundulus diaphanus) and eastern mudminnow (Umbra pygmaea) plotted along vectors of substantial anthropogenic disturbance (for example, POPDEN and IR) and organic enrichment (MSOC). In addition, other tolerant species such as golden shiner (Notemigonus crysoleucas), yellow bullhead (Ameiurus natalis), common carp (Cyprinus carpio), and pumpkinseed (Lepomis gibbosus) and bluegill (Lepomis macrochirus) sunfish plotted along the non-road ISC vector in the fish pCCA near the fourth most urbanized site (Rahway). In many New Jersey streams, however, the autecological classification of “tolerant” may be ambiguous, as 82 percent of the species captured in streams during this study were classified as tolerant (table 2).

The algal species Nitzschia palea appeared to be most abundant in the most disturbed streams along the ULUG. It plotted in ordination space between the POPDEN and SOC vectors and was near Elizabet. This species is particularly tolerant of nutrient enrichment (van Dam and others, 1994) and was described as a "pollution" indicator nearly 100 years ago (for example, Kolkwitz and Marsson, 1908). N. palea also is considered by some investigators to be a nitrogen heterotroph (for example, Cholnoky, 1968). This species appears to be highly tolerant to most types of anthropogenic disturbance and commonly co-occurs with Gomphonema parvulum, another highly tolerant taxon (Stephen Porter, U.S. Geological Survey, oral commun., 2001). G. parvulum, however, plotted in a small cluster with three other algal taxa, Navicula cryptocephala, Navicula viridula avenacea, and Achnanthes affinis, in the algal pCCA. All of these taxa fall on the "disturbed" side of the ordination and are aligned with the population density and nutrient vectors. N. cryptocephala and A. affinis generally are considered to be cosmopolitan species, but appear to be tolerant of nutrients and also are halotolerant. G. parvulum is the classic "indicator" in this group, but the abundance of all four species appears to increase as nutrient concentrations increase along a gradient of anthropogenically modified streams.

## COMPLEMENTARY ANALYTICAL TECHNIQUES

Some investigators may find some analytical approaches used in this study to be redundant; for example, partial constrained ordination methods commonly incorporate reciprocal averaging and multiple regression into a single multivariate analysis. ter Braak and Prentice (1988), however, recommend the use of partial constrained ordination after regression analysis to relate the residual variation to other environmental variables. They also state that this type of analysis is particularly useful when the explanatory variables can be subdivided into two sets, a set of covariables and a set of environmental variables whose effects are of particular interest. The authors found that the methods used in this study (MLR and pCCA) were highly complementary and provided a more comprehensive evaluation of the many anthropogenic factors affecting aquatic communities along an ULUG than either approach used independently. In addition, the approach used herein provided a statistically appropriate method for reducing the large number of environmental variables aggregated for this study to a smaller set that accounted for a large proportion of the variation in the aquatic community data. For example, as much as 91 percent of the community variation was accounted for along the first axis of the fish urban-gradient MLR analysis (table 8). Many of the variables in the MLR analysis also proved to be significantly related ($p<0.05$) to one or more axes in the pCCA (tables 11, 12, and 13), improving ecological interpretation.

## SUMMARY AND CONCLUSIONS

This paper documents the statistical relations between environmental characteristics and aquatic-assemblage structure along an urban-land-use gradient in 36 New Jersey streams. Anthropogenic alterations of the landscape commonly associated with population growth and urban development have been linked consistently to the deterioration of aquatic-assemblage structure and appear to contribute to the degradation of many streams throughout New Jersey. Multivariate statistical methods, including PCA, DCA, MLR, and pCCA, were used to identify important environmental gradients and evaluate the anthropogenic factors that accounted for a significant proportion of variability in aquatic-assemblage structure. More than 400 environmental variables were summarized and variability associated with natural landscape factors such as latitude, elevation, and physiography was reduced or eliminated during the stratification and analytical process. Results of this study particularly emphasize the importance of landscape features evaluated at multiple spatial scales and anthropogenic effects associated with population growth, urban expansion, and habitat modifications along the stream channel. A multiple community approach was especially valuable in differentiating among fish, aquatic macroinvertebrate, and algal community responses to anthropogenic disturbance. Communities respond differently to perturbations in the environment and assessing community responses at different trophic levels helped provide a more complete evaluation of the effects of increasing urbanization in New Jersey watersheds. Differences in disturbance thresholds among fish, invertebrate, and algal assemblages indicate a disparity in response to multiple environmental stressors. Moderate to severe impairment was detectable in New Jersey streams when impervious surface cover in the drainage basin exceeded approximately 18 percent; however, high index (IBI, NJIS, and TDI) variability at the low end of the urban gradient obscured appropriate discrimination.

Partial canonical correspondence analysis proved to be an appropriate and responsive direct ordination technique for evaluating unimodal relations between aquatic-community structure and the environmental variables of interest after the effects of the natural environmental covariables were partialed out. This approach allowed a detailed examination of the anthropogenic factors affecting fish, invertebrate, and algal communities along an ULUG without the confounding effects of natural environmental variability, which weaken such analyses and result in ambiguous and less meaningful interpretations of community relations. Although a strong environmental framework with appropriate stratification of sampling sites is important in any study, it does not preclude the need to reduce or eliminate additional sources of natural environmental variability associated with altitude, longitude, and drainage area from unimodal-based community analyses.

A significant proportion of the community variation was accounted for by variables that were inversely related to increasing urban land use. The area of forest and wetlands in the watershed was positively related to the health and condition of aquatic invertebrate communities. These land uses help maintain favorable environmental conditions for disturbance-intolerant and highly desired species and protect surface-water quality by providing a natural filter for sediment and nutrients. Thus, forest and wetlands help mitigate many of the adverse effects of other human-induced landscape alterations. The presence of high-quality cobble substrate was important for all aquatic communities studied. It accounted for a significant amount of the variation in many of the MLR models evaluated and was significantly correlated to one or more axes in the pCCA.

These relations indicate that proactive measures such as protecting remaining tracts of forested land and wetlands, minimizing disturbance to streambanks, and reducing sediments in urbanizing basins may mitigate some anthropogenic effects. No single approach, however, is sufficient to address the full range of watershed effects resulting from urbanization. Identifying specific environmental perturbations and integrating a suite of management measures (for example, stabilizing streambanks, maximizing infiltration, reestablishing and widening riparian buffers, reconnecting streams with their floodplains, and preserving forests, wetlands, and riparian corridors) also are important.

The results of this study indicate that integrated management measures that temper the effects of impervious surface and other results of human activity may be useful in minimizing the continued deterioration of the quality of water and the condition of aquatic communities in New Jersey streams. The results of this and other related studies currently are being used in New Jersey to help prioritize approaches that integrate preventative, restorative, and other mitigative measures.

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