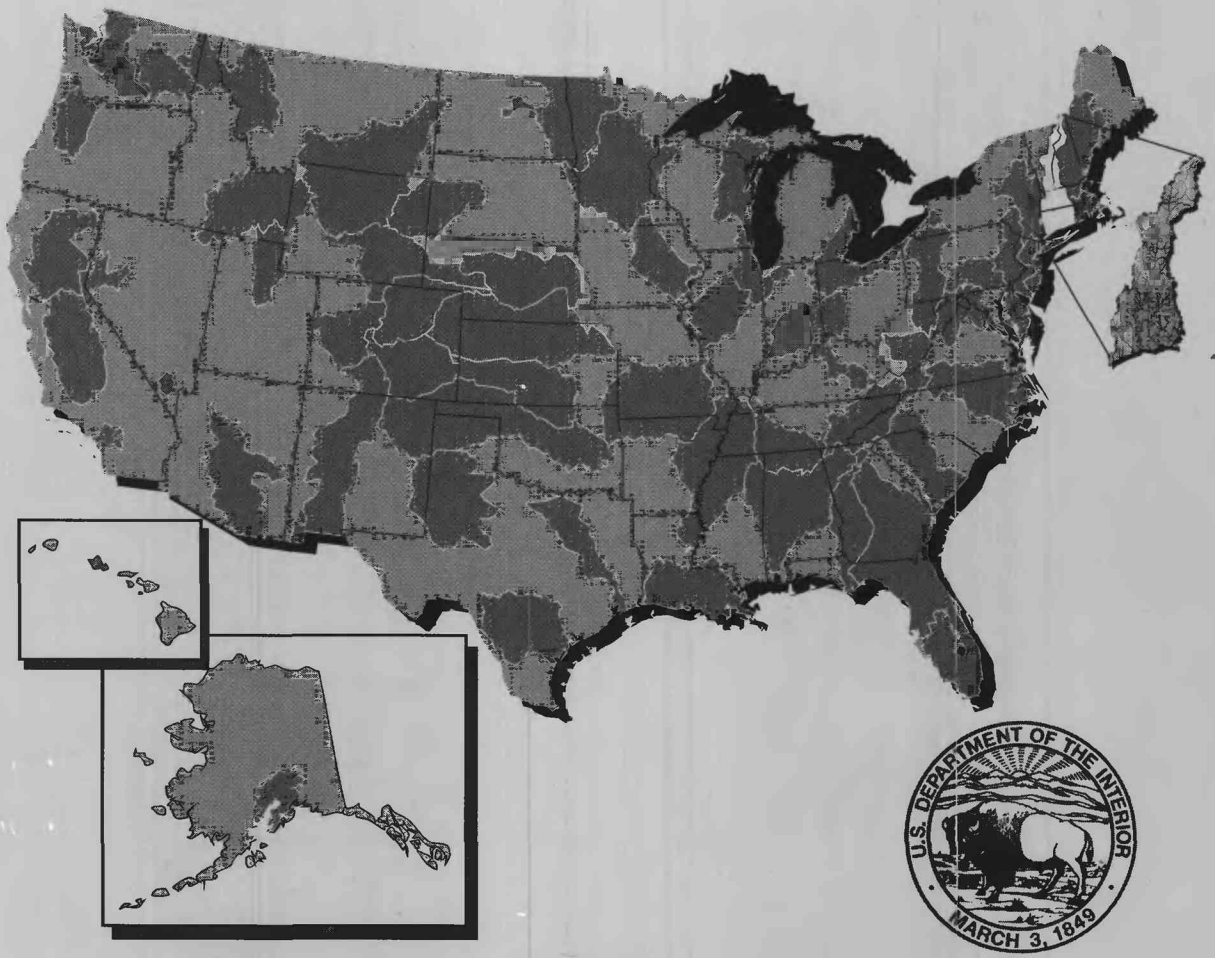
Water-Quality Assessment of the Connecticut, Housatonic, and Thames River Basins Study Unit: Analysis of Available Data on Nutrients, Suspended Sediments, and Pesticides, 1972-92

U.S. Geological Survey
Water-Resources Investigations Report 95-4203

National Water-Quality Assessment Program



Water-Quality Assessment of the Connecticut, Housatonic, and Thames River Basins Study Unit: Analysis of Available Data on Nutrients, Suspended Sediments, and Pesticides, 1972-92

By MARC J. ZIMMERMAN, STEPHEN J. GRADY, ELAINE C. TODD TRENCH, SARAH M. FLANAGAN, and MARTHA G. NIELSEN

U.S. Geological Survey
Water-Resources Investigations Report 95-4203

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FOREWORD

The mission of the U.S. Geological Survey (USGS) is to assess the quantity and quality of the earth resources of the Nation and to provide information that will assist resource managers and policymakers at Federal, State, and local levels in making sound decisions. Assessment of water-quality conditions and trends is an important part of this overall mission.

One of the greatest challenges faced by water-resources scientists is acquiring reliable information that will guide the use and protection of the Nation's water resources. That challenge is being addressed by Federal, State, interstate, and local water-resource agencies and by many academic institutions. These organizations are collecting water-quality data for a host of purposes that include: compliance with permits and water-supply standards; development of remediation plans for specific contamination problems; operational decisions on industrial, wastewater, or water-supply facilities; and research on factors that affect water quality. An additional need for water-quality information is to provide a basis on which regional- and national-level policy decisions can be based. Wise decisions must be based on sound information. As a society we need to know whether certain types of water-quality problems are isolated or ubiquitous, whether there are significant differences in conditions among regions, whether the conditions are changing over time, and why these conditions change from place to place and over time. The information can be used to help determine the efficacy of existing water-quality policies and to help analysts determine the need for and likely consequences of new policies.

To address these needs, the U.S. Congress appropriated funds in 1986 for the USGS to begin a pilot program in seven project areas to develop and refine the National Water-Quality Assessment (NAWQA) Program. In 1991, the USGS began full implementation of the program. The NAWQA Program builds upon an existing base of water-quality studies of the USGS, as well as those of other Federal, State, and local agencies. The objectives of the NAWQA Program are to:

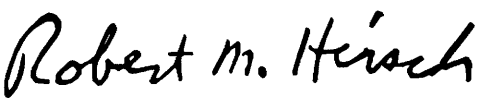
- Describe current water-quality conditions for a large part of the Nation's freshwater streams, rivers, and aquifers.
- Describe how water quality is changing over time.
- Improve understanding of the primary natural and human factors that affect water-quality conditions.

This information will help support the development and evaluation of management, regulatory, and monitoring decisions by other Federal, State, and local agencies to protect, use, and enhance water resources.

The goals of the NAWQA Program are being achieved through ongoing and proposed investigations of 60 of the Nation's most important river basins and aquifer systems, which are referred to as study units. These study units are distributed throughout the Nation and cover a diversity of hydrogeologic settings. More than two-thirds of the Nation's freshwater use occurs within the 60 study units and more than two-thirds of the people served by public water-supply systems live within their boundaries.

National synthesis of data analysis, based on aggregation of comparable information obtained from the study units, is a major component of the program. This effort focuses on selected water-quality topics using nationally consistent information. Comparative studies will explain differences and similarities in observed water-quality conditions among study areas and will identify changes and trends and their causes. The first topics addressed by the national synthesis are pesticides, nutrients, volatile organic compounds, and aquatic biology. Discussions on these and other water-quality topics will be published in periodic summaries of the quality of the Nation's ground and surface water as the information becomes available.

This report is an element of the comprehensive body of information developed as part of the NAWQA Program. The program depends heavily on the advice, cooperation, and information from many Federal, State, interstate, Tribal, and local agencies and the public. The assistance and suggestions of all are greatly appreciated.



Robert M. Hirsch
Chief Hydrologist

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PLATE

[Plate is in pocket]

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CONVERSION FACTORS, VERTICAL DATUM, AND ABBREVIATED WATER-QUALITY UNITS

CONVERSION FACTORS

Multiply	By	To obtain
acre	4,042	square meter
acre-foot (acre-ft)	1,233	cubic meter
cubic foot per second (ft^3/s)	0.02832	cubic meter per second
foot (ft)	0.3048	meter
inch (in.)	25.4	millimeter
inch per year (in/yr)		
mile (mi)	1.609	kilometer
million gallons per day (Mgal/d)	0.04381	cubic meter per second
pound (lb)	0.4536	kilogram
pound per square mile (lb/mi^2)	0.1753	kilogram per square kilometer
square mile (mi^2)	2.590	square kilometer
ton per square mile (ton/mi^2)	0.3503	megagram per square kilometer
ton per year (ton/yr)	0.9072	megagram per year

Temperature in degrees Fahrenheit (^oF) can be converted to degrees Celsius (^oC) as follows:
$^oC = 5/9\ (^oF - 32)$.

VERTICAL DATUM

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

Milligram per liter (mg/L) is a unit expressing the concentration of a chemical constituent in solution as weight (milligram) per unit volume (liter) of water.

Microgram per liter (µg/L) is a unit expressing the concentration of a chemical constituent in solution as weight (microgram) of solute per unit volume (liter) of water.

Water-Quality Assessment of the Connecticut, Housatonic, and Thames River Basins Study Unit: Analysis of Available Data on Nutrients, Suspended Sediments, and Pesticides, 1972-92

By Marc J. Zimmerman, Stephen J. Grady, Elaine C. Todd Trench, Sarah M. Flanagan, *and* Martha G. Nielsen

Abstract

This retrospective report examines available nutrient, suspended sediment, and pesticide data in surface and ground water in the Connecticut, Housatonic and Thames Rivers Study Unit of the National Water-Quality Assessment Program. The purpose of this study is to improve the understanding of natural and anthropogenic factors affecting water quality in the study unit. Water-quality data were acquired from various sources, primarily, the U.S. Geological Survey and the U.S. Environmental Protection Agency. The report examines data for water years 1972-92, focusing on 1980-92, although it also includes additional data from as early as 1905.

The study unit lies within the New England Physiographic Province and altitudes range from sea level in coastal Connecticut to 6,288 feet above sea level at Mount Washington, New Hampshire. Two major aquifer types underlie the study unit—unconsolidated glacial deposits and fractured bedrock. The climate generally is temperate and humid, with four distinct seasons. Average annual precipitation ranges from 34 to 65 inches. The study unit has a population of about 4.5 million, which is most highly concentrated in southwestern Connecticut and along the south-central region of the Connecticut River Valley.

Surface-water-quality data were screened to provide information about sites with adequate numbers of analyses (50) over sufficiently long periods (1980-90) to enable valid statistical analyses. In order to compare effects of different types of land use on surface-water quality, examination of data required application of several statistical and graphical techniques, including mapping, histograms, boxplots, concentration-discharge plots, trend analysis, and load estimation. Spatial and temporal analysis of surface-water-quality data indicated that, with a single exception, only stations in the Connecticut water-quality network had sufficient data collected over adequately long time periods to use in detailed analyses.

Ground-water nutrient and pesticide data were compiled from several Federal and State agencies, primarily the U.S. Geological Survey, U.S. Environmental Protection Agency, and Connecticut Department of Health Services. Nutrient data were available for several thousand wells; nitrite plus nitrate as nitrogen was the most commonly reported constituent. Most wells with nutrient data are in Massachusetts and Connecticut.

Relative to nutrient data in ground and surface water, pesticide data are less common. Pesticide data were available for slightly more than 200 surface-water sites and less than 500 wells; about 95 percent of the wells are completed in

stratified-drift or till aquifers. Data for 81 pesticide compounds were available in various data bases. 2,4-D and silvex were the most commonly detected herbicides in surface water and dieldrin and diazinon were the most commonly detected insecticides. Most surface-water pesticide samples and detections are from bed sediment, but much of the data are not recent.

Ethylene dibromide (EDB), a soil fumigant used in tobacco farming was detected in 268 wells in a 50 square-mile area of north-central Connecticut; EDB contamination also was detected in wells in Massachusetts. Atrazine, an herbicide commonly used in corn farming, commonly was detected in wells installed in tilled agricultural fields. Corn herbicides were commonly detected in the northern part of the study unit, although the sampling has been less frequent than in the southern part of the study unit. Pesticides were seldom detected in public-supply wells in Connecticut.

Urban sites with relatively high population densities and high concentrations of dischargers were characterized by having the highest nutrient concentrations and loads when adjusted for differences in drainage area or population. Particularly, the Pequabuck, Naugatuck, and Quinnipiac River Basins were characterized by high nutrient concentrations—median total nitrogen concentrations ranged from 3.3 to 4.2 mg/L (milligrams per liter) and median total phosphorus concentrations ranged from 0.42 to 0.8 mg/L. In contrast, the predominantly forested and low density residential land-use sites, such as Saugatuck and Salmon River Basins, were characterized by low nutrient concentrations—median total nitrogen ranged from 0.50 to 0.60 mg/L and median total phosphorus concentrations ranged from 0.01 to 0.02 mg/L. Estimated total nitrogen loadings in median discharge years ranged from 940 kilograms per square mile at the Salmon River near East Hampton, Conn., to 5,800 kilograms per square mile at the Naugatuck River at Beacon Falls, Conn. Water quality, in terms of nutrient concentrations and areally adjusted loadings, for sites with large drainage basins integrating a wide variety of land-use categories fell between the extremes of the urban and forested sites—total nitrogen was 1,400 kilograms per square mile per year at the Connecticut River at Thompsonville, Conn.

Nitrate concentrations in ground water occasionally exceeded the safe drinking-water standard of 10 mg/L as nitrogen. The greatest number of detections exceeding the standard, however, were not in public-water supplies but in shallow observation wells in agricultural settings (the most frequently sampled type of well). None of the public-supply wells in Massachusetts exceeded the standard. Although nitrate concentrations for Vermont and New Hampshire generally were low, few data were available and those were seldom reported on the basis of drainage basin, making analysis difficult.

Trend analysis indicated that flow-adjusted concentrations of total and dissolved phosphorus generally decreased during the period of analysis, however, total nitrogen did not change substantially. Decreases in ammonia concentrations with time were usually accompanied by increases in nitrate, suggesting improvements in sewage treatment.

The lack of adequate data from more or less exclusively agricultural areas points to the need for further study of the effects of farming on surface-water quality in the study unit. Furthermore, additional information is needed on the rates, transformations, and movements of nutrients and other materials through and between the aquatic and terrestrial components of the study unit.

INTRODUCTION

Water quality in the Connecticut, Housatonic, and Thames Rivers study unit (fig. 1) has been adversely affected by various human activities in urban, residential, industrial, commercial, agricultural, and recreational areas. Historically, waste effluents from population centers and industries have created some of the most serious water-quality problems on major streams. A growing concern is the degradation of water quality in small streams affected primarily by nonpoint pollution sources such as residential septic systems, storm runoff, and agricultural areas. Surface water, streambed sediments, and ground water at some locations in the study unit have been contaminated by nutrients, bacteria, pesticides, trace metals, and synthetic organic chemicals. The presence of these contaminants may affect the suitability of water for drinking, industrial use, recreation, or aquatic life. The quality of fresh water in the study unit is well-documented in some areas and unknown or minimally documented in others.

The quality of fresh water in the study unit also affects Long Island Sound, which is the receiving water for all streams in the study unit. The waters of Long Island Sound are affected by streamflow, point discharges, and nonpoint discharges from Westchester County, New York; the New York City area; the northern part of Long Island, New York; and the Connecticut, Housatonic, and Thames River Basins study unit. The movement of nutrients, especially nitrogen, from these areas into the waters of Long Island Sound is a major regional concern. Excessive nitrogen in the water of Long Island Sound promotes excessive growth of aquatic algae. When these plants die, they decompose, consuming the oxygen dissolved in the water and contributing to a condition called hypoxia, or low dissolved oxygen. Hypoxia adversely affects the fish and invertebrates of the Sound in various ways. State and Federal agencies have attempted to determine the sources of nutrients that have impaired the Sound's water quality. However, the relative contributions of different geographic areas and land uses are not adequately understood in some cases.

Purpose and Scope

The purposes of this retrospective report for the study period 1972-92 are to (1) document existing nutrient, suspended-sediment, and pesticide data in the Connecticut, Housatonic, and Thames Rivers study unit; (2) provide a data base for the first National scale systematic analysis of nutrient, suspended-sediment, and pesticide data; (3) use existing data to develop a conceptual model of the spatial and temporal patterns of the concentrations and loads, where possible, of nutrients, suspended-sediment, and pesticides in the surface- or ground-water resources in the study unit; (4) quantify nutrient loads leaving the study unit; and (5) guide future data collection by demonstrating where additional coverage is needed, identifying appropriate conditions for synoptic surveys or case studies, and suggesting how analytical coverage of nutrient, suspended-sediment, and pesticide constituents could be augmented or modified.

From among the large number of aquatic chemical constituents, this report focuses on nitrogen and phosphorus—essential plant nutrients occurring in several chemical forms that may affect their availability for uptake by aquatic plant life. Nitrogen in ground water also was studied. This report also presents information on suspended sediment, sometimes referred to as suspended particulate matter, which is associated with the movement of nutrients, pesticides, bacteria, metals, and organic compounds.

In addition, the report examines the available data on the occurrence and distribution of pesticide compounds in surface and ground water, as well as in the streambed sediments, of the study unit. Information is presented on any pesticide compounds present in the historical data from the various sources inventoried for this effort.

Acknowledgments

The authors wish to thank all those who have facilitated this effort by providing information and water-quality data. Dr. Paul Godfrey, Director, University of Massachusetts, Water Resources Research Center, provided the data from the Acid Rain Monitoring Program and reviewed our analysis of these data. Dr. Gerald Iwan, Chief of the Water Supply Section, Connecticut Department of Health Services,

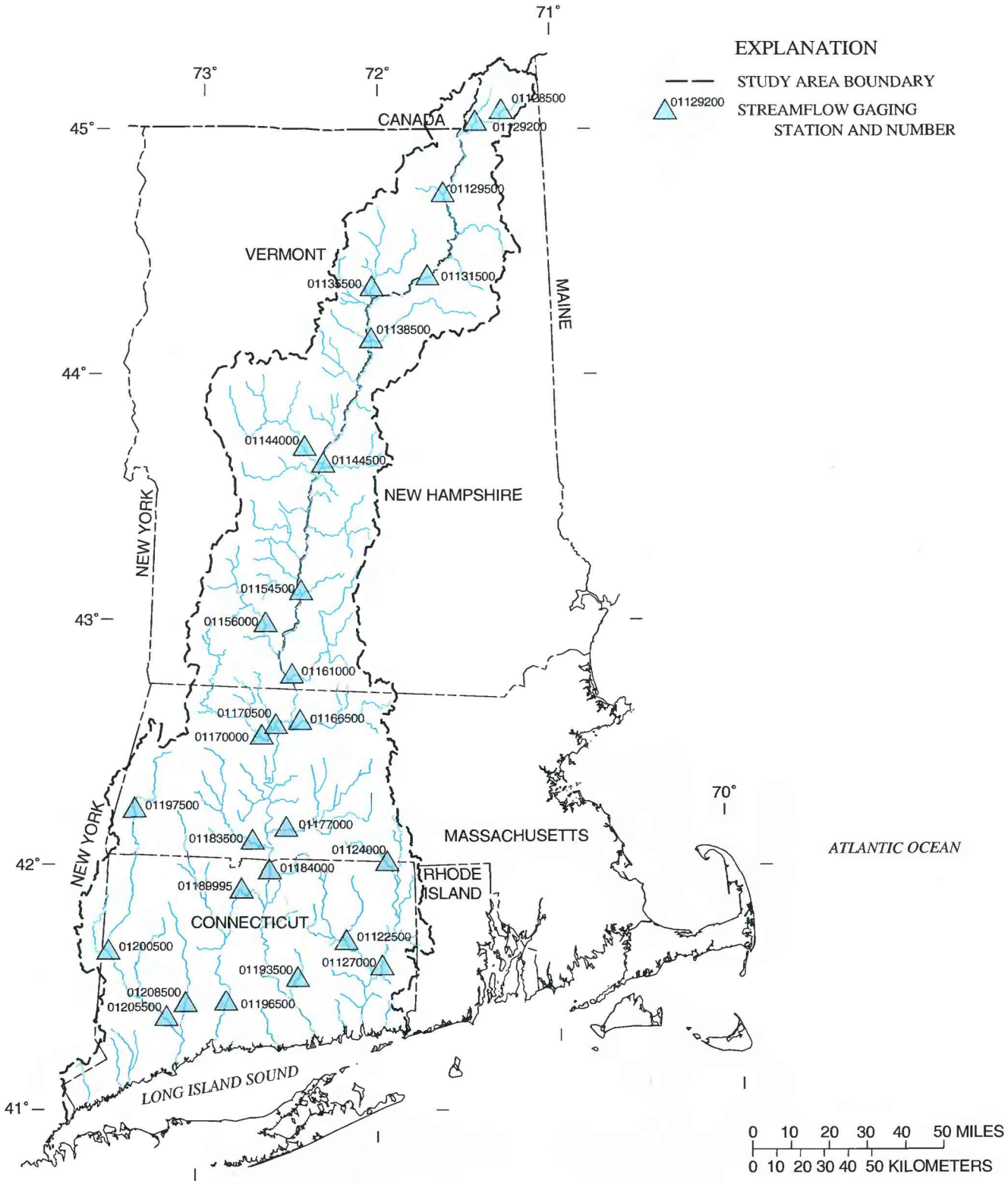


Figure 1. Location of principal rivers, major tributaries, and selected streamflow-gaging stations in the Connecticut, Housatonic, and Thames River Basins study unit.

and regional engineers Paul Ritsick, Robert Rivard, and Michael Hage assisted by providing access to records for public-supply wells and reservoirs. Dr. Stephen P. Garabedian, Project Chief of this study, performed innumerable tasks facilitating the writing of this report and also contributed text to the study unit description. Peter Steeves, U.S. Geological Survey (USGS), Marlborough, Massachusetts, provided much of the cartography and Geographic Information Systems (GIS) analysis. Joseph Nielsen, USGS, Augusta, Maine, compiled hydrological data used in description of the study unit. Christopher Hudon, USGS, Hartford, Connecticut, inventoried the Connecticut Department of Health Services paper files and created a computerized data base on pesticide samples. Laura Medalie and Marilee A. Horn, USGS, Bow, New Hampshire, supplied water-use data. Jacob Tinus, Volunteer for Science, USGS, Marlborough, Massachusetts, compiled and described pesticide data from the State of Vermont. Keith Saxon and Teige Davidson, USGS, Marlborough, Massachusetts, assisted with graphics.

DESCRIPTION OF STUDY UNIT

The Connecticut, Housatonic, and Thames Rivers study unit comprises an area of almost 16,000 mi^2, extending from Canada to coastal Connecticut (fig. 1). The boundaries of the study unit are defined by the drainage divides separating the Connecticut and Housatonic River Basins from the Lake Champlain-St. Lawrence River and Hudson River Basins on the west, and by the drainage divides separating the Connecticut and Thames River Basins from the Androscoggin, Saco, Merrimack, Blackstone, and Pawcatuck River Basins on the east. The study unit includes a 114 mi^2 segment of the Province of Quebec, Canada, 3,928 mi^2 of eastern Vermont, 3,047 mi^2 of western New Hampshire, 3,490 mi^2 in west-central Massachusetts, nearly all (4,853 mi^2) of Connecticut, and small parts of New York (263 mi^2) and Rhode Island (63 mi^2)—an area of 15,758 mi^2.

In addition to the multi-State drainage basins of the Connecticut, Housatonic, and Thames Rivers, the study unit also includes small coastal drainage basins in Connecticut. Major streams in the study unit generally flow from north to south and all eventually drain into Long Island Sound at the southern border of Connecticut. The study unit encompasses varied land uses from forested wilderness area in the north to densely populated cities in the south. Southern parts of the study unit include some of the country's oldest industrial areas.

The study unit is less than one-half of 1 percent of the total area of the Nation. In 1990, the population was about 4.5 million people, or about 2 percent of the Nation's population (U.S. Bureau of the Census, 1991). The study unit is characterized by diverse population density (fig. 2), ranging from sparsely populated (less than 25 persons per square mile), rural agrarian, and wilderness areas of northern Vermont and New Hampshire to densely populated (more than 1,500 persons per square mile), urban areas of southwestern Connecticut and the south-central part of the Connecticut River Valley. The major urban centers of Springfield, Mass., and Hartford, New Haven, and Bridgeport, Conn., are all in the southern part of the study unit.

Land Use

The land-use development of New England has affected the current pattern of land use throughout the study unit. The landscape was colonized and broken up into small farms and small, closely spaced town centers more than 200 years ago. During the intervening years, the number of farms has decreased (with a concurrent increase in forested land), but the size of the farms has not substantially changed. This contrasts with land use in the Western and Midwestern United States, where the size of individual land holdings is very large and towns are widely spaced. Thus, the current pattern of agricultural, forested, and urban land use in the study unit resembles a patchwork with small pieces and the environmental settings comprise heterogeneous mixtures of land use.

Information on the pattern of land use and the history of development of a region is important to developing an understanding of spatial and temporal changes in water quality. Land use across the study unit reflects physiography, geology and soil types, and hydrography. Coastal areas and river valleys were developed first, as inhabitants of the region from the earliest 17th century European colonists have depended on its harbors and navigable waterways for commerce and transportation, and on its tillable river valleys for agriculture. Large areas of the coastal lowlands, river valleys, and rolling upland terrain were progressively cleared of forests for expanding

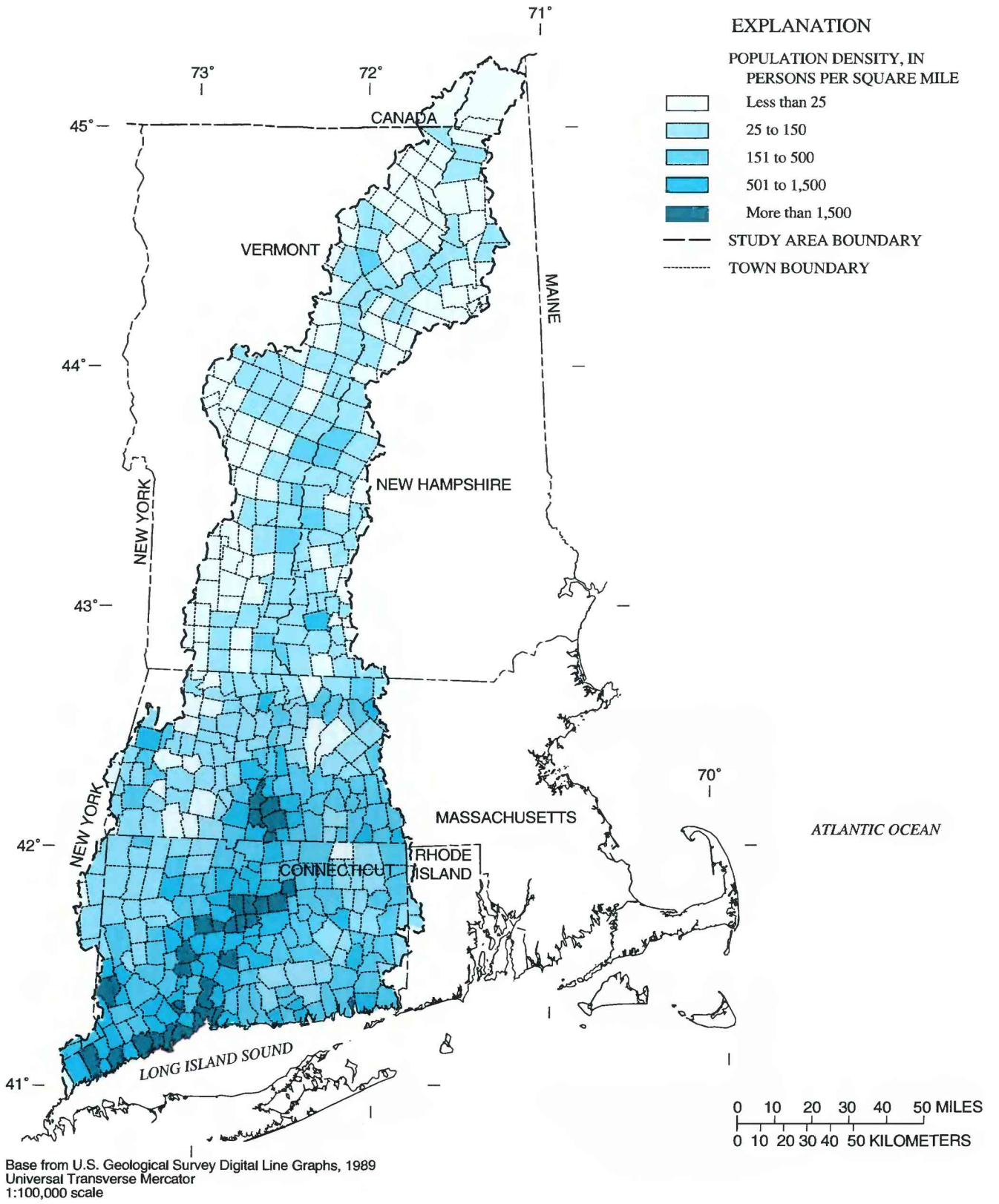


Figure 2. Population density by town in the Connecticut, Housatonic, and Thames River Basins study unit, 1990.

agricultural use through the middle of the 19th century. Water-powered factories of the industrial revolution brought swelling populations to "mill towns" that developed along many of the rivers draining the upland sections of the region. By 1880, 243 different industries flourished among the more than 200 Connecticut mill towns (Lewis and Harmon, 1986, p. 99). During the 20th century, manufacturing continued and expanded in the region but hydroelectric power replaced hydraulic power and industrialization and population growth centered on large metropolitan areas at the expense of the upland mill towns. In the last 20 years, the economy of the southern part of the study unit shifted from manufacturing to services and financial sectors. With the development of an improved highway network, population increased and land use changed mostly in suburban and rural areas. The pattern of land use resulting from these physical and historical factors is a mosaic of land uses interspersed at a scale of acres to tens of acres.

The generalized land-use data (fig. 3) have been aggregated to Level I categories (Anderson and others, 1976). Although outdated in some parts of the study unit, the USGS land-use/land-cover data, compiled at a scale of 1:250,000 during 1970-78 (U.S. Geological Survey, 1978a, 1978b; 1979a, 1979b, 1979c, 1979d, 1979e; 1980a, 1980b; 1981), represent the most current basinwide information available. The land-use data are contemporaneous with water-quality data collected during 1972-79.

Overall, 78 percent of the land in the study unit can be classified as undeveloped, including a relatively small part (4 percent) of the area comprising surface-water bodies, wetlands, or other barren land. A large part of the area (74 percent) remains forested (fig. 3). The largest expanse of forest cover is in the northern part of the study unit in New Hampshire and Vermont. Forests cover 84 percent of the upper Connecticut River Basin, where evergreen or mixed evergreen/deciduous forest is the principal forest type. Silviculture for lumber and paper products and recreational use are the principal human activities associated with the forested lands in this part of the study unit. In the more highly developed, southern parts of the study unit, deciduous forest or woodland still covers much of the area. Sixty-nine percent of the lower Connecticut River Basin is forested, mostly in the upland areas. Seventy-three percent of the Connecticut coast east of the Connecticut River (the area drained by the Thames, Quinebaug, and Shetucket Rivers and other smaller rivers discharging to Long Island Sound) is forested; whereas the area west of the Connecticut River (the area drained by the Housatonic, Naugatuck, and Quinnipiac Rivers plus smaller rivers discharging to Long Island Sound) is the least forested (58 percent) in the study unit.

Agricultural lands occupy 12 percent of the study unit (fig. 3), and include land in active, tilled cultivation (harvested and unharvested fallow, idle, or cover crops), pasture or other noncropland used for forage, and areas classified as "other" agricultural land (feedlots, farmsteads, and nonproductive farmland). Although the total amount of agricultural land use is decreasing (U.S. Water Resources Council, 1978a, 1978b) as formerly agricultural lands are developed for housing or industrial and commercial uses, or as unused farmland reverts to forest cover, the percentage of agricultural land use for food production is increasing. The major crops produced are corn, potatoes, tobacco, soy beans, fruit and vegetables, and lesser amounts of small grains, ornamental shrubs, and Christmas trees. About 2 percent of the harvested cropland is irrigated (U.S. Water Resources Council, 1978b). Dairy farming, once a principal agricultural enterprise, is decreasing throughout much of the region.

About 10 percent of the study unit is classified as urban (fig. 3), including residential, commercial, industrial, and institutional land uses and other developed lands (golf courses, parks, and cemeteries; highways, airports, and utility easements) in municipalities or metropolitan areas. Urban areas commonly contain high-density, residential land (greater than 150 persons per square mile). The percentage of urban land use varies greatly across the study unit. Twenty-two percent of the Connecticut coast west of the Connecticut River is urban, compared to only 9 percent of the coast east of the Connecticut River. The southern part of the Connecticut River Basin includes 12 percent urban land use, however, the upper Connecticut River Basin has only 2 percent.

In the major urban areas of the region, water is supplied by public and private utilities and is commonly imported from more rural areas. Sanitary-sewer collection systems convey wastewater to treatment plants that discharge into major streams and tributaries or into coastal waters. The use of individual wells for domestic-water supplies and on-site sewage-disposal systems (septic tanks and cesspools) is much more prevalent in nonurban areas.

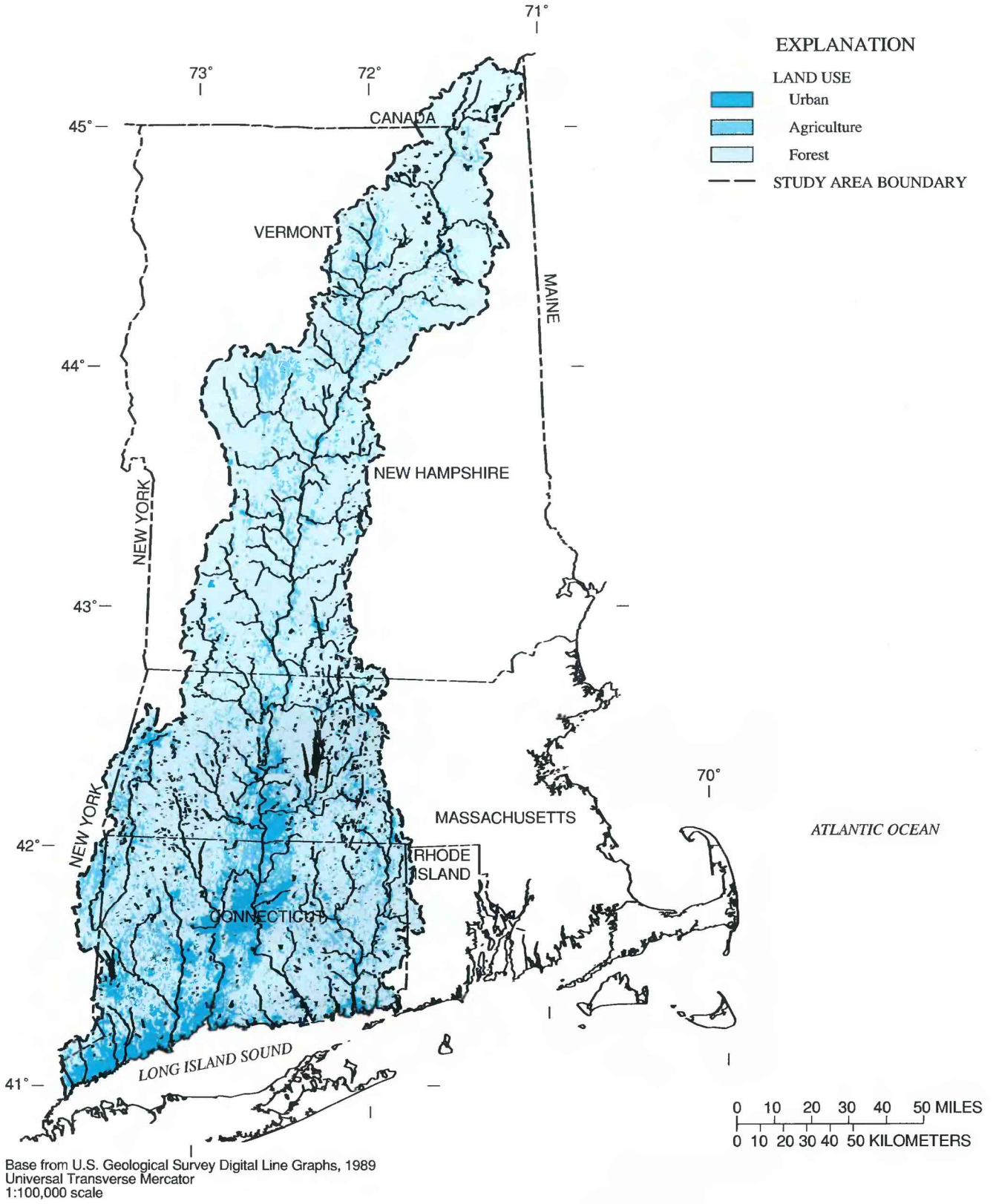


Figure 3. Generalized land use in the Connecticut, Housatonic, and Thames River Basins study unit.

Environmental Settings

The study unit is located entirely in the New England Physiographic Province (Fenneman, 1938), a plateau-like upland that rises gradually from the sea but includes numerous mountain ranges and individual peaks. Altitudes range from sea level in coastal Connecticut to 6,288 ft above sea level at the peak of Mount Washington in the White Mountains of New Hampshire.

The environmental setting classification in this study uses Fenneman's (1938) physiographic classification in conjunction with bedrock geology to define five major environmental settings—New England Upland, Connecticut Valley Lowland, Housatonic Valley Lowland, Mountain Upland, and Coastal Lowland (fig. 4).

New England Upland

The largest environmental setting in the study unit is the New England Upland. This environmental setting corresponds closely to the New England Upland Physiographic Province, but includes upland areas of the Taconic Section and excludes valley lowlands underlain by carbonate-rich bedrock. The New England Upland is characterized by hilly topography, small, narrow valleys, and isolated monadnocks of crystalline bedrock. Local topographic relief can exceed 1,000 ft, but is typically on the order of several hundred feet. Crystalline igneous and metamorphic bedrock typify the bedrock geology, although small local areas are underlain by carbonate-rich metamorphic rocks. The glacial geology is mainly locally derived till, which for the most part, blankets the bedrock. River valleys are partly filled with generally coarse-grained stratified drift. The glacial till typically yields rocky, thin soils best suited for agricultural uses such as pasture and orchards. Locally, valley bottoms are cultivated for corn and other row crops. Forest is the dominant land cover for the entire environmental setting, although, locally, agricultural or urban land use may significantly affect water quality. In the northern part of the study unit, the land is dominated by forest developed only for logging. There are no major urban areas in this environmental setting. The principal urban centers include small cities such as Waterbury, Conn., Keene, N.H., and St. Johnsbury, Vt.

Connecticut Valley Lowland

A subarea within the New England Upland, the Connecticut Valley Lowland consists of the wide valley bottom in west-central Massachusetts and central Connecticut underlain by Mesozoic clastic sedimentary bedrock, primarily arkosic sandstones and siltstones, interlayered with igneous basalts. Stratified drift, consisting mainly of extensive fine-grained lake-bottom sediments that are capped by or interfinger with scattered sands and gravels, covers the Mesozoic rocks in the lowland. The landscape is characterized by a broad, flat bottomland with occasional prominent bedrock ridges of basalt rising above the valley bottom. Urban and agricultural land uses predominate in this environmental setting. Agricultural practices are intensive, with heavy use of fertilizers and pesticides on crops of corn, tobacco, potatoes, and other row crops. Much of the valley that historically was agricultural land has been converted to urban and suburban areas. The metropolitan areas of Springfield and Northampton, Mass., and Hartford, Meriden, and New Haven, Conn., are located in this setting.

Housatonic Valley Lowland

The Housatonic Valley Lowland lies in the southwestern part of the study unit. This environmental setting incorporates valley areas of the Taconic Section and New England Upland that are underlain by a carbonate-rich bedrock. Most of the carbonate-rich bedrock in the study unit is in the Housatonic Valley and consists of a heterogeneous mixture of carbonate metamorphic and sedimentary rocks, and igneous and other metamorphic rocks. Mostly, coarse-grained stratified drift fills the valley bottom. The topography consists of broad valleys interrupted by steep hills. Substantial areas of agricultural land are in the northern or central parts of the valley. Urbanization has claimed much of the southern parts of the valley and the northern and central parts of the valley are a heterogeneous mixture of forested land, small agricultural fields, and small urbanized areas. Major urban centers in the valley include Pittsfield, Mass., and Danbury, Conn.

Mountain Upland

The Mountain Upland environmental setting consists of areas in the study unit in the Green and White Mountains Physiographic Provinces. The Green Mountains of Vermont are in the west-central part of

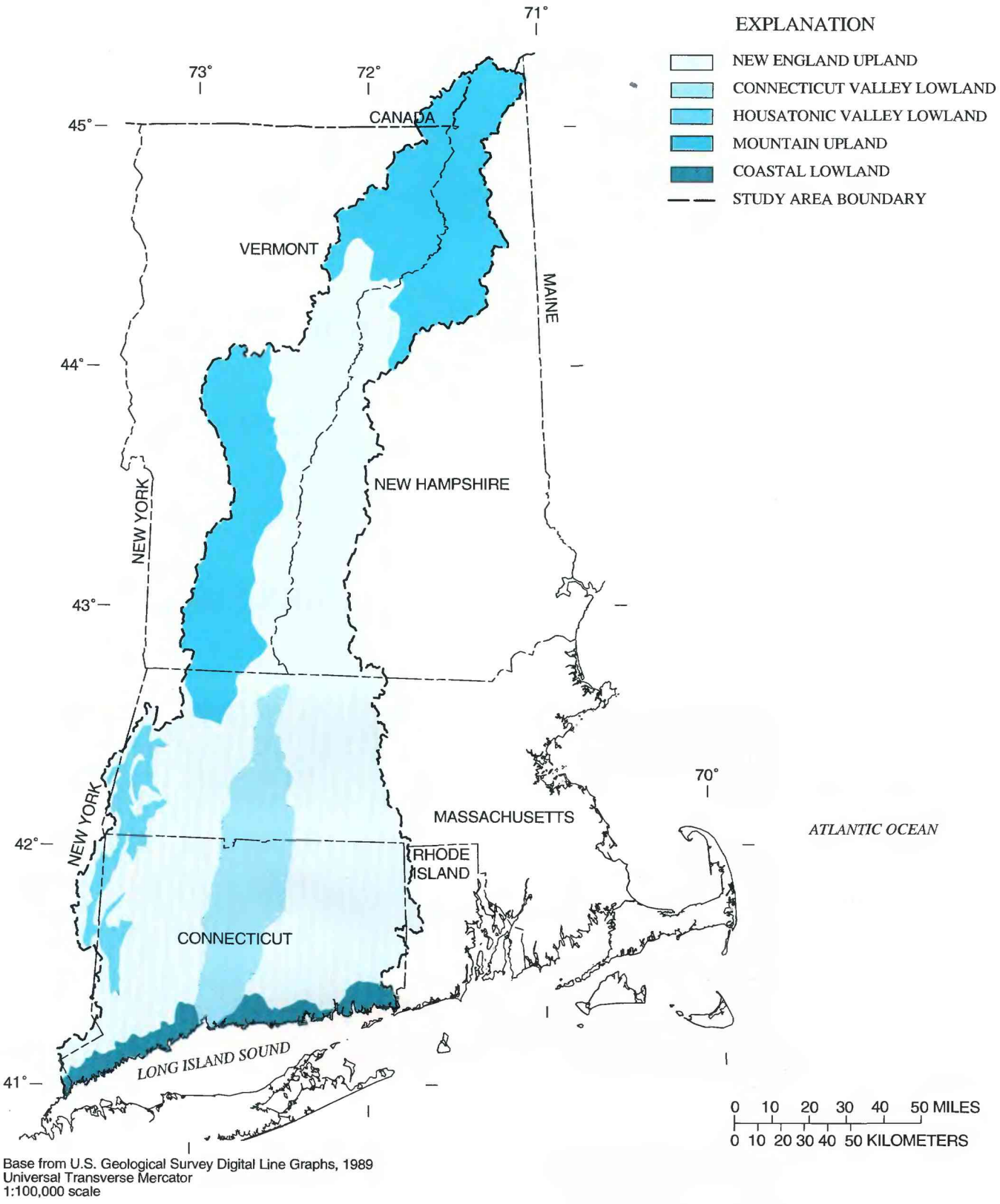


Figure 4. Environmental settings in the Connecticut, Housatonic, and Thames River Basins study unit.

the study unit and the White Mountains of New Hampshire lie in the northeastern part of the study unit. The mountain upland is formed by crystalline, igneous, and metamorphic bedrock that is resistant to weathering and erosion. Local topographic relief is as much as 4,500 ft near Mt. Washington, N.H. (the highest point in the Northeastern United States, with an altitude of 6,288 ft above sea level), and 3,000 ft near Killington Peak, Vt. Valleys are steep and narrow, while floodplains are rare. Coarse-grained stratified-drift deposits occur in the deeper valleys. Much of the area is pristine, undeveloped forest (except for local heavy logging activities in the national forests), although scattered small towns and built-up tourist areas do occur in some of the larger river valleys. A few of the mountains have major ski areas. The climate differs from the rest of the study unit in that winters are longer and colder, and precipitation is greater than in the New England Upland or any of the lowland settings.

Coastal Lowland

A strip of land generally 6 to 16 mi wide along the coast of Long Island Sound constitutes the Coastal Lowland environmental setting. Topographic relief generally is low and the geology is characterized by till overlying crystalline metamorphic bedrock. Stratified drift fills valleys that extend from the adjacent New England Upland. Population and land-use characteristics change substantially from west to east. On the west, closer to New York City, urban land uses dominate and population density is consistently high (see fig. 2). Major urban areas include Norwalk, Stamford, and Bridgeport, Conn. East of New Haven, the dominant land cover is forest and population density decreases considerably, with numerous, small, low-density suburban areas. New London, Conn., is in this environmental setting.

Soils

Soils in the study unit are highly variable in terms of depth, texture, and drainage characteristics, and have definite relations to topography, direction and angle of hillslopes, regional geology, and climate. Throughout much of the New England Upland and Seaboard Lowland sections, the most extensive soils are developed in acidic glacial till. Soils derived from glacial till are moderately to very stony. Their thickness and drainage characteristics are quite variable. Upland soils derived from till generally are not well suited for intensive agricultural development because of stoniness, shallow depth, or steep slopes. However, agricultural soils are productive in some upland areas of the study unit. Soils in floodplain areas generally are deeper, free of stones, and better suited for agricultural use.

Soils developed in glaciofluvial materials along major streams are deep, well to excessively drained, with sandy to gravelly loam horizons underlain by sand and gravel (Ilgen and others, 1966). The deep, well-drained, fine to medium textured soils of the Connecticut Valley Lowlands support the most intensive agriculture in the study unit. Developed from fine-grained glaciolacustrine sediments, Connecticut Valley Lowlands soils typically consist of 20 to 30 in. of a silty material over sandy loam. Soils in the Taconic, Green Mountain, and White Mountain sections are developed largely on schist or granite, or in glacial till derived from those rock types, and primarily are very stony and well drained. Most of the soils are moderately deep but shallow soils and rock outcrops are common. East of the Connecticut River the soils generally are acidic, but west of the river lime-bearing soils are more common. Soils derived from the carbonate rocks of the Housatonic River Valley are deep and well drained, and are exceptionally productive agricultural soils, containing higher concentrations of the nutrients calcium and magnesium than Connecticut Valley Lowland soils (Gonick and others, 1970).

Climate

The study unit is in the northeastern corner of the continental United States, from 41° and 45° north latitude, and lies in the path of prevailing westerly winds that alternately transport cool, dry, continental-polar, and warm, moist, maritime-tropical air masses into the region, resulting in frequent weather changes. Although the climate varies considerably in the study unit, it is generally temperate and humid with four distinct seasons. The climate is moderated by maritime influence along coastal sections, although a more extreme climate is characteristic of the northern, interior parts of the study unit. Average annual temperature ranges from less than 40°F in the northern mountainous areas to about 50°F in southwestern coastal Connecticut.

There is substantial areal variation in precipitation that reflects a pronounced orographic effect. Average annual precipitation ranges from about 34 in. in parts of the northern Connecticut River Valley to more than 65 in. on adjacent mountainous regions (Knox and Nordenson, 1955). The maritime influence provides for slightly greater precipitation amounts in places along coastal sections than falls just inland. Mean annual precipitation averages about 43 in. for the entire study unit.

Total annual precipitation is highly variable for any station in the study unit, commonly fluctuating annually by as much as 20 in. from the mean. Distribution of precipitation is relatively even on a monthly and seasonal basis, although some variation occurs. Mean monthly precipitation frequency is bimodal at most stations, and is the highest in November-December and March-April. Mean monthly precipitation typically is the lowest in May-June or February.

Prolonged periods of drought were severe in the study unit during 1929-36, 1939-45, and 1947-51. The most extensive drought to affect the region lasted from 1960 to 1969. The 1960's drought period greatly affected water supplies, agriculture, water quality, and economic activity.

Ground-Water Hydrology

Two principal types of aquifers underlie the study unit—unconsolidated glacial deposits and fractured bedrock. Thin (a few feet to a few tens of feet) deposits of alluvium may overlie glacial deposits and bedrock in major river valleys. These local deposits may yield water to wells but are not widely developed aquifers in this study unit. The unconsolidated glacial deposits include a generally thin (less than 15 ft), discontinuous layer of till—an unsorted mixture of boulders, gravel, sand, silt and clay that mantles the bedrock (Melvin and others, 1992a, 1992b). Till was deposited directly by active ice or through the melting and collapse of stagnant ice. The hydraulic properties of till generally are unsuitable for the development of substantial ground-water supplies and till is not a major aquifer in the study unit (Melvin and others, 1992a). Where glacial sediments were transported by meltwaters, the sediment was sorted and deposited in layers of similar grain size with hydraulic properties favorable for the development of large ground-water supplies. Although unevenly distributed (fig. 5), these stratified-drift aquifers along stream valleys and in lowland areas generally are the most productive sources of ground water in the study unit.

Unconsolidated stratified-drift aquifers store and transmit water through interconnected pores between individual grains of sediment. Differences in the thickness, extent, and permeability of stratified-drift aquifers, and their proximities to surface-water bodies that are sources of recharge, significantly affect the availability of water to wells.

Ground water in most stratified-drift aquifers and shallow bedrock aquifers occurs under unconfined conditions, and the water table generally is a few feet to several tens of feet below land surface. Recharge occurs directly from precipitation that falls on the land surface overlying the aquifer and infiltrates down to the water table, but more than one-half of the water entering glacial-valley aquifers may be derived from upland runoff (Morrissey and others, 1988). Surface water is an additional large potential source of recharge to aquifers. In many locations, pumping causes induced infiltration near surface-water bodies.

The size of ground-water-flow systems in the study unit generally is controlled by the topography (Randall and others, 1988). Flow is predominantly localized within shallow flow systems (less than 300 ft deep) between the topographic divide and perennial streams—typically, a few hundred to a few thousand feet.

Fractured bedrock aquifers underlie the entire study unit and are an important source of water for self-supplied domestic, commercial, and industrial users. Bedrock aquifers primarily store and transmit water through intersecting fractures in consolidated nonporous rock (Randall and others, 1988). Well yields depend on the number, size, and degree of interconnection of water-bearing fractures. Arkosic sedimentary bedrock of the Connecticut Valley Lowlands also has a primary intergranular porosity and carbonate bedrock of the Taconic Upland has a secondary solution porosity. These characteristics enhance the water-yielding properties of these bedrock aquifers that have a limited distribution in the study unit (fig. 6).

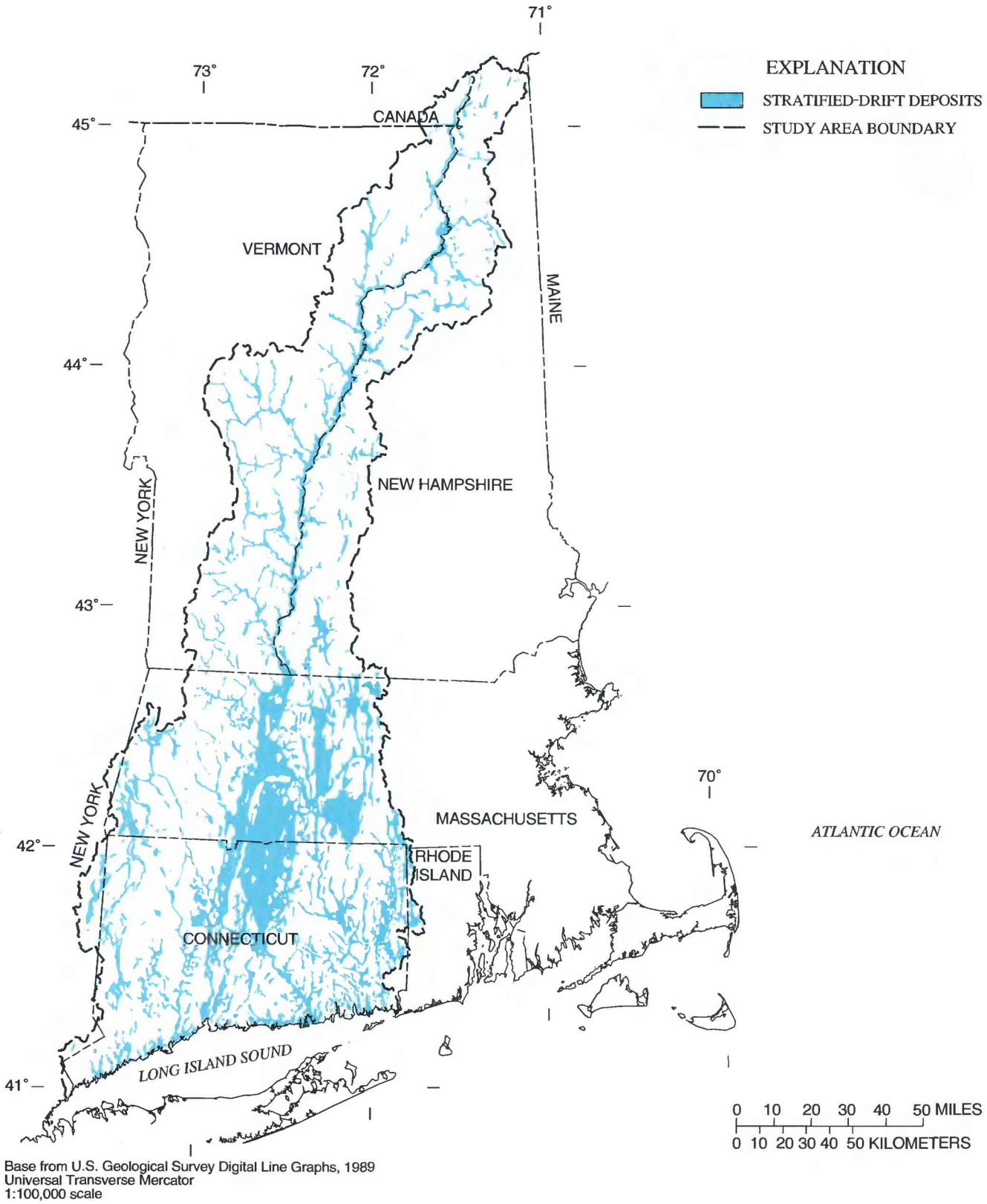


Figure 5. Distribution of stratified-drift deposits in the Connecticut, Housatonic, and Thames River Basins study unit. (Compiled from Goldthwait and others, 1951; Lang and others, 1960; Feininger, 1962; Stewart and MacClintock, 1970; Stone and others, 1985; Cadwell and Dineen, 1987; Cadwell, 1989, and Stone and others, 1992.)

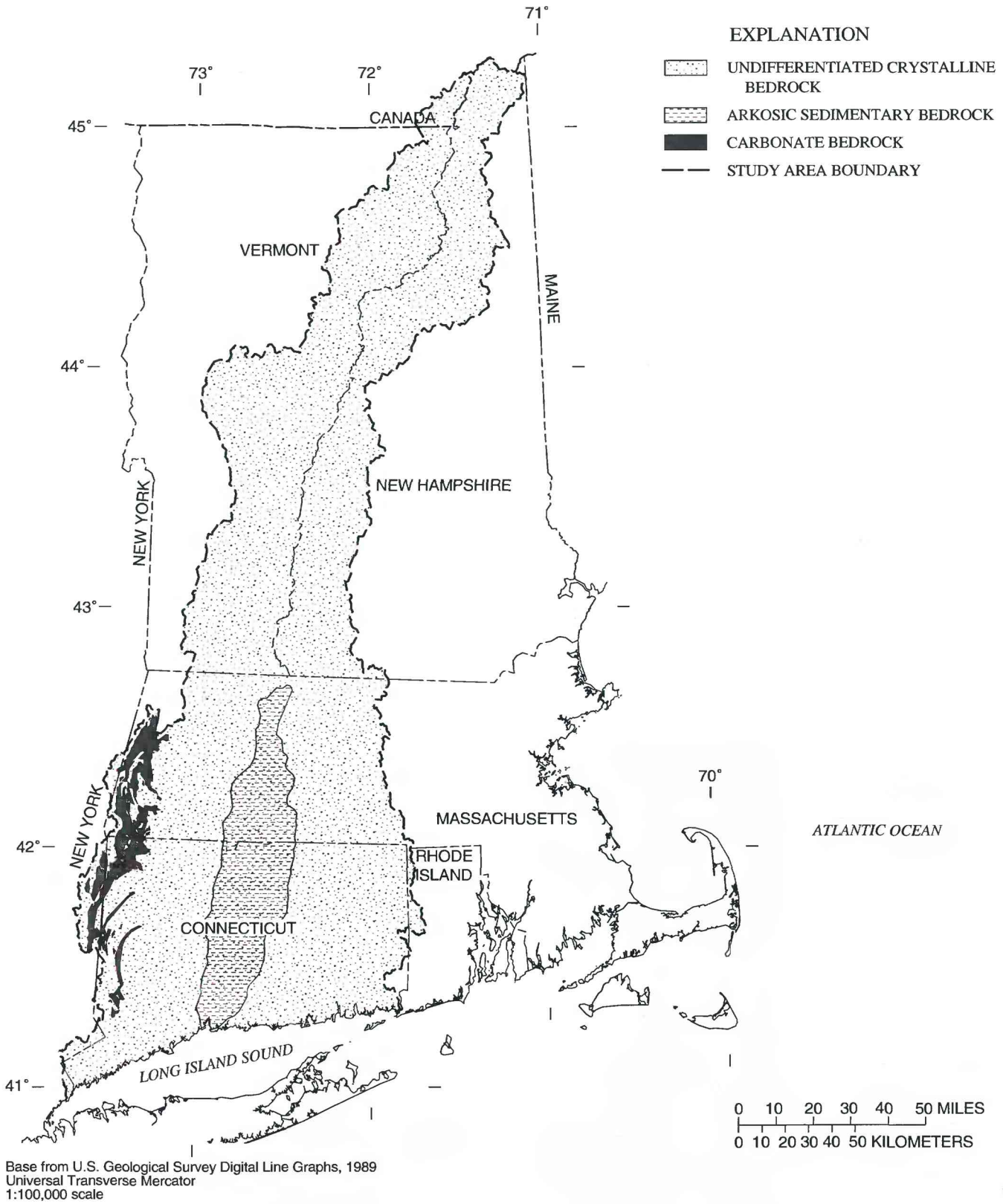


Figure 6. *Distribution of three general types of fractured-bedrock aquifers in the Connecticut, Housatonic, and Thames River Basins study unit.* (Compiled from Doll and others, 1961; Fisher and others, 1970; Quinn, 1971; Zen and others, 1983; Rogers, 1985; Lyons and others, 1986.)

Surface-Water Hydrology

The Connecticut River, the principal river in the study unit, flows about 400 mi from its source in the Connecticut Lakes of northern New Hampshire to its outlet at Long Island Sound (fig. 1). The Connecticut River drains 11,260 mi^2, or about 72 percent of the study unit. Average discharge of the Connecticut River at gaged sites ranged from 196 ft^3/s at First Connecticut Lake in New Hampshire (table 1) to 16,600 ft^3/s at Thompsonville near the Connecticut-Massachusetts border.

Annual mean discharge was high in the study unit during the mid-1950's and 1970's. Major floods have occurred along the Connecticut River in March 1936, September 1938, and August and October 1955 (Weiss and Cervione, 1986). Since the 1955 floods, flood-retention reservoirs have been built on many of the tributaries to the Connecticut River to reduce the flood peaks.

Average discharge for the major tributaries to the Connecticut River range from about 1,300 ft^3/s for the Deerfield River near West Deerfield, Massachusetts, to 632 ft^3/s for the West River at Newfane, Vermont (table 1). Other major streams in the study unit include the Housatonic and Thames Rivers, which together drain 3,420 mi^2, or about 20 percent of the study unit. The Housatonic River drains a basin area of 1,950 mi^2, is about 160 mi long, and has its headwaters in western Massachusetts. Average discharge in the Housatonic River at Stevenson, Connecticut, is 2,610 ft^3/s. The Thames River is tidal along its length, and discharge is measured for only its major tributaries: the Quinebaug, Shetucket, and Yantic Rivers. There are many flood and hydroelectric power impoundments in the Housatonic and Thames River Basins. Numerous other smaller streams and rivers that flow directly into Long Island Sound, most notably the Quinnipiac River, collectively drain 1,070 mi^2 in coastal parts of the study unit.

Average monthly discharge in the study unit generally peaks in the spring and there may be a secondary peak in late autumn to early winter (fig. 7). At locations on the Connecticut River, discharge peaks in April with a secondary peak in late autumn. Low streamflow conditions in the study unit generally occur from late summer to early autumn. Average monthly discharge for the Housatonic River at Stevenson, Connecticut, is highest in March and April.

Water Use

Total freshwater withdrawals in the study unit during 1990 were about 2,264 Mgal/d. Surface water is the dominant source, supplying 88 percent (1,996 Mgal/d) of the fresh water used. This surface-water use is subdivided into three geographic regions: the Connecticut River in Connecticut and Massachusetts, the Connecticut River in New Hampshire and Vermont, and the coastal rivers in Connecticut and Massachusetts, including the Housatonic, Thames, and Quinnipiac Rivers plus all others draining directly to Long Island Sound. In both Connecticut River regions, thermoelectric-power generation is the largest volume use of surface freshwater withdrawals—94.5 percent of surface-water withdrawals in New Hampshire and Vermont and 67.0 percent in Massachusetts and Connecticut (Healy and others, 1990; Simcox and others, 1990; U.S. Geological Survey, 1990a, 1990b). By contrast, public supply in the coastal river basins of the study unit is by far the largest volume use of fresh surface water.

Public supply, an important use throughout the study unit, is the dominant off-stream use in the Housatonic and other coastal river basins. An important interbasin transfer source of public-water supply for the Boston Metropolitan area is Quabbin Reservoir in the Chicopee River Basin, a tributary of the Connecticut River. Quabbin Reservoir, with a 1.2 million acre-ft capacity, is the largest impoundment or lake in the study area. The average flow diverted out of the study unit from Quabbin Reservoir for Boston's use during 1973-82 was 192 Mgal/d (Gadoury and Wandle, 1986). Currently, the average diversion from Quabbin Reservoir is 300 Mgal/d (R.A. Gadoury, U.S. Geological Survey, oral commun., 1993).

Minor uses of surface water in the study unit include the domestic, commercial, and agricultural categories. Agriculture has a minor impact on surface-water use in the study unit because of the abundance of precipitation during the growing season. There is relatively little use of surface water for irrigation in the study unit.

Ground water supplied only 12 percent (268 Mgal/d) of the freshwater withdrawals—primarily for public- and self-supplied domestic use. Nearly 75 percent of all ground-water withdrawals in the study unit were from aquifers in Connecticut. There are numerous towns in other States in the study unit where ground water is the only source of drinking water.

Table 1. Streamflow characteristics at selected streamflow-gaging stations in the Connecticut, Housatonic, and Thames Rivers study unit, water years 1905-91

[Data from the U.S. Geological Survey's National Information System. Site No.: See figure 1 for location of streamflow-gaging stations. Discharge for indicated exceedance probability: At sites affected by regulation, exceedance probability may be affected by changes in regulatory practices. mi^2, square mile; ft^3/s, cubic foot per second; in/yr, inch per year]

Station No.	Station name	Drainage area (mi^2)	Period of record (water years)	Average discharge (ft^3/s)	Average runoff (in/yr)	Standard deviation of average discharge	Coefficient of variation (percent)	Discharge for indicated exceedance probability (ft^3/s)	
								99 percent	1 percent
	Connecticut River								
01128500	Connecticut River at First Connecticut Lake, N.H.	83.0	1918-90	196	32.1	39.5	20.2	5.1	913
01129200	Connecticut River below Indian Stream, N.H.	254	1957-90	562	30.1	85.3	15.2	46.8	1,720
01129500	Connecticut River at North Stratford, N.H.	799	1931-90	1,580	26.9	256	16.2	208	9,170
01131500	Connecticut River near Dalton, N.H.	1,510	1928-90	2,890	26.0	486	16.8	407	17,000
01138500	Connecticut River at Wells River, Vt.	2,640	1951-90	4,930	25.4	921	18.7	454	24,100
01144500	Connecticut River at West Lebanon, N.H.	4,092	1912-76, 1980-90	7,120	23.6	1,420	19.9	333	39,300
01154500	Connecticut River at North Walpole, N.H.	5,493	1943-90	9,510	23.5	1,990	20.9	233	52,800
01170500	Connecticut River at Montague City, Mass.	7,860	1905-90	13,900	24.0	2,720	19.6	737	72,200
01184000	Connecticut River at Thompsonville, Conn.	9,660	1929-90	16,600	23.3	3,470	20.9	1,830	83,500
	Connecticut River Tributaries								
01135500	Passumpsic River at Passumpsic, Vt.	436	1930-90	744	23.2	142	19.1	93.7	4,710
01144000	White River at West Hartford, Vt.	690	1916-90	1,180	23.2	250	21.2	101	8,420
01156000	West River at Newfane, Vt.	308	1920-90	632	27.9	142	22.5	25.0	5,350
01161000	Ashuelot River at Hinsdale, N.H.	420	1908-11, 1915-90	678	21.9	184	27.1	46.9	3,830
01166500	Millers River at Erving, Mass.	372	1916-90	636	23.2	170	26.7	45.7	3,340
01170000	Deerfield River near West Deerfield, Mass.	557	1941-90	1,300	31.7	290	22.3	73.1	6,820
01177000	Chicopee River at Indian Orchard, Mass.	689	1929-90	900	17.7	304	33.8	85.7	4,230
01183500	Westfield River near Westfield, Mass.	497	1915-90	925	25.3	251	27.1	86.2	5,600
01189995	Farmington River at Tariffville, Conn.	577	1972-91	1,200	28.2	337	28.1	189	5,730
01193500	Salmon River near East Hampton, Conn.	100	1928-91	185	25.2	54.8	30	6.5	1,050
	Housatonic River Basin								
01197500	Housatonic River near Great Barrington, Mass.	282	1914-90	526	25.3	134	25.5	54.8	2,880
01200500	Housatonic River at Gaylordsville, Conn.	996	1941-90	1,680	22.9	467	27.8	180	8,380
01205500	Housatonic River at Stevenson, Conn.	1,544	1929-90	2,610	23.0	729	27.9	59.2	13,600
01208500	Naugatuck River at Beacon Falls, Conn.	260	1919-90	514	26.9	154	30.0	62.1	3,140
	Thames River Basin								
01122500	Shetucket River near Willimantic, Conn.	404	1929-90	710	23.9	203	28.6	45.6	4,030
01124000	Quinebaug River at Quinebaug, Conn.	155	1932-90	273	23.9	79.0	28.9	13.9	1,400
01127000	Quinebaug River at Jewett City, Conn.	713	1919-90	1,290	24.6	349	27.1	50.0	6,240
	Coastal River Basins								
01196500	Quinnipiac River at Wallingford, Conn.	115	1931-90	214	25.3	63.5	29.7	33.9	1,210

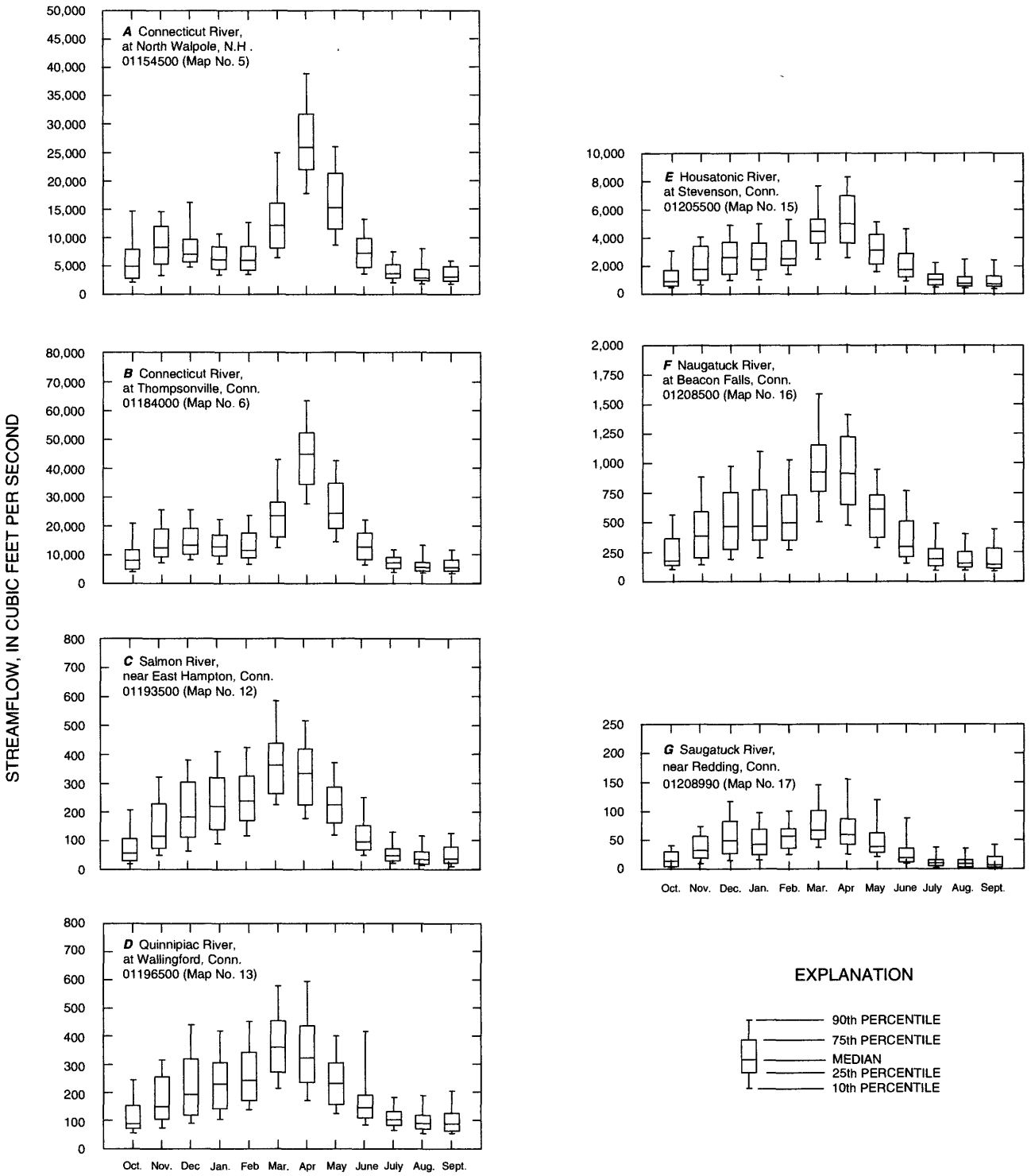


Figure 7. Variations in monthly discharge at selected sites in the Connecticut, Housatonic, and Thames River Basins study unit. *A.* Connecticut River at North Walpole, N.H. (period of record, 1943-92). *B.* Connecticut River at Thompsonville, Conn. (period of record, 1929-92). *C.* Salmon River near East Hampton, Conn. (period of record 1928-91). *D.* Quinnipiac River at Wallingford, Conn. (period of record, 1931-92). *E.* Housatonic River at Stevenson, Conn. (period of record, 1929-92); *F.* Naugatuck River at Beacon Falls, Conn. (period of record, 1919-92). *G.* Saugatuck River near Redding, Conn. (period of record, 1965-92)

Fertilizer Use

Alexander and Smith (1990) provided fertilizer-use estimates for the study unit. In their study, nitrogen and phosphorus fertilizers included those commercially available for farm and nonfarm applications; manure applications of nitrogen were not included, but may have been significant. Fertilizer application rates (by State) were obtained from the U.S. Department of Agriculture (USDA). These data were disaggregated to the county level by using data from the USDA's Census of Agriculture for the number of fertilized acres in each county which in turn, were multiplied by the State's fertilizer-use rate in kilograms per acre to obtain estimated fertilizer use in each county. Differences in application rates between counties were not taken into account and the technique may have underestimated usage in counties in which most usage is by nonfarm applicators.

Total nitrogen fertilizer applications in counties at least partly lying in the study unit increased from 17 million kg in 1970 to about 20 million kg in 1985. Whereas annual nitrogen use increased from 1970 to 1985, phosphorus use decreased from 6.8 to 5.8 million kg/yr during this period (fig. 8; see plate 1 for county locations). The increase in nitrogen use was geographically widespread. The biggest exception was Hartford County, Connecticut, which experienced a decrease in agricultural land and in nitrogen fertilizer use during that period. The decrease in phosphorus use can be seen in all counties, except for several counties in New Hampshire, where phosphorus use increased. No immediate explanation for this increase was available.

National fertilizer use by State for 1985 ranges from a minimum of 20 kg/acre for nitrogen fertilizers, to a maximum of 90 kg/acre (Alexander and Smith, 1990). The median state application rate for the Nation for nitrogen fertilizers was 40 kg/acre. The minimum State application rate for phosphorus fertilizers was 4 kg/acre, the maximum was 24 kg/acre, and the median was 9 kg/acre. The estimates of nitrogen fertilizer use for each State in the study unit range from 20 to 50 kg/acre. Estimated phosphorus usage for each State in the study unit ranges from 7 to 24 kg/acre (Alexander and Smith, 1990).

Pesticide Use

Pesticides, a general term for a wide range of substances that control undesirable organisms, include insecticides, herbicides, fungicides, rodenticides, and disinfectants. Since the 1940's, thousands of chemicals have been synthesized and introduced into the environment for the purpose of controlling nuisance organisms that interfere with the production of food and fiber, or that threaten human health. As a consequence of their widespread use, pesticides are detected widely in surface- and ground-water resources and in fish and wildlife populations.

Information on pesticide use is needed to understand the occurrence and distribution of these possibly hazardous chemicals in the environment. Knowledge of when and where pesticides are applied helps scientists and regulators to elucidate where environmental contamination may occur and how to prevent or minimize it. Currently, our knowledge of pesticide-use patterns is limited by the availability of data on pesticide application. Some pesticide-use data are available for the study unit from various sources, however, deficiencies in the scope, period, and methods of reporting pesticide use prevent an accurate and comprehensive assessment at this time.

Sources of information on pesticide use in the study unit include a national data base from Resources For the Future (Gianessi and Puffer, 1988, 1990) and several State inventories. Resources for the Future (RFF) estimated the national use of 25 compounds on 64 crops at the county level for 1982-85. These estimates were calculated using crop-acreage information for each county (from the 1982 Census of Agriculture) and typical application rates for each crop in a State. Crop-application rates, expressed as the number of pounds of active ingredient typically applied per year, were based on expert opinions obtained from State agencies and other sources; no independent surveys were conducted to verify these opinions. Estimated pesticide usage equals the number of acres treated with a compound multiplied by the crop-application rate (Gianessi and Puffer, 1988). As the data represent use on selected crops only, noncropland applications are unaccounted for, as are uses on crops other than those selected. Crops included in the estimates for the study unit include alfalfa, apples, silage corn, pasture, potatoes, squash, sweet corn, barley, and other hay.

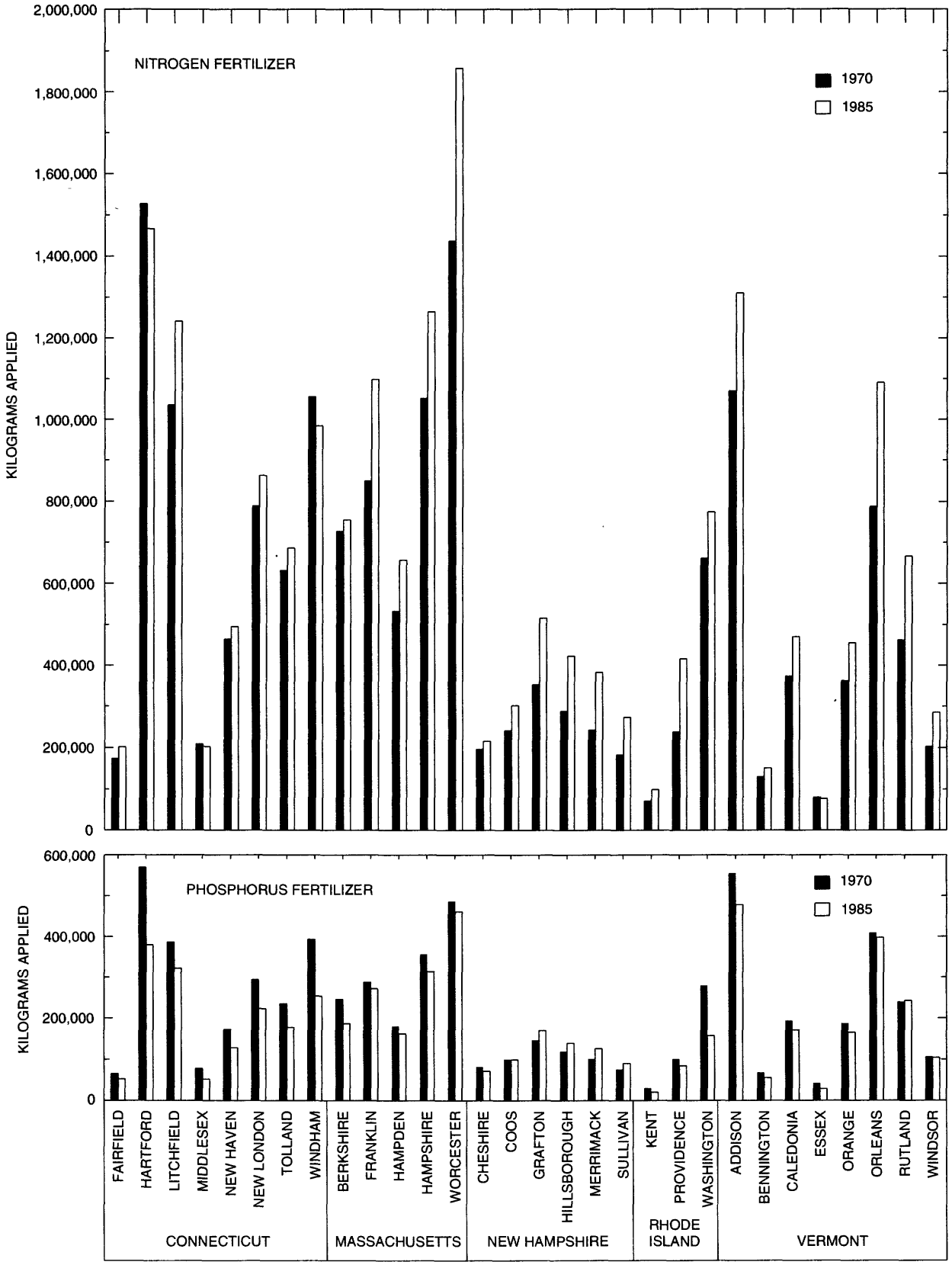


Figure 8. Nitrogen and phosphorus fertilizer use by county in Connecticut, Massachusetts, New Hampshire, Rhode Island, and Vermont, 1970 and 1985. (Data from Alexander and Smith, 1990.)

Estimates of average annual pesticide use from Gianessi and Puffer (1988) during 1982-85 for the nine most heavily used compounds are listed in table 2. Pesticide use is estimated for entire counties, not just the parts in the study unit. A few counties having only a small fraction of their area in the study unit (Orleans and Washington, Vermont; Hillsborough, New Hampshire; Duchess, Putnam, and Westchester, New York; Providence and Kent, Rhode Island) were omitted from the table. Vermont and Connecticut account for most of the pesticides used (72 percent). This may reflect the somewhat greater extent of agricultural land in these States compared with Massachusetts and New Hampshire (see fig. 3).

Pesticide compounds applied most heavily to agricultural land in the study unit are atrazine and alachlor (table 2). These are used on corn grown as forage for dairy cows. The three next most heavily used pesticides are metolachlor, metiram, and methyl parathion. Methyl parathion, an organophosphate insecticide used heavily on apples in Massachusetts and Connecticut (Gianessi and Puffer, 1988), is now a fully restricted product in Vermont. Since the compilation of the RFF data base, when metiram was used on apples, the use of metiram has been limited to potatoes and roses, which are not major crops in the study unit. Some of these pesticides, notably 2,4-D, are used more on noncrop areas in the study unit, such as on golfcourses and lawns, than on crop areas. Therefore, their actual total use for the mid-1980's probably is greater than estimated by Gianessi and Puffer (1988).

Although the RFF data represent agricultural pesticide use for the early to mid-1980's, somewhat more recent data are available from three States that constitute a large part of the study unit (Connecticut, New Hampshire, and Vermont). Commercial pesticide applicators are required to report all pesticide use in Connecticut, New Hampshire, and Vermont. In addition, farmers who apply pesticides that have restricted uses, also must report the quantities of these pesticides that they use. However, each State maintains its own list of restricted use pesticides, which commonly includes pesticides restricted nationally by the U.S. Environmental Protection Agency (USEPA). Homeowner use is not reported in any State. Massachusetts is currently developing a computer data base of reported pesticide use (Gail Kaprelian, Massachusetts Department of Food and Agriculture, Division of Pesticide Control, oral commun., May 5, 1993); however, summaries for Massachusetts are not currently available. Vermont data represent reported use in 1991 and the most current available New Hampshire records date to 1989. In 1987, the State of Connecticut compiled a data base of commercially licensed pesticide applicators (B.R. Robinson, Connecticut Department of Environmental Protection, written commun., July 26, 1989). Private farm applications of pesticides are not included in the compilation for Connecticut, although they are included with commercial applications in New Hampshire and Vermont for restricted compounds.

Because some compounds (or some formulations of certain compounds) are unrestricted, data collected by the States are incomplete. In Vermont and Connecticut, farmers are not required to report their use of unrestricted pesticides. These compounds, which are unregulated because of their formulations, methods of application, low soil leaching potential, relatively fast degradation in the environment, low toxicity, or other reasons, are reported for commercial applicators only.

The level of reporting completeness for various pesticides may be broken down into several categories: (1) those with primarily agricultural uses that are restricted—such as carbofuran, chlordane, and cyanazine—are well accounted for; (2) those used agriculturally with restrictions in use for formulations of more than 2 to 5 percent active ingredient—such as alachlor, atrazine, methyl bromide, and metolachlor—are relatively well accounted for; and (3) those used primarily on nonagricultural areas (turf, lawns, right-of-ways) by commercial applicators and homeowners—(such as 2,4-D, MCPP, bensulide, siduron, vinclozolin, or tryclopyr)—are relatively well accounted for, assuming that total homeowner use is not significantly large in comparison to commercial use. The real gaps in our knowledge of pesticide use fall within categories (2) and (3) as cited above: pesticides with wide agricultural uses but nonstringent reporting requirements (only formulations of more than 20 to 30 percent, or, in one case, more than 65 percent active ingredient, require reporting), such as chlorpyrifos, DCPA, diazinon, dicamba, isofenphos, or pendimethalin; and those that are entirely unrestricted but have heavy agricultural use, such as benefin, carbaryl, chlorothalanil, glyphosate, iprodione, and trifluralin. For benefin and trifluralin, the only information available on their use is for commercial applicators, which includes only a small fraction of the total agricultural use of these compounds.

Table 2. Estimated average annual pesticide use for counties in Connecticut, Massachusetts, New Hampshire, and Vermont in the Connecticut, Housatonic, and Thames Rivers study unit, 1982-85

[Data from Gianessi and Puffer (1988). Pesticide use estimated from the number of acres treated multiplied by the number of pounds of active ingredient applied per acre for the nine most heavily used pesticides in the study unit. Estimates are based on expert opinions and no independent surveys were conducted to verify these opinions. Usage data are for the whole county, not just for that part in the study unit]

County	Estimated pesticide use (pounds of active ingredient per year)									
	2,4-D	Alachlor	Atrazine	Carbaryl	Carbofuran	Cyanazine	Methyl parathion	Metolachlor	Metiram	Total
Connecticut										
Fairfield	347	634	1,584	729	163	158	1,907	478	2,376	8,376
Hartford	437	5,297	13,242	2,252	1,203	1,324	5,506	4,713	7,270	41,244
Litchfield	854	9,229	23,072	774	2,181	2,307	1,638	6,930	1,987	48,972
Middlesex	207	1,126	2,816	687	269	282	1,778	844	2,214	10,223
New Haven	249	2,839	7,098	1,538	699	710	3,957	2,136	4,920	24,146
New London	605	8,348	20,870	1,026	1,919	2,087	2,383	6,269	2,935	46,442
Tolland	196	8,708	21,770	466	1,898	2,177	614	7,366	1,200	44,395
Windham	297	11,800	29,500	1,208	2,620	2,950	2,785	8,850	3,426	63,436
Total	3,192	47,981	119,952	8,680	10,952	11,995	20,568	37,586	26,328	287,234
Massachusetts										
Berkshire	679	3,382	9,202	673	1,188	2,386	1,565	1,904	1,758	22,737
Franklin	674	3,386	9,246	2,233	1,227	2,402	4,943	1,966	6,704	32,781
Hampden	436	2,394	5,697	2,016	710	1,350	4,619	1,294	5,358	23,874
Hampshire	635	4,278	11,250	2,557	1,516	2,856	3,776	2,564	6,579	36,011
Worcester	1,463	5,650	14,270	6,898	1,923	3,527	17,063	3,113	20,647	74,554
Total	3,887	19,090	49,665	14,377	6,654	12,521	31,966	10,841	41,046	189,957
New Hampshire										
Cheshire	347	851	1,642	37	617	304	12	3,040	184	7,034
Coos	971	595	1,148	19	433	213	0	2,126	24	5,529
Grafton	1,063	1,250	2,410	88	907	446	39	4,464	586	11,253
Merrimack	680	1,100	2,120	124	799	393	62	3,928	934	10,140
Sullivan	641	716	1,382	24	519	256	6	2,558	97	6,199
Total	3,702	4,512	8,702	292	3,275	1,612	119	16,116	1,825	40,155
Vermont										
Addison	3,222	48,379	60,474	913	5,658	5,498	386	12,095	5,713	142,338
Bennington	604	4,675	5,843	38	550	531	0	1,169	0	13,410
Caledonia	1,816	5,866	7,333	59	692	667	6	1,467	89	17,995
Essex	554	0	0	6	3	0	0	0	0	563
Orange	3,706	9,951	12,439	95	1,159	1,131	25	2,488	375	31,369
Rutland	3,773	17,350	21,688	248	2,032	1,972	81	4,338	1,220	52,702
Windham	1,427	6,938	8,672	254	798	788	150	1,734	2,252	23,013
Windsor	2,148	7,000	8,749	133	817	795	57	1,750	850	22,299
Total	17,250	100,159	125,198	1,746	11,709	11,382	705	25,041	10,499	303,689
Total for study unit	28,031	171,742	303,517	25,095	32,500	37,510	53,358	89,584	79,698	821,035

State compilations of available pesticide-use data (in Connecticut, New Hampshire, and Vermont only) have been tabulated for 55 active-ingredient pesticides having a reported annual use of more than 1,000 lb (table 3). The entire State of Connecticut is included in the Connecticut figures (commercial applicators only); in Vermont, only Essex, Caledonia, Orange, Washington, Windsor, and Windham Counties are included (see pl. 1 for county locations); and pesticide use in 90 New Hampshire towns falling at least partly in the study-unit boundaries is provided. The compounds included represent 86 percent of the reported pesticide use in New Hampshire, 89 percent of reported use in Vermont, and 88 percent of the reported commercial pesticide use in Connecticut. Several factors limit the use of these data—noncommercial use of nonregulated compounds is missing, as is all use for Massachusetts, and the data do not cover the same year in each State, which introduces errors due to changes in pesticide use with time when making comparisons among the States. Despite these limitations, the data do complement the RFF estimates, as they include primarily commercial use, however, the RFF data included solely agricultural use.

While taking into account the limitations of the pesticide-use data, the State and RFF estimates indicate that the most heavily used compounds in the study unit include: atrazine (150,000 lb/yr), chlorpyrifos (147,000 lb/yr), carbaryl (68,000 lb/yr), metolachlor (65,000 lb/yr), alachlor (57,000 lb/yr), and pendimethalin (42,000 lb/yr). Glyphosate, although not in this group, may deserve special mention because officials have cited its heavy use (Jeff Comstock, Vermont Department of Agriculture, oral commun., 1993; Gail Kaprelian, Massachusetts Department of Food and Agriculture, oral commun., 1993).

Alachlor, atrazine, and metolachlor are used primarily on silage corn and sweet corn. Other agriculturally used insecticides such as carbofuran, chlorpyrifos, iprodione, and methoxychlor are not commonly used in urban areas. Pesticides generally used for turf or lawn care in urban and suburban areas in the study unit include the herbicides 2,4-D, bensulide, DCPA, and MCPP, and the insecticides diazinon, carbaryl, isophenphos, and trichlorofon. Some pesticides, such as glyphosate, pendimethalin, and chlorpyrifos are used ubiquitously on urban and agricultural lands. Since 1987, there has been a strong shift away from using 2,4-D and few agricultural or commercial users apply it currently (F. Himmelstein, University of Connecticut Cooperative Extension Service, oral commun., April 1993), although it still receives use in homeowner products. The use of 2,4-D decreased in New Hampshire from 1988 to 1989 by 25 percent in towns in the study unit (data from University of New Hampshire Complex Systems Research Center) and Vermont also reports a decrease in use (Jeff Comstock, Vermont Department of Agriculture, oral commun., 1993). In Connecticut, alachlor's use is decreasing as are other pesticides' including chlordane—discontinued in 1988—and carbofuran (B.R. Robinson, Connecticut Department of Environmental Protection, oral commun., May 1993). Dicamba use in Vermont is decreasing (Jeff Comstock, Vermont Department of Agriculture, oral commun., 1993). Although usage of these pesticides decreased, usage of atrazine, metolachlor, and pendimethalin increased in Vermont. Sulfonylurea pesticide use has begun in Connecticut, but not in Vermont. Chlorpyrifos was the most widely used pesticide reported in Connecticut in 1987 and its use has continued to increase. Pendimethalin use also increased in Connecticut, and, by contrast, the use of triazine herbicides, such as atrazine and cyanazine, decreased because of increased triazine resistance in weeds in cornfields (B.R. Robinson, Connecticut Department of Environmental Protection, oral commun., May 1993).

Because of the problems with under-reported chemicals in most States, a complete breakdown of usage in various sectors is not possible. However, in conjunction with the compilation of pesticide use by commercial applicators conducted in Connecticut in 1987 (B.R. Robinson, Connecticut Department of Environmental Protection, written commun., July 1989), estimates of the amounts used in agricultural and nonagricultural sectors (table 4) can be made (Keeney, 1991). Total agricultural use (private-agricultural and commercial-agricultural applications) accounts for 46 percent of the total estimated pesticide use in Connecticut. Total nonagricultural use (outdoors—commercial and homeowner) in urban and suburban areas accounts for 46 percent of the total estimated pesticide use. These land uses may contribute to total pesticide use in approximately equal proportions to agricultural use in that State.

Table 3. Reported pesticide use for areas of Connecticut (1987), New Hampshire (1989), and Vermont (1991) in the Connecticut, Housatonic, and Thames Rivers study unit

[Uses: Data source—Sine, 1991. **Sources for reported use, Conn., 1987:** Data Source: B.R. Robinson, Connecticut Department of Environmental Protection, Pesticide Control Section, written commun., July 1989. Restricted use status: Connecticut Department of Environmental Protection, 1992. Restricted use status has little bearing on reporting in this compilation, as only commercial applicators report. No private use of restricted pesticides is included for Connecticut. **N.H., 1989:** Data Source: Fay Rubin, University of New Hampshire, Complex Systems Research Center, written commun., April 1993. Restricted use status: New Hampshire Department of Agriculture, 1992. **Vt., 1991:** Data Source: Jeffrey Comstock, Vermont Department of Agriculture, Plant Industry Section, written commun., April 1993. Restricted Use Status: Vermont Department of Agriculture, 1991. Reported pesticide use includes only commercial applicators for Connecticut; all restricted pesticide use reported for New Hampshire and Vermont. For entries with no footnote, no restrictions on the use of this chemical (only commercial use reported); nr = no reported use; <, actual value is less than value shown; 2,4-D, (2,4-dichlorophenoxy) acetic acid; 2,4-DP, 2-(2,4-dichlorophenoxy) propionic acid; DCPA, dimethyltetrachloroterephthalate; MCPP, 2-(2-methyl-40-chlorophenoxy) propionic acid; MSMA, monosodium methanearsonate; PCNB, pentachloronitrobenzene]

Pesticide	Uses	Reported use by State and year (pounds of active ingredient)		
		Conn. 1987	N.H. 1989	Vt. 1991
2,4-D	Systemic selective herbicide, turf, pasture, corn	[1]31,017	170	[1]582
2,4-DP (Dichloroprop)	Systemic herbicide, rights-of-way, aquatic weeds	2,960	nr	[1]40
Acephate	Contact and systemic insecticide, corn, alfalfa	2,436	<1	2
Alachlor	Pre-emergence herbicide for corn, potatoes	[2]1,425	[2]1,421	[2]4,121
Ammonium sulfamate	Contact, translocated herbicide, discontinued 1988	17,181	nr	nr
Anilazine	Foliar fungicide, ornamentals, turf, vegetables, others	4,995	nr	237
Atrazine	Selective herbicide (for corn)	[2]13,872	[2]7,278	[1]9,732
Bendiocarb	Residual insecticide (indoor use)	[2]2,422	[1]nr	[1]642
Benefin	Preemergent herbicide for alfalfa, clover	12,223	1.5	nr
Benomyl	Systemic fungicide for fruits, vegetables, turf, field crops	2,844	82	[1]9.4
Bensulide	Herbicide for lawns, some vegetables	10,208	nr	242
Captan	Protectant-eradicant fungicide for fruit trees	1,588	654	452
Carbaryl	Broad spectrum insecticide for crops, lawn	56,107	175	677
Carbofuran	Broad spectrum insecticide, nematicide, corn, potatoes	[2]270	[2]748	[2]nr
Chlordane	Contact insecticide (highly toxic to fish)	[1]14,391	[2]nr	[2]nr
Chloroneb	Fungicide, seed treatment	3,470	nr	87
Chlorothalanil	Fungicide for row crops and vegetables	28,270	136	1,308
Chlorpyrifos	Insecticide for alfalfa, corn, other field crops, termiticide	[1]146,801	56	[1]327
Cyanazine	Selective herbicide for corn	[2]1,730	[2]1,170	[2]3,937
DCPA (Dacthal)	Herbicide for turf, vegetables, ornamentals	24,148	nr	[1]nr
Diazinon	Multi-purpose insecticide, nematicide	[1]17,885	90	[1]200
Dicamba	Herbicide	[1]4,055	62	[1]248
Dimethoate	Systemic insecticide-acaricide, many crops	1,399	60	nr
Diuron	Herbicide, for fruit orchards and vineyards	5,423	<1	[1]1,656
Fosamine ammonium	Brush control agent, growth regulant	1,308	nr	108
Fosetyl-Al	Systemic fungicide, fruit and vegetable crops	1,426	nr	35.2
Glyphosate	Nonselective herbicide, urban and agricultural uses	13,075	525	2,686
Iprodione	Contact fungicide, potatoes, fruits, turf	8,913	nr	2,297
Isofenphos	Insecticide for turf, lawns	[1]17,275	[2]5.0	[1]576
Malathion	Insecticide for fruits, vegetables	9,739	15.4	6.4
Mancozeb	Fungicide, fruit, vegetable, field crops	9,385	1,577	nr
Maneb	Fungicide for potatoes, vegetables, tobacco	[1]1,710	627	nr
MCPP (potassium salt)	Herbicide for grasses, lawns	17,593	1.2	239
Metalaxyl	Fungicide seed dressing	2,393	8	10.3
Methoxychlor	Insecticide, fruit & shade trees, vegetable gardens	3,763	32	nr
Methyl bromide	Fumigant for insect and rodent control	[1]1,978	[2]nr	[2]nr
Metiram	Fungicide, for fruits, vegetables, ornamentals, field crops	nr	1,017	38.4
Metolachlor	Selective herbicide for corn, potatoes, others	9,100	7,082	[1]11,303
MSMA	Herbicide, crabgrass and broadleafs in turf	1,469	nr	nr

Table 3. Reported pesticide use for areas of Connecticut (1987), New Hampshire (1989), and Vermont (1991) in the Connecticut, Housatonic, and Thames Rivers study unit—*Continued*

Pesticide	Uses	Reported use by State and year (pounds of active ingredient)		
		Conn. 1987	N.H. 1989	Vt. 1991
Oil	Sprayed to control insects, insect eggs, plants	99,984	951	106.3
Oryzalin	Selective preemergent herbicide, fruit trees, turf	1,208	19	40.5
PCNB	Soil fumigant, soil dressing against smuts	943	nr	2,406
Pendimethalin	Selective herbicide for corn, potatoes, tobacco	[1]39,011	1,314	[1]1,373
Permethrin	Insecticide for corn, vegetables; wood preservative	[1]1,677	[2]55	[1]nr
Siduron	Herbicide for grasses	3,482	nr	76.5
Simazine	Selective herbicide for corn, alfalfa, fruit trees, lawns	[1]3,409	1,045	[1]370
Sulfometuron methyl	Herbicide for noncropland, reforestation areas	1,393	nr	49.3
Thiophanate	Systemic fungicide, discontinued 1989	3,156	nr	137
Thiram	Fungicide, animal repellant for fruit trees, shrubs	4,986	9.1	197
Triadimefon	Systemic fungicide for fruit trees, vegetables, pine	3,254	nr	134
Trichlorofon	Insecticide for corn, fruit trees, turf, forests, others	10,902	nr	809
Triclopyr	Systemic herbicide, rights-of-way, pastures, industrial sites	[1]3,480	nr	1,581
Trifluralin	Selective preemergent herbicide for alfalfa, vegetables	5,130	13	1,250
Vinclozolin	Fungicide for turfgrass, vegetables	1,439	43	33.4
Vorlex	Soil fumigant for potatoes, tobacco, vegetables	[1]36,485	1,607	[2]nr

[1] Some formulations of this chemical are restricted (all commercial use reported, some private use reported).

[2] Use of this chemical is restricted (all use reported);

Table 4. Total estimated annual pesticide use in Connecticut, 1987

[Data from Keeney, 1991]

Type of pesticide use	Total estimated annual use (pounds of active ingredients)	Percentage of total use
Agricultural		
Applied by farmer or landowner	530,000	37
Applied by commercial applicator	130,000	9.0
Total	660,000	46
Non-agricultural use		
Outdoors		
Applied by commercial applicators	630,000	43
Applied by homeowners	42,000	3.0
Total	670,000	46
Indoors		
Applied by commercial applicators	64,000	4.0
Applied by homeowners	58,000	4.0
Total	120,000	8.0
Overall total	1,450,000	100

COLLECTION, COMPILATION, SCREENING, AND ANALYSIS OF AVAILABLE WATER-QUALITY DATA

This report incorporates information from seven data-collection programs. Agencies providing data include the USGS, USEPA, Connecticut Department of Health Services, Colorado State University, and University of Massachusetts. The data encompass six major types of sampling sites, including surface water (streams), estuaries, lakes, wells, springs, and atmospheric-deposition collectors. Available data for nutrients, suspended sediment, and pesticides were evaluated and compiled for each type of site and in each program (table 5).

The highest priority nutrients from the National perspective of the NAWQA program, are total nitrogen, dissolved ammonia as nitrogen, and dissolved nitrate as nitrogen. Data for these constituents were selected where possible. Where data for these constituents are limited or unavailable, data for the most closely related constituents were selected. Similarly, phosphorus data analyses focused on total phosphorus, dissolved phosphorus, and dissolved orthophosphate. Among the data bases evaluated in this report, data for suspended sediment were available only from the USGS's National Water Information System (NWIS), an automated information system for processing, storing, and retrieving water data.

Quality-assurance practices affecting the data included in this report have evolved substantially during the 1972-92 study period. In general, increasingly more stringent and well-documented quality-assurance practices for the collection and analysis of water-quality data have been implemented during the 1980's and 1990's. In each agency or program, quality-assurance practices ideally govern each step leading to the creation of water-quality data; selection and maintenance of equipment; field measurements; collection, treatment, and shipping of samples; laboratory analysis; and computer data entry and verification. Such practices vary among different agencies and programs, and, in some cases, are unreported.

National Water Information System

NWIS constitutes the primary source of information analyzed in this report. Federal and Federal-State cooperative programs support the collection of water-quality data stored and maintained in NWIS.

The USGS collects surface-water-quality samples from bridges, piers, riverbanks, and boats, and instream by wading. Historically, methods used to collect samples in the study unit have included point, grab, single vertical, multiple vertical, and equal-width-increment (EWI) sampling (Edwards and Glysson, 1988). The methods most frequently used in recent years have been single- and multiple-vertical sampling. Sampling methods have changed at some stations during the study period, and changes have not been simultaneous at all stations in the study unit. Data analyzed for this report typically include more than one sampling method.

The USGS collects ground-water-quality samples from public-supply, domestic, industrial, and observation wells using various methods (Rainwater and Thatcher, 1960; Wood, 1976; and Claassen, 1982). In recent years, sample-collection protocols and materials compatible with sampling for trace level, synthetic organic compounds have been implemented (Grady and Weaver, 1988; Mullaney and others, 1991). Kulp and Hunter (1987) describe the Survey's precipitation-collection methods.

Data stored in the NWIS include values less than and greater than analytical detection limits. Data less than a detection limit are sometimes referred to as censored data or "less-thans." The numerical values of data less than a detection limit are unknown, but still contain information that can be used in some analyses. All analyses in this report incorporate censored data, where present, unless otherwise indicated.

Nutrients and Suspended Sediment

NWIS lists numerous sampling sites in the study unit, and records exist for many constituents over a long period of record. For this reason, sites and constituents were screened to obtain the most useful data.

Table 5. Summary of water-quality data-collection sites, water-quality records, and water years of record by source of data, type of site, and constituents analyzed for the Connecticut, Housatonic, and Thames Rivers study unit, water years 1915-93

[Data from the U.S. Geological Survey's National Water Information System (NWIS) and from the U.S. Environmental Protection Agency's STOrage and RETrieval system (STORET). Each water-quality record refers to one sampling, which may include data for several constituents and properties. --, indicates no data]

Water-quality constituent(s)	Source of data					
	NWIS			STORET		
	Number of sites	Number of water-quality records	Period of record (water year)	Number of sites	Number of water-quality records	Period of record (water year)
Surface water						
All constituents	1,194	28,000	1952-93	1,791	238,127	1939-91
Nutrients	839	10,958	1952-93	962	20,067	1959-91
Suspended sediment	121	3,950	1965-91	0	0	--
Pesticides	90	333	1969-92	14	34	1962-87
Estuaries						
All constituents	69	2,190	1952-93	40	280	1969-89
Nutrients	48	2,023	1952-93	38	247	1970-89
Suspended sediment	8	21	1971-76	0	0	--
Pesticides	12	44	1972-82	0	0	--
Lakes						
All constituents	269	2,025	1921-91	590	10,903	1966-91
Nutrients	193	999	1921-91	251	3,600	1972-87
Suspended sediment	1	1	1983	0	0	--
Pesticides	22	34	1975-91	2	2	1973-77
Wells						
All constituents	2,357	4,075	1915-93	64	581	1939-90
Nutrients	1,872	2,508	1917-93	25	341	1971-84
Pesticides	225	512	1978-89	0	0	--
Springs						
All constituents	42	52	1951-93	2	3	1988-90
Nutrients	42	47	1951-93	0	0	--
Pesticides	8	8	1988	0	0	--
Meteorological						
All constituents	43	1,366	1963-91	0	0	--
Nutrients	32	780	1964-91	0	0	--
Pesticides	1	2	1987	0	0	--

Constituent Screening and Selection

Requirements of the NAWQA program at the National level and the availability, quality, and abundance of data at long-term surface-water sampling stations in the study unit guided selection of water-quality constituents for nutrient and suspended sediment. Historically, NWIS stored nutrient data under many different parameter codes, depending on changes in laboratory methods and other factors. Sixteen parameter codes (table 6) served to screen surface-water-quality stations with nutrient and sediment data, as described below. These parameter codes for nitrogen, phosphorus, and suspended sediment represented the bulk of the recent and long-term nutrient and sediment data collection in the study unit, and included constituents of priority at the National level. Surface-water nutrient and sediment data retrieved from the NWIS include laboratory analytical values and values calculated from other analyzed values.

Table 6. Nutrient and suspended sediment parameter codes used in selecting surface-water-quality stations for data analysis

[Data from the U.S. Geological Survey's National Water Information System (NWIS) and from the U.S. Environmental Protection Agency's STOrage and RETrieval system (STORET). Parameter code is a 5-digit number used to uniquely identify a specific constituent. Codes used in NWIS are the same as those used in STORET. U.S. Environmental Protection Agency assigns and approves all requests for new codes. mg/L, milligram per liter]

Parameter code	Constituent (mg/L)
00620	Nitrogen, nitrate, total (as N)[1]
00618	Nitrogen, nitrate, dissolved (as N)[1]
00615	Nitrogen, nitrite, total (as N)
00613	Nitrogen, nitrite, dissolved (as N)
00630	Nitrogen, nitrite plus nitrate, total (as N)
00631	Nitrogen, nitrite plus nitrate, dissolved (as N)
00610	Nitrogen, ammonia, total (as N)
00608	Nitrogen, ammonia, dissolved (as N)
00605	Nitrogen, organic, total (as N)[1]
00625	Nitrogen, ammonia plus organic, total (as N)
00623	Nitrogen, ammonia plus organic, dissolved (as N)
00600	Nitrogen, total (as N)[1]
00665	Phosphorus, total (as P)
00666	Phosphorus, dissolved (as P)
00671	Phosphorus, orthophosphate, dissolved (as P)
80154	Sediment, suspended

[1]Calculated.

All available nutrient data in NWIS for groundwater sites (wells and springs) were retrieved. The resulting nutrient data cover an extensive period of record (water years 1917-93), and the data include more than 2,500 analyses containing one or more of 20 nutrient water-quality constituents (table 5).

Sample counts and summary statistics for the constituents retrieved were reviewed in the process of selecting constituents for data analysis in this study. Because of the abundance of data in the study unit, complete data analyses for all these constituents were not included in this report.

Data in the study unit are considerably more extensive for total nitrite plus nitrate than for dissolved nitrate. Furthermore, data for selected surface-water-quality stations indicate that at most locations, the concentration of dissolved or total nitrite is usually negligible. Consequently, data on total nitrite plus nitrate have been presented in this report, rather than dissolved nitrate. Statistical analyses on a national data set show no significant difference between paired values of total and dissolved nitrite plus nitrate samples because of a laboratory analytical problem (D.A. Rickert, U.S. Geological Survey, written commun., 1992). For this reason, and because of negligible nitrite concentrations, reported concentrations of total nitrite plus nitrate are considered essentially equivalent to concentrations of dissolved nitrate, unless otherwise indicated.

There is an abundance of data on total nitrogen for the study unit. Total nitrogen, calculated as part of the data-retrieval process in NWIS is the sum of analyzed concentrations for total nitrite plus nitrate and total ammonia plus organic nitrogen (table 6). If either of the two constituent concentrations were less than the detection limit, total nitrogen was not calculated. This represents a potential loss of data at the low end of the concentration range for total nitrogen.

In analyzing total nitrogen data, we compared the use of censored data with the elimination of censored data. Three methods were used to evaluate censored data for the constituents used to calculate total nitrogen: (1) censored data eliminated; (2) censored data set high (the detection limit); and (3) censored data set low (zero). For method 1, the method used by NWIS, if either constituent concentration was censored, total nitrogen was not calculated and the sample was eliminated. This procedure can result in either a positive or negative bias in the estimated distribution of total nitrogen concentrations. For method 2, if either constituent concentration was censored, the detection limit for that constituent was used in the calculation of total nitrogen. This method also could result in a positive bias. For method 3, zero was used in place of censored data in the calculation. This method could result in a negative bias.

Boxplots (Chambers and others, 1983) of total nitrogen were constructed using the three methods for evaluating censored data. In this report, data percentiles define the boxplots. Any change in the boxplots resulting from methods of evaluating censored data reflected changes in the percentiles. Boxplots for seven of the eight stations examined generally were insensitive to the method used to evaluate censored data (Michael Turtora, U.S. Geological Survey, written commun., 1993). The data sets evaluated were large enough, and the number of censored data low enough, for the percentiles represented by the box diagram to remain unchanged.

Evaluation of total nitrogen data for these stations indicates that total nitrogen concentrations retrieved from and calculated by NWIS, where the number of values is large (about 100 or more), adequately represent the range and distribution of this constituent in the study unit for the purposes of this report. Where the number of concentrations were small, the reported percentiles may reflect a positive bias. That is, the percentages for total nitrogen may be somewhat higher than the actual concentrations, particularly for relatively pristine drainage basins where there is likely a large number of censored data used to calculate total nitrogen.

Station Screening and Selection

For the analysis of surface-water quality, all stations with nutrient or suspended-sediment data were retrieved from NWIS. Lakes, impoundments, estuaries, and all other stations where tides affect stage were eliminated. A total of 452 stations had at least one observation for at least one of the nutrient or sediment parameter codes listed in table 6. Of these 452 stations, 119 stations had at least 10 observations for at least one of the nutrient or sediment parameter codes during water years 1972-92[1]. These 119 stations were selected for analysis.

Further screening of these 119 stations yielded a group of 18 major water-quality stations. Each major station: (1) had at least 50 observations for one or more of the parameter-coded water-quality constituents; (2) was associated with a streamflow-gaging station served by a continuous water-stage recorder; and (3) had continuous streamflow records for water years 1980-90. Only one of these stations was outside of Connecticut.

A subset of eight stations (table 7) was selected from the 18 major stations for use in many of the analytical procedures of this study. Water quality at the remaining 10 stations is discussed in the text, but, in general, is not presented graphically. These eight stations were selected for geographic coverage of the study area, along with consideration of the following factors that affect the quality of the data: (1) continuity of water-quality-sampling site with time; (2) consistency of streamflow data; (3) proximity of the water-quality-sampling station to the stream-gaging station, and (4) lack of problems with streamflow calculations for the water-quality-sampling site. These criteria minimize or remove potential sources of error in analyses, in particular, trend analysis and load estimation. Four stations are part of the USGS's National Stream Quality Accounting Network (NASQAN) in the study unit's three major river basins and are referred to as "integrator" stations because their relatively large drainage basins include a variety of environmental settings and land uses whose cumulative effects on water quality tend to mask the water-quality characteristics of any specific setting. The other four stations represent smaller drainages in relatively homogeneous environmental settings.

NWIS ground-water sites were included in this report if they were in the study unit as verified by plotting the location of all ground-water sites. If site location information was inconsistent (such as latitude and longitude not agreeing with site identification number, hydrologic unit code, or State and county code), some effort generally was made to confirm or rectify dubious entries. If the site data could not be verified, the site was excluded.

The NWIS ground-water-quality data base reflects the nature of most hydrogeologic investigations undertaken in the past by the offices of the USGS's, Water Resources Division in the study unit. Most of these previous investigations have primarily focused on cooperating agencies' need to determine the

Table 7. Surface-water-quality stations selected for detailed assessment

[Data from the U.S. Geological Survey's National Water Information System. Map No.: See plate 1 for locations of water-quality sampling stations]

Map No.	Station No.	Station name
2	01122610	Shetucket River at South Windham, Conn.[1]
5	01154500	Connecticut River at North Walpole, N.H.[1]
6	01184000	Connecticut River at Thompsonville, Conn.[1]
12	01193500	Salmon River near East Hampton, Conn.
13	01196500	Quinnipiac River at Wallingford, Conn.
15	01205500	Housatonic River at Stevenson, Conn.[1]
16	01208500	Naugatuck River at Beacon Falls, Conn.
17	01208990	Saugatuck River near Redding, Conn.

[1]NASQAN, National Stream Quality Accounting Network.

[1]A water year is the period of time beginning on October 1 and continuing through 30 September of the following calendar year.

availability and general quality of ground-water resources. As such, these investigations sampled a wide variety of wells and aquifers. Some recent studies have focused on ground-water-quality problems, both natural and anthropogenic. No effort was made to screen and eliminate sites installed to sample contaminated ground water. The NWIS data base used for the analysis of nutrients in ground water in the study unit includes 1,872 wells and 42 springs (table 5).

Effects of Changes in Laboratory Analysis Methods on Historic Nutrient Data

Changes in laboratory analytical methods for a constituent can affect the comparability of data between different periods of the historical record (Alexander and others, 1993). Evaluation of methods and the resulting data also can yield information that affects the validity of data for certain interpretive analyses. Information on some of these changes and evaluations, and their possible effects on NWIS data, is available from the Survey's Office of Water Quality (OWQ) and National Water Quality Laboratory (NWQL). The following discussion refers only to NWIS data.

Most samples from water-quality stations in Connecticut were analyzed in the Survey's Albany, New York, laboratory before May 1977. Connecticut samples were analyzed in the Atlanta, Georgia laboratory from May 1977 to November 1985, and Connecticut samples for all constituents, except suspended sediment, were analyzed at the NWQL in Denver, Colorado, from December 1985 to the present.

Nitrogen

The Denver laboratory experienced an ammonia contamination problem from 1980 through 1982 that affected ammonia and Kjeldahl nitrogen concentrations (D.R. Helsel, U.S. Geological Survey, written commun., 1992). The Atlanta laboratory analyzed samples from the Connecticut stations during that time period, and, thus, there would not be an analytical bias in these data.

Analyses for total ammonia, total nitrite, and total nitrite plus nitrate were discontinued at the NWQL as of January 1, 1993, because the laboratory method was determined to be unsuitable for accurately measuring total concentrations for these species, and because a statistical analysis showed no significant difference between paired values of total and dissolved samples (D.A. Rickert, U.S. Geological Survey, written commun., 1992). Data for total nitrite-plus-nitrate are included in this report with the cautionary note that the reported concentrations may not accurately represent environmental concentrations in some cases. These data are believed, however, to give a qualitative picture of general concentration variations across the study unit. The accuracy of total nitrite plus nitrate concentrations potentially affects concentrations of total nitrogen, because analyzed concentrations of total nitrite plus nitrate are used in the calculation of total nitrogen concentrations. The nature of the laboratory analytical problem for total nitrite plus nitrate is such that total nitrogen concentrations could be underestimated in some situations in the study unit.

Phosphorus

Information from the NWQL indicates that phosphorus samples analyzed at the Denver laboratory showed a positive bias from 1980 through 1982; a much smaller bias in phosphorus may be present during the same time period for the Atlanta laboratory (D.R. Helsel, U.S. Geological Survey, written commun., 1992). The Atlanta laboratory analyzed samples for the Connecticut stations during that time period, so the bias for data for these stations is presumed to be small. The positive bias for 1980-82 was detected through analysis of data from the Survey's Standard Reference Water Samples quality-assurance program (R.B. Alexander, U.S. Geological Survey, oral commun., 1994).

A negative bias in phosphorus data, related to a different aspect of the analytical process, has been reported by the OWQ. Total phosphorus data produced prior to October 1, 1991, tend to be negatively biased, that is, the reported analytical concentrations probably are lower than actual environmental concentrations under certain conditions (D.A. Rickert, U.S. Geological Survey, written commun., 1992). The negative bias probably is greater in data produced prior to May 1, 1990.

National statistical analyses have shown that the negative bias in total phosphorus data increases in samples with high concentrations of particulate phosphorus, suspended sediment, and organic carbon. Specifically, the bias was notable in samples with total phosphorus concentrations exceeding 0.2 mg/L when suspended-sediment concentrations exceeded 50 mg/L.

Bias was not statistically significant for samples with total phosphorus concentrations less than or equal to 0.2 mg/L.

Counts of samples were evaluated to determine the possible influence of the reported bias in total phosphorus concentrations for NWIS data in the study unit. Counts of samples less than and greater than the threshold concentrations of suspended sediment and total phosphorus noted above were made for several stations. Five stations in the study unit have 40 or more suspended-sediment samples collected on a quarterly basis. These five stations represent large basins, of several hundred square miles or more, and include a large part of the study unit. In each case, concentrations greater than 50 mg/L constituted less than 10 percent of the record. In a smaller drainage basin with more than 300 daily suspended-sediment samples, concentrations greater than 50 mg/L constituted less than 5 percent of the record.

Eighteen stations in the study unit have substantial total phosphorus records; all but one have more than 100 samples, and 11 have more than 200. The percentage of samples exceeding 0.2 mg/L varies substantially among the stations. Twelve of the stations have fewer than 10 percent of the total phosphorus data exceeding 0.2 mg/L. Most total phosphorus exceeded 0.2 mg/L at three stations with highly urbanized drainage basins or major point-source discharges.

Preliminary evaluation of these data indicates that for large drainage basins in the study unit, and for smaller, less developed drainage basins, phosphorus data are essentially unbiased because concentrations of total phosphorus and suspended sediment typically were less than the threshold concentrations where bias is reported. Bias may be greater in small, highly urbanized basins, in basins with major point discharges, and locally in other environmental settings with relatively high suspended-sediment concentrations.

Bias in total phosphorus data could have implications for use of the data in trend analysis and load estimation. Limitations on the data for these uses are discussed in those sections of the report.

Pesticides

This analysis included all NWIS sites that were confirmed to be in the study unit that had been sampled for any pesticide. In addition to an extensive list of chemicals that are exclusively used to control pests, the list of pesticides included in this report is broadly interpreted to include a number of volatile organic compounds (VOCs) and semi-VOCs that have some pesticide use, or are a component of a pesticide application. In total, one or more of 81 different pesticide compounds or their metabolites have been sampled 933 times at 358 sites in the study unit during water years 1969-92 (table 5). Ground-water sites (wells and springs) account for nearly two-thirds (65.1 percent) of the sites sampled; most have only been sampled once and most have been sampled within the last decade. Stream-gaging stations comprise about 25 percent of the sites sampled but many were sampled more than once, and both water and bed sediments also were sampled.

The NWIS stream-gaging stations sampled for pesticides include estuaries and lakes, as well as streams. The estuarine and lacustrine stations are included in this analysis even though they are not within the scope of the NAWQA study because they augment the limited data available for streams in the study unit. The estuary stations generally are in the lower reaches of the three principal streams in the study unit, in tidally affected tributaries, or in the harbors of coastal cities. The lake stations are principally impounded, run-of-the-river lakes along the Housatonic and Connecticut Rivers.

Quality Assurance

Guy and Norman (1970), Wood (1976), Wershaw and others (1987), Edwards and Glysson (1988), Britton and Greeson (1989), and Fishman and Friedman (1989) have described USGS procedures for field measurements and for collecting, treating, and shipping samples. The National Field Quality Assurance Program was initiated in 1979 to monitor the accuracy and precision of field measurements (Stanley and others, 1992).

The NWQL currently (1992) performs all the analytical work for the NWIS constituents discussed in this report, with the exception of suspended sediment, which is analyzed at the Lemoyne, Pennsylvania office of the USGS. Guy (1969), Wershaw and others (1987), Britton and Greeson (1989), and Fishman and Friedman (1989) have described laboratory analytical methods. Friedman and Erdmann (1982) and Pritt and Raese (1992) have described quality-assurance practices at the NWQL. Since 1981, the NWQL has

issued a series of reports (Lucey, 1989) that evaluate the precision and bias for inorganic major ion and trace metal constituent data. Dempster and Merk (1991) have described quality assurance for the NWIS data base.

Individual USGS offices are responsible for verifying correct entry of field data, checking data released from the NWQL for inconsistencies reported by the laboratory, and checking data against historical data for values that are outside normal ranges. Accepted data are entered into the water-quality data base of NWIS (Maddy and others, 1989). The routine office and laboratory procedures for checking water-quality data are considered adequate to ensure the reliability of data for the purposes of this report.

Storage and Retrieval System (STORET)

The USEPA generally does not collect its own data. Consultants, Federal and State agencies, universities, and others contract with the USEPA to perform studies and submit the water-quality data to USEPA for entry into STORET. Samples are collected and analyzed according to USEPA guidelines. Methods used to collect surface-water-quality samples have not been clearly documented in STORET, but it is assumed that most surface-water-quality samples were collected using the grab sample method.

Nutrients

This report includes nutrient data from STORET for water years 1972-90 from water-quality stations with at least 10 samples for one or more of the 16 nutrient constituents previously identified (table 6). This criterion eliminated more than 1,800 water-quality stations with limited data, leaving 114 stations. Locations of the 114 stations with adequate data were plotted on 1:24,000 or 1:25,000 scale USGS topographic maps to verify that they plotted along the identified stream reach. One station was eliminated based on this check. Finally, STORET stations were compared with NWIS stations to eliminate duplication. Four STORET stations were eliminated on this basis, leaving 109 STORET stations for analysis. All ground-water sites in STORET with nutrient water-quality data for any water years were retained for data analysis. Because of the limited nutrient water-quality data from lake, reservoir, meteorological, or estuary stations in STORET, these stations were not included in the analysis.

Pesticides

Pesticide data were available for only a few of the more than 2,000 STORET sites with water-quality data. Screening the pesticide data in STORET further limited the number of sites selected for this analysis. Eight sites with pesticide data were eliminated from the analysis because the source of water to the site was unknown. Thirty-nine wells with pesticide data were eliminated because the wells were located at two hazardous-waste sites. Twenty surface-water and two reservoir stations (table 5) were selected for further analysis. Pesticide data were not available for estuary, meteorological, or spring sites.

Quality Assurance

Little is known about the quality of STORET data. Information on sample-collection methods or quality assurance/quality control (QA/QC) practices is unavailable. Methods of analysis also are unknown, but it is assumed that they follow USEPA guidelines. Analytical methods are known to have changed for the period of record, because the data include multiple detection limits for individual constituents and, in some cases, zero values were assigned to older samples prior to the routine use of the less-than symbol (<).

National Pollutant Discharge Elimination System and Permit Compliance System

The USEPA's National Pollutant Discharge Elimination System (NPDES), developed to enforce discharge limitations under the Clean Water Act, contains information compiled about permitted dischargers in the study unit as of 1990. NPDES includes information about each discharger's permit, such as permitted discharge volume, location, and Standard Industrial Classification code. NPDES does not include water-quality data. Of 706 NPDES-reported dischargers in the study unit, 177 had permitted to discharge volumes exceeding 1 Mgal/d.

The Permit Compliance System (PCS), a data base-management system, functions within the NPDES. This system contains information on the chemical status of discharges regulated under NPDES. The data base is accessible to registered users nationwide through the USEPA. Data are stored as records consisting of concentrations and loads of nutrients sampled in waste streams to comply with individual

facility permits. PCS data are usually reported on a monthly basis; the period of record spans 1986-91. Nutrient species reported in the data base include total nitrogen, Kjeldahl nitrogen, nitrite, nitrate, ammonia, nitrite plus nitrate, total phosphorus, and phosphate. It is unknown whether the samples for each facility are filtered or unfiltered.

PCS data for gross outfalls only were analyzed. Data were checked for consistency and some data were discarded for sample periods where the reported maximum concentration, or load, was less than the average concentration or load. All sites were included that had gross outfall data for one of the constituents of interest.

The quality of data reported in NPDES and PCS depends on several factors: (1) whether the state or the USEPA has primary responsibility for maintaining the data base in each State; (2) the type of facility; (3) the rigor with which each State inspects dischargers; (4) data-collection and laboratory analysis procedures for each facility; (5) the year the data were collected, and (6) the State or the USEPA's quality-assurance plan at the data-entry level. Major public wastewater-treatment facilities and major industrial dischargers receive highest priority for the USEPA's QA/QC activities. Less attention is paid to small industrial dischargers. Checking information on the discharge permit is the responsibility of the USEPA, regardless of whether USEPA or the State is responsible for maintaining the monthly records data base. Since 1990, USEPA Region I has made a concerted effort to improve the data base's reliability. Because each facility uses a separate laboratory to analyze its samples, the data inevitably include multiple detection limits. However, concentrations of nutrients in effluent generally are high enough that few if any data are less than the detection limit for constituents of interest.

Connecticut Department of Health Services Pesticide Monitoring Program

The Connecticut Department of Health Services (CTDOHS) regulates the quality and adequacy of the States's public-water supplies under Section 25-32 of the Connecticut General Statues. Pursuant to that responsibility, the Water Supply Section of the CTDOHS monitors, or requires monitoring, of about 240 reservoirs and streams and about 1,600 wells that provide potable water for about 700 community water-supply utilities that serve 82 percent of the State's population (Prisloe and Sternberg, 1983, p. 3). In Connecticut, a community water supply is defined as any water system that serves at least 2 residences or 25 residents. Community water supplies range in size from large investor-owned water companies or municipal water departments to State institutions, residential developments, and seasonal communities. About 100 of these systems serve at least 1,000 people, and collectively supply 95 percent of all public-water-supply recipients (Healy and others, 1990, p. 196).

Since 1976, the Federal Safe Drinking Water Act and, subsequently, the Connecticut Public Health Code have required analysis for selected pesticides. All surface-water supplies have been sampled for endrin, lindane, methoxychlor, toxaphene, 2,4-D, and 2,4,5-TP (Silvex) under Federal regulations and the State has required that EDB (ethylene dibromide) as well as other VOCs be sampled. Large community supplies (those serving more than 1,000 people) from surface-water sources are tested more frequently than small supplies, but all surface-water supplies must be sampled at least once every 3 years. Federal statutes have not required analysis for pesticides in ground-water sources, but the CTDOHS has required that VOCs be sampled for in every system every 3 years. Large ground-water supplies are tested approximately annually and CTDOHS also conducts random sampling of about 100 community supply wells each year (P. Ritsick, Connecticut Department of Health Services, oral commun., 1992). Frequently, the pesticides identified above also were analyzed in ground-water samples. Occasionally, additional pesticide compounds (including alachlor, aldicarb, atrazine, and metabolites, carbofuran, chlordane, and dieldrin) have been analyzed in surface- and ground-water supplies from areas close to agricultural activities. In addition, some community water supplies conduct their own monitoring and may report these additional samples to the CTDOHS.

The CTDOHS maintains the data from public-water-supply monitoring in paper files at the Hartford office of the Water Supply Section. For this investigation, the USGS inventoried information on pesticide analyses available in the CTDOHS active files during the summer of 1992. The period of record on file varied for different community water supplies. Generally, files maintained on small utilities (less than 1,000 people served) contained data from 1979 through 1992, whereas files on large utilities typically contained data for 1988 through 1992 only.

All CTDOHS pesticide data included in the active files were inventoried for this report, except for analyses that indicated the sample was from a mixed source (either multiple surface-water or ground-water sources, or a mixture of surface- and ground-water sources), or that indicated the source sampled was treated (chlorination, fluoride augmentation, filtration, softening, and control of iron, manganese and pH). Samples generally were collected from the intake to the treatment plant, from near the well head (for treated ground-water supplies), or from some point in the distribution system (for untreated ground-water supplies). In total, data for 75 surface-water sources (mostly reservoirs, but including some stream diversions) and 238 community-supply wells were inventoried for this report. These data include analyses for one or more of 16 pesticide compounds in 433 samples collected from 1979 to 1992.

There is little information readily available concerning field and laboratory QA/QC practices and protocols for the sample collection and analysis of pesticides in public-water supply. The CTDOHS data include samples collected by agency staff, utility staff, servicing laboratories, and consultants. Analyses are performed by as many as 23 State-approved laboratories. Monitoring conducted in compliance with the Safe Drinking Water Act (Public Law 93-523) is required under Section 1401(1) of the Act to include "check samples and special samples in appropriate case" (U.S. Environmental Protection Agency, 1975, p. 59569). The USEPA mandates use of State-approved laboratories that conduct analyses for specific pesticide compounds using USEPA-approved methods. Connecticut Public Health Code (Connecticut Department of Health Services, 1985) similarly requires "representative samples....be tested in accordance with regulations of the Federal Environmental Protection Agency."

National Atmospheric Deposition Program

The National Atmospheric Deposition Program (NADP) of the Natural Resource Ecology Laboratory, Colorado State University, Fort Collins, Colo., compiled data on concentrations and net deposition of major ions in precipitation at six regional stations in New England and New York for 1980-91. Nutrient species reported include nitrate and ammonia.

Data from all six sites were screened for completeness during the period of record. Observations not meeting the criteria, which included percentage of period represented both in volume of precipitation and in time of collection, were excluded from further analysis. This screening did not eliminate any sites or constituents.

The NADP maintains an extensive QA/QC program, affecting field operations, laboratory activities, and data base management (Aubertin and others, 1990). The field operations quality-control activities include external audits of field parameters, performance audits, measurements of dynamic blanks (measurements made on ion content of collection buckets during weeks of no precipitation), operation of duplicate sampling stations, and audits of site configuration and equipment. Laboratory QA/QC procedures include duplicate analyses of samples (2 percent), analyses of standard reference materials, internal blind samples, and equipment blanks, and maintenance of a quality-control officer at the laboratory. All national samples are handled and analyzed by the same central laboratory. Data-management quality-control procedures are used for reviewing field forms for internal consistency, checking data entry, flagging of suspected contaminated samples, verifying lab data at sampling site, and checking rain gauge charts against information from field forms.

Acid Rain Monitoring Project

Volunteers, trained by professional staff from the University of Massachusetts—Amherst Water Resources Research Center (WRRC)—collected major ion samples and physicochemical data at more than 5,000 sites in Massachusetts for the WRRC's Acid Rain Monitoring Project (ARMP) during 1984-91. These sites primarily included ponds and small streams, as well as a number of large rivers. Data were collected at many sites on an irregular basis; 800 ponds and streams have been sampled quarterly from 1986 to 1991. Analyses for nitrate, the only nutrient species included in the study, were performed on unfiltered grab samples.

Sites on streams and rivers in the study unit with 10 or more nitrate measurements for the period of record were selected for analysis for this report. Of the original 5,000 sites, 1,179 were stream or river sites,

and of these, 185 were selected for analysis. Some of these sites were designated as "upper" and "lower" although they occurred at the same locations. For screening purposes, all observations at a single site were combined to represent one location.

Quality control of the samples included spikes, duplicates, and blanks. According to the WRRC, the data conform to USEPA QA/QC standards. Concentrations less than the detection limit were originally assigned zeros in the data base. Based on the lowest reported nitrate concentrations (0.2 mg/L), these values were reassigned as less than 0.2 in this analysis.

Data Analysis

Choices of analytical methods are based on the reliability, quantity, and distribution of the data. Sites where data are minimal, or where quality-assurance procedures are unknown, are used only in qualitative analyses, such as site location maps, or areal distributions of constituent detections or median concentrations. Long-term sites with a well-documented history and established quality-assurance procedures are used in quantitative analyses such as trend analysis and load estimation. All maps, graphs, and tables indicate the source of the data, because of the wide variability in documentation available for the different sources.

Areal Distributions

Most areal compilations and analyses used a Geographic Information System (GIS). Maps have been used to display locations of surface- and ground-water sites in the NWIS, STORET, NPDES, and ARMP data bases sampled for nutrients or suspended sediment, and to display the areal distribution of median concentrations of nutrients and suspended sediment.

For the 18 major surface-water-quality stations selected from NWIS, land use in each basin was classified according to the Anderson Level II classifications (Anderson and others, 1976), and basin totals for each classification were prepared, using a GIS. The number and major types of NPDES point-source dischargers within each of these 18 basins also were documented.

Maps showing locations of sites in the NWIS, STORET, and CTDOHS data bases where pesticides were sampled and detected were prepared using GIS. NWIS and STORET sites that were coded as streams, lakes, or estuaries are shown on the same maps. The CTDOHS reservoir and public-supply well locations are shown together on a single map. NWIS ground-water well and spring sites sampled for pesticides are shown on separate maps.

Categorical Distributions

Evaluation of sample and site distribution made extensive use of bar graphs and clustered bar graphs. In some cases, numerous graphs for sites or constituents were inspected and compared to select representative graphs for inclusion in this report.

Distribution of surface-water samples for several constituents at 18 NWIS stations were evaluated by year, month, and decile of long-term flow duration. Numbers of ground-water samples in NWIS were evaluated by aquifer type, well type, water use, well depth, water level, and sample depth, comparing sample counts for nutrients to sample counts for all constituents. Permitted NPDES dischargers were categorized by type of facility: sewage-treatment plants, industries, paper and paper-products plants, utilities, and services. Facilities in each category were enumerated.

Bar graphs and clustered bar graphs were used to describe the temporal distribution of samples or the distribution of pesticide detections that relate to various site categories, for example, land use or aquifer type. Numbers of ground-water samples in NWIS were evaluated by period of record, aquifer type, well type, water use, well depth, water level, and sample depth, contrasting the number of pesticide samples with sample counts for nutrients and all constituents. Detection frequencies of selected pesticides were evaluated by land use and crop type.

Summary Statistics

Summary statistics, including median concentrations and other concentration percentiles, are presented to describe selected nutrient data for surface- and ground-water sites in NWIS and STORET. The number of observations and the 75th, 50th, and 25th percentiles were calculated for five nitrogen, three phosphorus, and one suspended-sediment water-quality constituents for 83 NWIS surface-water stations with 15 or more observations for a constituent. Because of the uncertain quality of the STORET data and the

variability in the numbers of samples at each station, only the median concentrations and number of samples were calculated and tabulated for STORET stations. Further analyses include the ranges of median concentrations and their site locations in figures of the study unit. The number of observations and median concentrations were calculated for three nitrogen and three phosphorus water-quality constituents for 109 STORET water-quality stations where there are 10 or more samples for a constituent. Median concentrations of nitrate in precipitation were calculated for each site selected from the ARMP data base. Information on the number of NWIS and STORET ground-water sites and samples with nutrient data is summarized. The minimum and maximum concentrations, as well as the 10th, 25th, 50th, 75th, and 90th percentiles, were reported for 9 nitrogen and 10 phosphorus water-quality constituents in ground water at each site where there were at least 10 samples.

Information on the frequency of detection and the range in concentration of pesticides detected in samples of whole water and bed sediment from water-quality stations reported in NWIS and STORET, and for ground-water samples from NWIS sites, is summarized for this report. Because pesticide detections generally are infrequent, pesticide concentration distributions are strongly censored at the analytical detection limit, and are not amenable to characterization or estimation of median concentrations or other summary statistics.

Site Comparisons

Boxplots were used to compare constituent medians, ranges, and interquartile ranges at eight NWIS surface-water-quality stations, with attention to differences in environmental settings among the station drainage basins. Comparisons also were made with boxplots summarizing national constituent data for surface water. Concentrations of selected nutrients in ground water were compared for data grouped by aquifer type, well type, well depth, and land use. Boxplots were used to evaluate pesticide detections by sample depth.

Hypothesis Tests

Kruskal-Wallis tests (Helsel and Hirsch, 1992, p. 159) were computed to evaluate and compare data on nitrite-plus-nitrate and pesticides in ground water from specific hydrogeologic or land-use settings. The null hypothesis tested is that there is no significant difference among the samples segregated by hydrogeological or land-use settings with respect to nitrite-plus-nitrate concentrations or pesticide detections. Rejection of the null hypothesis at the 0.05 level of significance is evidence of a relation between nitrite-plus-nitrate concentration or pesticide occurrence and the factor tested. Results of the Kruskal-Wallis tests, as with other statistical methods used for this analysis, are reported as the attained significance level (p value), which defines the probability that the measured differences are due to chance rather than the tested effect. Tukey's test, a multiple comparison test, was used to discriminate among populations with measured significant differences (Helsel and Hirsch, 1992, p. 198).

Trend Analysis

Trend analysis of nutrient and sediment data was accomplished using an automated program developed by the USGS. The statistical test used by this program to detect trends is the seasonal Kendall test (Hirsch and others, 1982; Smith and others, 1982). This test is a distribution-free or nonparametric test which requires no assumption of a particular statistical distribution. The magnitudes of the data are used only to rank the data. The probability of obtaining the observed pattern of the ranks is then calculated without the need for estimating the parameters of a distribution (Smith and others, 1982, p. 5). The seasonal Kendall test is based on the nonparametric Kendall's Tau test (Kendall, 1975), which compares the relative values of all possible pairs of data values in a time series. In the seasonal Kendall test, comparisons between data values are restricted to pairs of data values that are from the same time period annually; this period is defined as a "season." The seasonal Kendall test is not adversely affected by nonnormal data distributions, potentially large seasonal variability, values less than the detection limit, missing values, or outliers, all of which are typical of water-quality data.

The seasonal Kendall test also involves hypothesis testing for trend detection. The null hypothesis is that the parameter of interest, such as constituent concentration, and its time of observation are independent, which indicates no trend (Smith and others, 1982). The chance of incorrectly rejecting the

null hypothesis (detecting a trend) when a trend actually does not exist is measured by the probability level (p), also called the attained significance level. For example, a p value of 0.05 means that there is a 5-percent chance of making an error when rejecting the null hypothesis. In this report, test results that produced a p value less than or equal to 0.05 were considered to indicate significant upward or downward trends in constituent concentrations.

For many constituents, much of the variability in concentration is related to variability in streamflow and the effects of different sources of streamflow. The program used for trend analysis removes the effects of streamflow variability on constituent concentrations by computing a time series of flow-adjusted concentrations and testing this time series for trend. Removing streamflow effects from water-quality data increases the power and efficiency of trend-detection procedures (Hirsch and others, 1991).

The software used to perform the trend analyses examines the concentration data to see whether they span the period of analysis and whether the data are sufficient at the beginning and end of the period to make the required number of pair-wise comparisons for most seasons. If data criteria are met, the program determines whether a trend test can be performed, selects the test's seasonal definition (monthly, bimonthly, or quarterly), and performs flow adjustments.

Although a trend with time may be positive or negative, the relation between data values and time may not be linear. Smoothing lines are used to depict patterns in the data values throughout the selected period of record. The smoothing technique is called Locally Weighted Scatterplot Smoothing, or LOWESS (Cleveland, 1979; Helsel and Hirsch, 1992, p. 94-98). The LOWESS smooth provides graphical information about short-term changes in water quality within the selected period of record, whereas the trend indicates the general relation between values in the early part of the period of record and values near the end of the record.

Load Estimation for Surface-Water Constituents

A minimum variance unbiased estimator computer program (Cohn and others, 1992) calculated total nitrogen and total phosphorus loads at six water-quality stations in the study unit. Site selection was based on adequacy of the data for analysis. Stations with the best discharge and water-quality records, which included, at a minimum, water years 1980-90, were selected for analysis.

Stations used in the load estimation procedure were selected from the eight stations listed in table 7. These stations are part of a network set up to monitor ambient water-quality conditions and evaluate long-term trends. The regular sampling schedule in any given year may not encompass the major storm discharges that carry a large proportion of the annual load for some constituents. Further, even when water-quality samples are collected during high streamflow, the sampling time may not coincide with the peak load of the constituent in that particular storm. Despite these limitations, the records for these stations represent a wide range of streamflow conditions, including high flows, over a long period of record.

Development of a suitable model involves selecting a number of regression variables, generally related to streamflow and time, to calibrate the model. Individual models were created for each constituent at each station. As recommended, the longest possible continuous period of data collection was selected for calibrating the several models (T.A. Cohn, U.S. Geological Survey, oral commun., 1993). Low, median, and high flow years, based on annual mean discharge, were determined for each station. Because of the variability in factors affecting discharges among these stations, the same years could not be chosen to exemplify low, median, and high flow years for all stations; that is, each station had its own set of low, median, and high flow years.

Atmospheric Deposition Estimation

Estimates of total atmospheric deposition of nitrate and ammonia (the combination of wet, dry, and droplet deposition) were based on wet-deposition data collected at nearby regional NADP sites. Application of correction factors derived from a National study of atmospheric nitrate deposition (Sisterson, 1990) enabled inference of the dry and droplet components of total nitrate deposition. Studies have not been conducted on the distribution of ammonia deposition, so ammonia is presented here as uncorrected wet deposition and all discussion of deposition corrections (urban,

dry, and droplet) refers to nitrate only. In addition, urban effects added another level of complexity to the estimation of total deposition from wet deposition. Minimal urban influences were expected at NADP sites. Because deposition tends to be greater in urban areas, corrections were applied in this study to account for the amount of urban area in each drainage basin.

NADP wet-deposition data were available on a quarterly composited and an annual basis for each monitoring site. The median of quarterly composited data for each site, multiplied by four, was used as an estimate of the annual areally based deposition for each NADP site. These estimates were used to calculate regional (nonurban) wet deposition for each of the 18 basins, in the following manner: first, the geographic centroid of each basin was estimated visually using basin boundaries drawn at a scale of 1:250,000; then, the estimate of regional wet deposition at each centroid was obtained from a spatially weighted average of the annual areal-deposition estimates from the three NADP sites nearest the basin centroid.

The ratio of wet to dry nitrate deposition was calculated for each basin from Sisterson's (1990) wet- and dry-deposition estimates. This ratio was used to calculate areal regional dry deposition from the estimate of areal nonurban wet deposition. The total nitrate load to each basin for wet and dry regional deposition was then estimated by multiplying the areal estimates times the area of each basin. Urban areas are estimated to have 1.75 times as much wet deposition and five times as much dry deposition as nonurban areas. Using GIRAS data, the percentage of urban area for each basin was calculated. These percentages were used to correct the regional, nonurban-based nitrate load estimates for urban effects.

Droplet deposition can be a significant proportion of total deposition at high altitudes. The area of each basin exceeding 2,000 ft was multiplied by the areally based wet plus dry nitrate deposition and by an empirically derived correction factor (Sisterson, 1990).

NUTRIENTS, SUSPENDED-SEDIMENT, AND PESTICIDE DATA-COLLECTION SITES AND FREQUENCIES

Accurate characterization of water-quality conditions in the study unit, and reliable analyses of the distribution of constituent concentrations, constituent trends, and loads, require data that are well distributed with respect to several factors:

- major physiographic, hydrologic, geologic, and land-use features of the study area,
- period of water-quality record,
- seasons,
- range of streamflow conditions,
- major aquifers, and
- varying conditions along ground-water-flow paths.

This section of the report analyzes and evaluates existing surface-water and ground-water-quality data in terms of these major distributions, and notes important gaps in the data.

Nutrients and Suspended Sediment

Surface Water

Nutrient data are available for a substantial number of water-quality stations in the study unit. The number of analyses available at an individual location generally is much larger for surface-water-quality stations than for ground-water sites. Suspended-sediment data are less abundant than nutrient data at most water-quality stations.

Spatial Distributions

The USGS's NWIS and USEPA's STORET data bases yielded 228 water-quality stations with a minimum of 10 samples during the study period for at least one water-quality constituent (table 23 at back of report; pl. 1). These agencies have conducted different types of data-collection activities to meet various agency responsibilities. The requirements of these programs dictate the placement and number of sampling sites and affect the applicability and availability of the data for broad-scale interpretation.

The USGS stations (pl. 1, map reference numbers 1-119) are primarily along major streams and rivers in the States of Connecticut (75 stations), Vermont (23 stations), Massachusetts (17 stations), New Hampshire (3 stations), and New York (1 station). NWIS stations include long-term stations representing major drainage basins or tributaries, as well as local clusters of stations used for special studies of more limited duration and scope. These station locations were selected through cooperative studies and

programs with State and local agencies, or in some cases through Federal programs. Cooperative surface-water-quality studies between the States of New Hampshire and Vermont and the USGS have not been as extensive as those between the USGS and the State of Connecticut. Thus, the number of NWIS stations varies considerably in different parts of the study unit, and little or no water-quality data are available for a number of major streams, particularly in the northern part of the study unit (pl. 1).

Sites from STORET (pl. 1, map reference numbers 120-228) are in Vermont (46 sites), New Hampshire (20 sites), Massachusetts (24 sites), and Connecticut (18 sites). Many of the STORET sites are clustered along stream reaches with known or suspected water-quality problems. As is the case with NWIS stations, the STORET sites do not represent all major subbasins in the study unit (pl. 1).

Data from 185 sites in the Acid Rain Monitoring Program of the University of Massachusetts-Amherst were appropriate for use in this study (fig. 9). ARMP water-quality monitoring sites in the study unit primarily are in Massachusetts; a few sites are on streams in Connecticut. In Massachusetts, drainage basins of all major tributaries to the Connecticut, Housatonic, and Quinebaug Rivers include some of these sites. ARMP sites also are on minor tributaries to the Connecticut River as well as the Connecticut River itself.

Temporal Distributions

The temporal distribution of samples collected at each NWIS and STORET station was examined in terms of two characteristics: (1) the length of the water-quality record, and (2) the sampling frequency or seasonal distribution of samples. The period of record and approximate sampling frequency are shown in table 23 for each station.

The period of record for NWIS stations ranged from less than 2 years to more than 20 years, reflecting the inclusion of long-term monitoring stations as well as short-term project stations. Sampling may not have been continuous during the period of record (table 23). Only 11 stations, all in Connecticut, were sampled continuously during the 21-year study period of water years 1972-92.

Sampling frequency has varied considerably in the study unit (table 23). Sampling frequencies for individual water-quality constituents may differ at a given station, and sampling frequency has varied during the period of record at most stations. Most long-term NWIS stations in Connecticut were sampled monthly (fig. 10). However, sampling in the month of February was discontinued during water years 1983-91. Beginning in water year 1992, stations in Connecticut were sampled bimonthly during the first 8 months of the water year (October-May) and monthly from June through September. A few Connecticut stations were sampled on a bimonthly or quarterly basis. During water years 1988-90, several Connecticut stations, including seven stations evaluated in this study, were sampled for nutrients on a biweekly basis as part of the Long Island Sound Study (1990). Some NWIS stations in Massachusetts and Vermont were sampled on a monthly basis, but only for 6 or 7 months of the year.

Distribution in Relation to Streamflow

Sampling frequency in relation to flow duration was examined for five nitrogen, three phosphorus, and one suspended sediment water-quality constituents at the eight major NWIS water-quality stations (table 7). For the most part, data for the same water-quality constituents were available at the major stations. However, at some stations different analyses were routinely performed, making it necessary to examine alternative constituents. For example, data on total ammonia (parameter code 00610) were evaluated for stations with little or no data on dissolved ammonia (parameter code 00608). Similarly, data on dissolved nitrate (parameter code 00618) are limited. Consequently, data for either total nitrite-plus-nitrate (parameter code 00630) or dissolved nitrite plus nitrate (parameter code 00631) have been substituted. In addition, dissolved phosphorus (parameter code 00666) is discussed, because its record was more extensive than the otherwise preferable dissolved orthophosphate (parameter code 00671). Examination of data for stations in the study unit indicated that substitution of these alternative constituents would not affect data interpretation.

Distribution of data over the range of discharge conditions was evaluated using long-term flow duration data at or near a water-quality station. Flow duration refers to the percentage of time a specified discharge is equaled or exceeded. For example, the 70-percent flow duration is the daily mean discharge that is equaled or exceeded 70 percent of the time.

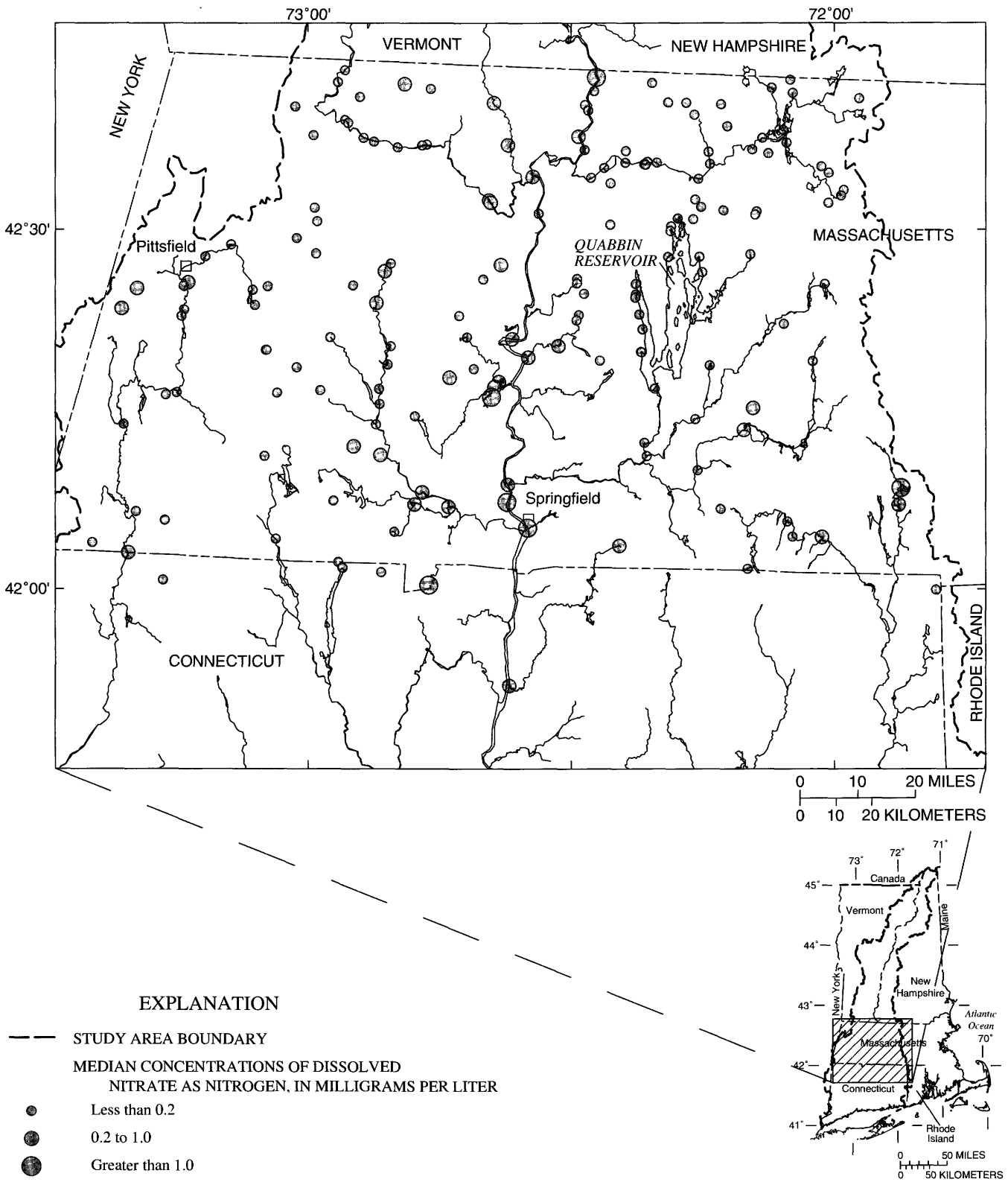


Figure 9. Areal distribution of median concentrations of dissolved nitrate as nitrogen at stream sites sampled by the Acid Rain Monitoring Project, 1984-91. (Data from the University of Massachusetts Water Resources Research Center.)

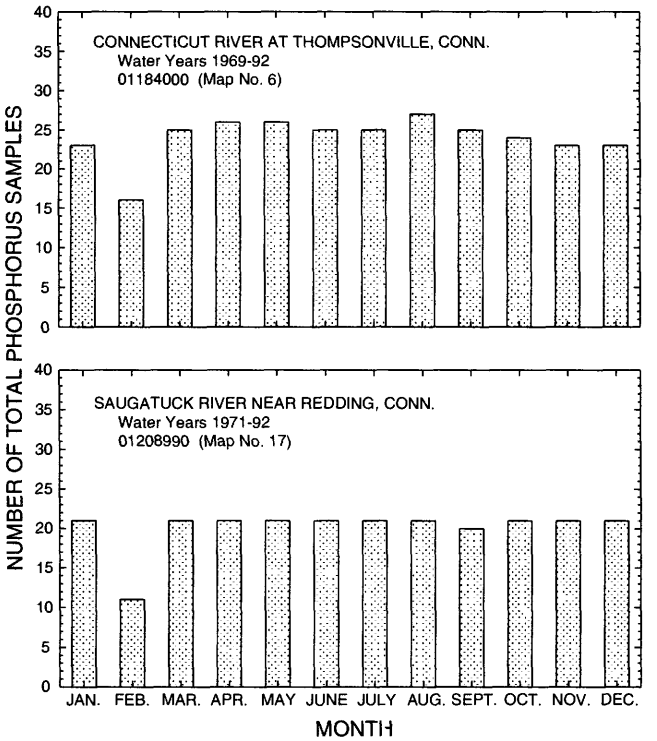


Figure 10. Number of monthly samples of total phosphorus at two selected stations on the Connecticut and Saugatuck Rivers in Connecticut.

Distribution of data for a particular constituent has been evaluated by dividing the long-term flow duration at a station into deciles (10-percent segments) and determining the constituent sampling frequency within each decile of flow, based on the stream discharge associated with each water-quality sample.

The distribution of total phosphorus samples by decile of long-term flow at eight water-quality stations (fig. 11) generally matches the frequency distribution for the other nutrients at the same station, although most nutrients were sampled less frequently than total phosphorus. Sampling frequencies for suspended sediment, available for four of these stations, generally were similar to nutrient sampling frequencies at the same station. The 0 to 10 decile represents infrequent, high-flow conditions (flows exceeded 10 percent of the time or less) and the 91 to 100 decile represents low-flow conditions exceeded most of the time. If sampling were evenly distributed with respect to flow, then each decile would include 10 percent of the samples, and all bars would be of equal length.

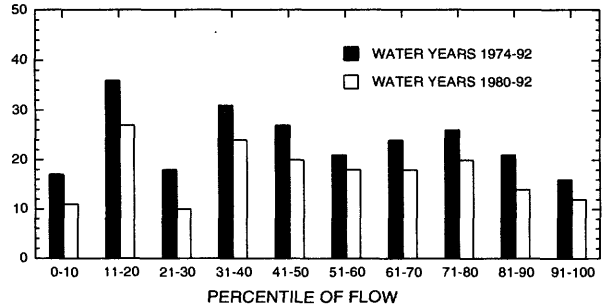
Sample distribution should be relatively even to avoid biasing a data analysis so that high flows, which may carry the largest loads of some constituents, are not under-represented, thereby causing underestimates of loads. With few exceptions, each decile of flow was well represented with samples for the eight stations selected for close scrutiny. The distribution was not ideal in all cases. Discharges at two stations, the Housatonic River at Stevenson, Conn., and the Connecticut River at North Walpole, N.H., are affected by regulation at hydropower dams immediately upstream. Flow-duration data for these two stations exhibit skewed or discontinuous patterns (fig. 11), resulting from controlled hydropower-generation discharges.

Sampling frequencies at the eight selected stations exhibit several similarities and differences (fig. 11). For stations on the Salmon, Connecticut, and Quinnipiac Rivers, in the central part of the study unit, the sampling frequency was higher for discharges greater than the median streamflow. Several of the stations were sampled relatively infrequently at very low flows. The frequency distribution on the Naugatuck and Saugatuck Rivers, in the western part of the study unit, is fairly uniform. By contrast, the bimodal distribution for the Housatonic River at Stevenson shows the effects of streamflow regulation by the hydroelectric power dam immediately upstream from the sampling station; most samples are collected at either very high or very low flows. According to the sampling distribution for the Connecticut River near North Walpole, N.H., few samples were collected when flows were low.

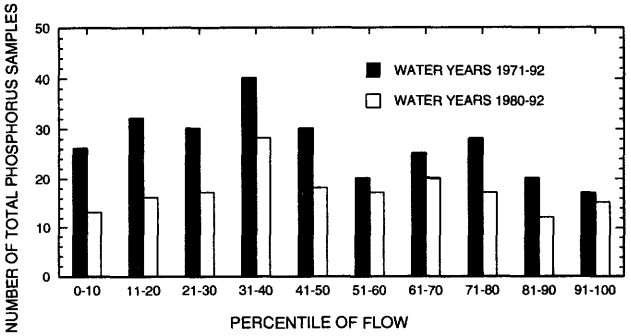
Ground Water

Data on nutrients in ground water in the study unit were fairly abundant and comprehensive. NWIS lists 1,872 wells and STORET has 25 wells where ground water was sampled for one or more of 20 nitrogen or phosphorus water-quality constituents (table 5). In addition, 42 springs in the NWIS data base were sampled for nutrients (table 5). There were, however, considerable differences in numbers, types, locations, and dates of nutrient samples represented.

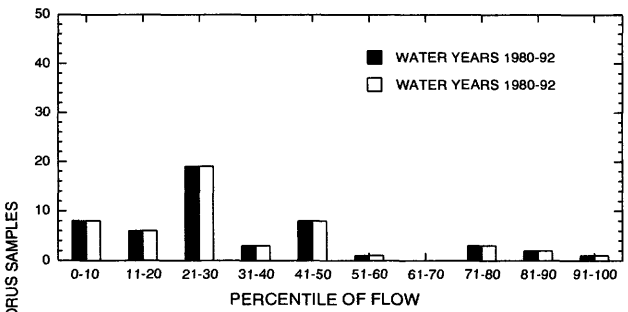
NWIS data were more comprehensive than STORET in the suite of nutrients represented (table 8), including 10 nitrogen and 10 phosphorus constituents, but most NWIS sites were sampled only once for any



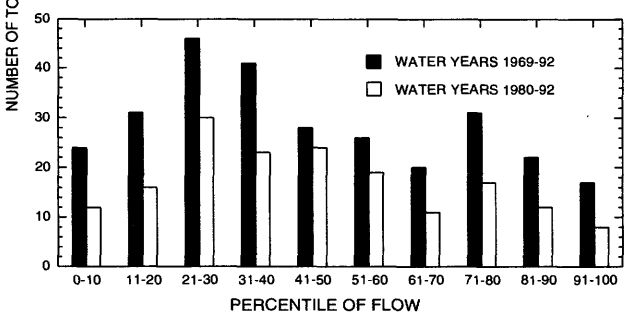
01122610 Shetucket River at South Windham, Conn.
(long-term flow for water years 1929-91)



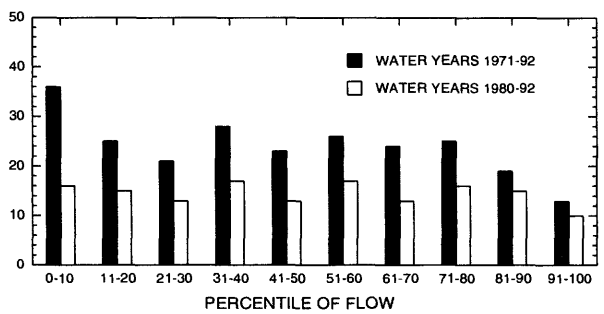
01196500 Quinnipiac River at Wallingford, Conn.
(long-term flow for water years 1931-91)



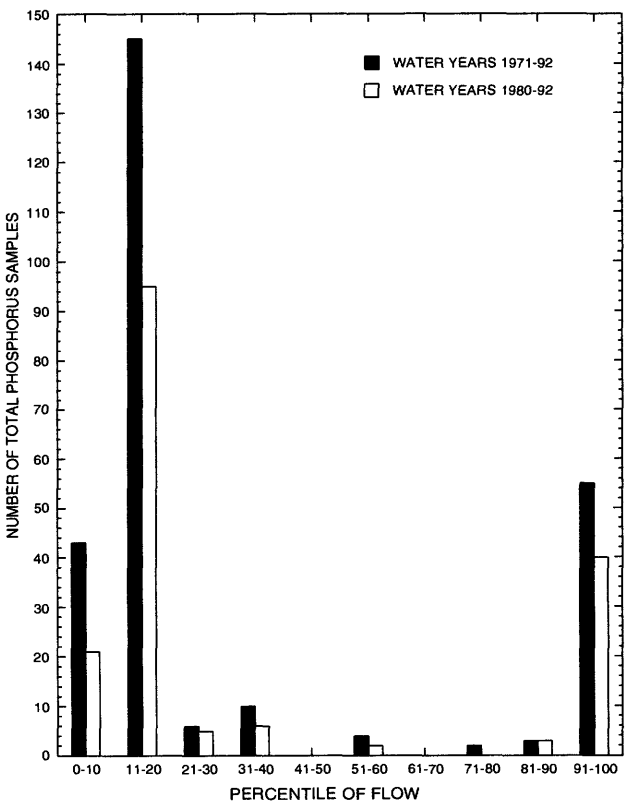
01154500 Connecticut River at North Walpole, N.H.
(long-term flow for water years 1943-91)



01184000 Connecticut River at Thompsonville, Conn.
(long-term flow for water years 1929-91)



01193500 Salmon River near East Hampton, Conn.
(long-term flow for water years 1929-91)



01205500 Housatonic River at Stevenson, Conn.
(long-term flow for water years 1929-91)

Figure 11. Distribution of total phosphorus samples by decile of long-term streamflow at selected water-quality stations.

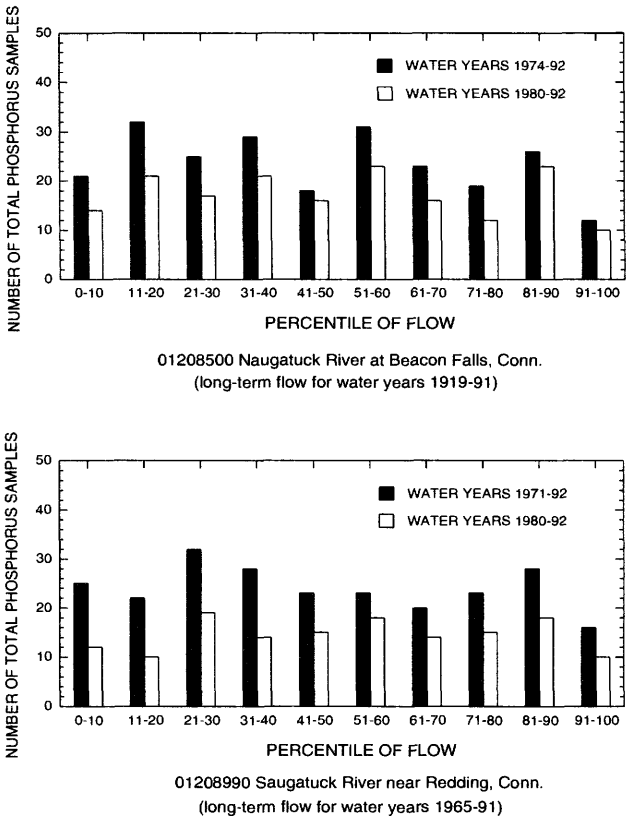


Figure 11.—*Continued.*

particular constituent. STORET data included only five nitrogen and one phosphorus constituent, but individual sites were sampled on 10 or more occasions. In terms of numbers of sites and samples, nitrite plus nitrate as nitrogen (parameter code 00630) was the most commonly reported constituent in NWIS and STORET.

Wells from a number of hydrogeologic categories were evaluated to illustrate the proportion of the 2,357 NWIS wells with water-quality data that also have nutrient data. These comparisons demonstrated the strengths, weaknesses, and biases in the existing ground-water data base with respect to assessing the occurrence and distribution of nutrient concentrations in the study unit.

Spatial Distributions

Most of the 1,872 NWIS wells and the 42 NWIS springs with nutrient data were in Connecticut and Massachusetts (pl. 1). These sites included a wide variety of wells and represent all the principal aquifer types in the study unit. Most NWIS wells were installed for areal appraisals of ground-water availability and included some limited ground-water-quality sampling; they also may have been installed to evaluate known or potential sources of ground-water contamination, such as landfills, industrial discharges, and agricultural activities. No effort was made to eliminate known or potentially contaminated sites from the population of NWIS sites used in this analysis.

Sites in New Hampshire were primarily installed or sampled during a recent cooperative, statewide, stratified-drift aquifer mapping program between the New Hampshire Department of Environmental Services (NHDES) and the USGS (Moore and others, 1994). NWIS water-quality data for the New Hampshire wells and springs reflected ambient conditions in stratified-drift aquifers in the Connecticut River Basin in New Hampshire, and included little or no data for other principal aquifers. Few wells and springs were sampled by the USGS in Vermont. Five wells clustered in northern Vermont (pl. 1) were installed and sampled for nutrients and other water-quality constituents for this study, as were eight wells in western Massachusetts and four wells in north-central Connecticut.

The 25 STORET ground-water sites with nutrient data were located exclusively in New Hampshire and Vermont and most are near the study-unit boundary (pl. 1). It is not known whether these wells were drilled and screened in stratified-drift, till, or bedrock aquifers; therefore, the only data analysis performed was the enumeration of samples per nutrient constituent (table 8).

Temporal Distributions

The temporal distribution of 2,508 nutrient samples from NWIS wells was similar to the distribution of all NWIS water-quality samples for the period of record (fig. 12) because one or more nutrient constituent commonly was included in the analysis of ground-water samples. The first nutrient sample collected from a ground-water site in the NWIS data base was in water year 1917. However, the next sample was not collected until 1949 and only 145 additional samples were collected through water year 1962.

Table 8. Number of ground-water sites and samples with nutrient water-quality data in the Connecticut, Housatonic, and Thames Rivers study unit

[Nutrients are in milligrams per liter unless otherwise noted. NWIS, U.S. Geological Survey's National Water and Information System; STORET, U.S. Environmental Protection Agency's Storage and Retrieval system. Data on number of samples from wells are for 1917-93 (NWIS) and 1971-84 (STORET); data on number of samples from springs are for 1951-93. µg/L, microgram per liter]

Parameter code	Nutrient	NWIS				STORET	
		Wells		Springs		Wells	
		Sites	Samples	Sites	Samples	Sites	Samples
91003	Nitrogen, nitrate, dissolved (as N) (µg/L)	3	3	0	0	0	0
00615	Nitrogen, nitrite, total (as N)	101	104	2	2	8	14
00613	Nitrogen, nitrite, dissolved (as N)	150	216	13	13	0	0
00630	Nitrogen, nitrite plus nitrate, total (as N)	192	367	9	10	25	340
00631	Nitrogen, nitrite plus nitrate, dissolved (as N)	402	576	12	14	0	0
00610	Nitrogen, ammonia, total (as N)	112	265	1	1	10	19
00608	Nitrogen, ammonia, dissolved (as N)	332	513	20	23	0	0
00625	Nitrogen, ammonia plus organic, total (as N)	56	93	7	8	15	89
00623	Nitrogen, ammonia plus organic, dissolved (as N)	120	181	12	14	0	0
00600	Nitrogen, total (as N)	47	68	7	8	15	317
00665	Phosphorus, total (as P)	98	160	8	11	0	0
00650	Phosphate, total (as PO_4)	135	205	0	0	14	228
00666	Phosphorus, dissolved (as P)	149	257	12	14	0	0
70507	Phosphorus, orthophosphate, total (as P)	121	316	1	2	0	0
00671	Phosphorus, orthophosphate, dissolved (as P)	312	516	5	5	0	0
00669	Phosphorus, hydrolyzable, total (as P)	2	2	0	0	0	0
00672	Phosphorus, hydrolyzable, dissolved (as P)	1	1	0	0	0	0
00678	Phosphorus, hydrolyzable plus orthophospate, total (as P)	4	5	0	0	0	0
00677	Phosphorus, hydrolyzable plus orthophospate, dissolved (as P)	18	36	0	0	0	0
91004	Phosphorus, orthophenol, dissolved (as P) (µg/L)	3	3	0	0	0	0

During water years 1963-72, a large number of nutrient samples was collected during ground-water-resources investigations in Connecticut (Cushman, 1964; Cushman and others, 1964; 1965; LaSala, 1964; 1968; Randall, 1964; Pauszek and Edmonds, 1965; Randall and others, 1966; Thomas and others, 1967; Thomas and others, 1968; Ryder and others, 1970; 1981; Cervione and others, 1972; Wilson and others, 1974; Mazzaferro and others, 1979; Weiss and others, 1982; Handman and others, 1986). Although the present study concerns itself primarily with water years 1972-92, historic data were included because eliminating them would have severely limited the spatial assessment of nutrient conditions in ground water.

Relatively few samples were collected during water years 1973-81. A resurgence in sampling occurred through the 1980's as Grady and Weaver (1988, 1989) conducted studies evaluating the effects of land use on ground-water quality in stratified-drift aquifers and Mullaney and others (1991) assessed the occurrence of pesticides and nutrients in ground water beneath agricultural settings in Connecticut. Few samples have been collected since 1989; the sample count for water year 1993 only included data in NWIS as of December 17, 1992.

Although most NWIS sites were sampled only once for nutrients, some nutrient samples have been collected during all seasons. Examination of the

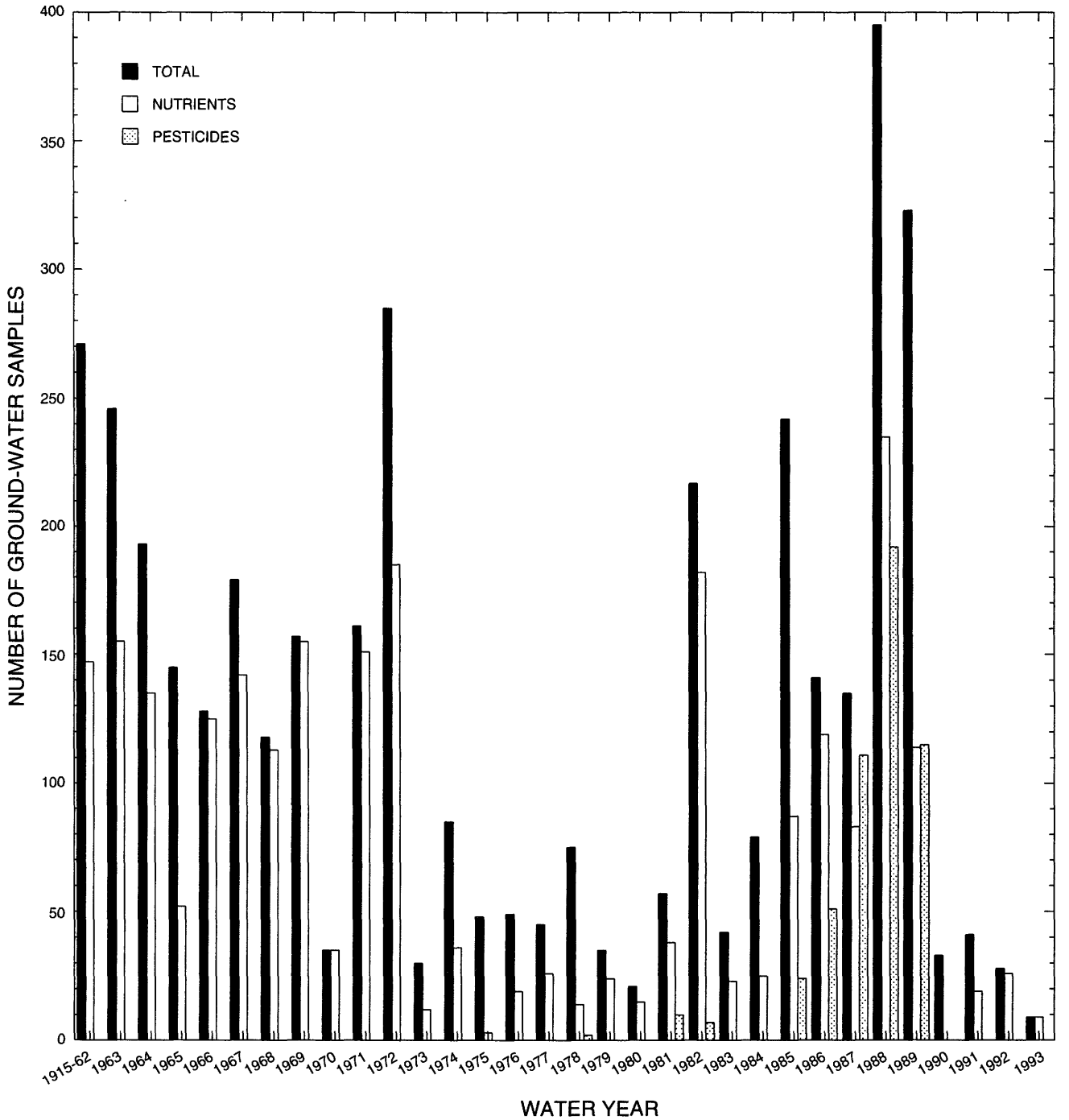


Figure 12. Total number of ground-water samples analyzed for nutrients and pesticides by water year for water years 1915-93. (Data from the U.S. Geological Survey's National Water Information System data base.)

number of nutrient samples collected by month (fig. 13) indicates a bimodal distribution (April and August) with two-thirds (1,674) of the samples collected from April through August.

Distribution by Well Type and Water Use

Most (70 percent) of the wells listed in NWIS were in the category of withdrawal wells (fig. 14). Two-thirds (876) of the withdrawal wells were used for domestic supply (fig. 15); about 15 percent (197) are public-supply wells; and the remainder were used for various purposes including commercial, industrial, institutional, agricultural, or other water-supply needs. Essentially all the withdrawal wells were selected and sampled by the USGS from a much larger population of existing wells inventoried during the course of various ground-water investigations. About 20 percent (375) of the wells sampled for nutrients in the NWIS data were identified as observation wells (fig. 14). These observation wells, together with wells coded as test wells, generally wells installed by the USGS, or unused wells (fig. 14), comprised the 416 wells sampled for nutrients in the "unused" water-use category (fig. 15). Generally, domestic wells were drilled and left as open holes in bedrock aquifers; high-yield, public-supply wells typically were screened in stratified drift and observation and test wells were completed primarily in stratified-drift aquifers.

Distribution by Aquifer Type

The distribution of wells sampled for nutrients by aquifer type (fig. 16) reveals that the largest number of wells sampled for nutrients (768, or 41 percent) are screened in stratified-drift aquifers (see fig. 5). These aquifers are the most productive sources of ground water in the study unit and have been studied more extensively than other aquifers in the study unit. The crystalline bedrock aquifer is the most broadly distributed in the study unit and has the second largest number of wells (644) with nutrient data. Combined, the stratified-drift and crystalline bedrock aquifers account for 75 percent of data on nutrients in ground water from NWIS wells. About 15 percent of the wells sampled are in the arkosic bedrock aquifer underlying the Connecticut Valley Lowland in Massachusetts and Connecticut. The other aquifers (alluvium, till, and carbonate bedrock) are poorly represented in the ground-water-nutrient data, largely because of their limited areal extent or lack of development as a water supply. About 2 percent of the NWIS wells with nutrient data could not be assigned to an aquifer type.

Distribution by Well Depth

An evaluation of the depth of wells sampled for nutrients in the NWIS data indicates a strongly skewed distribution showing a decrease in the number of sites sampled as well depth increases (fig. 17). Nearly one-third (600) of the wells sampled for nutrients are less than 50 ft deep, and 90 percent are less than 300 ft deep. This pattern reflects two factors that have bearing on the distribution of well depths: (1) the preponderance of wells used to sample the shallow stratified-drift aquifers and (2) the general decrease in well yields in fractured bedrock aquifers at depths greater than 300 ft below land surface (Cushman and others, 1953), resulting in few deep wells in New England.

Shallow water tables are prevalent in the study unit; hence, most wells (about 73 percent) are in the categories showing water levels less than 25 ft below land surface (fig. 18). Less than 5 percent of the wells have water levels greater than 50 ft below land surface. More than 30 percent of the wells sampled for nutrients, however, have no water-level data. The prevalence of shallow wells and shallow water tables in the study unit indicates that many of the wells with missing water-level data may be similarly distributed, and indicates that, in general, the set of NWIS wells sampled for nutrients may be appropriate for evaluating the relation of nutrient concentrations in ground water to the overlying land use.

Because the actual depth to the pump intake commonly is unknown or seldom reported, as are the pumping duration, the discharge rate, and other hydraulic data, the part of the aquifer or flow system represented by the sample is difficult to determine. However, calculation of the distance between the water table and the top of the open interval reasonably approximates the sample depth. Although nearly one-half of the wells (799) do not have sufficient data (missing water levels or well-construction data) to calculate the approximate sample depth (fig. 19), the available data indicate that 64 percent of the samples were from wells completed at depths less than 25 ft below the water table.

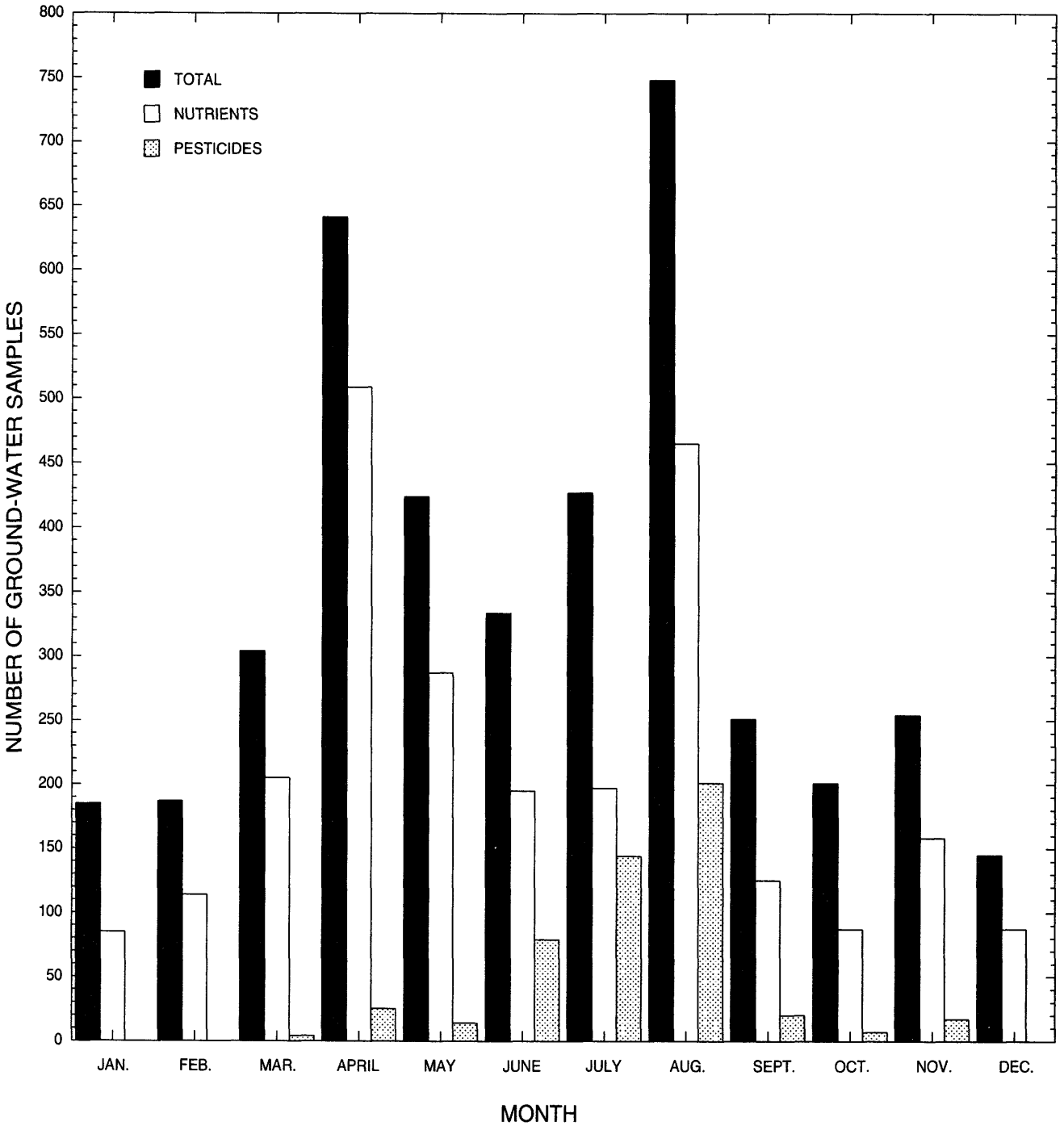


Figure 13. Total number of ground-water samples analyzed for nutrients and pesticides by month for water years, 1915-93. (Data from the U.S. Geological Survey's National Water Information System data base.)

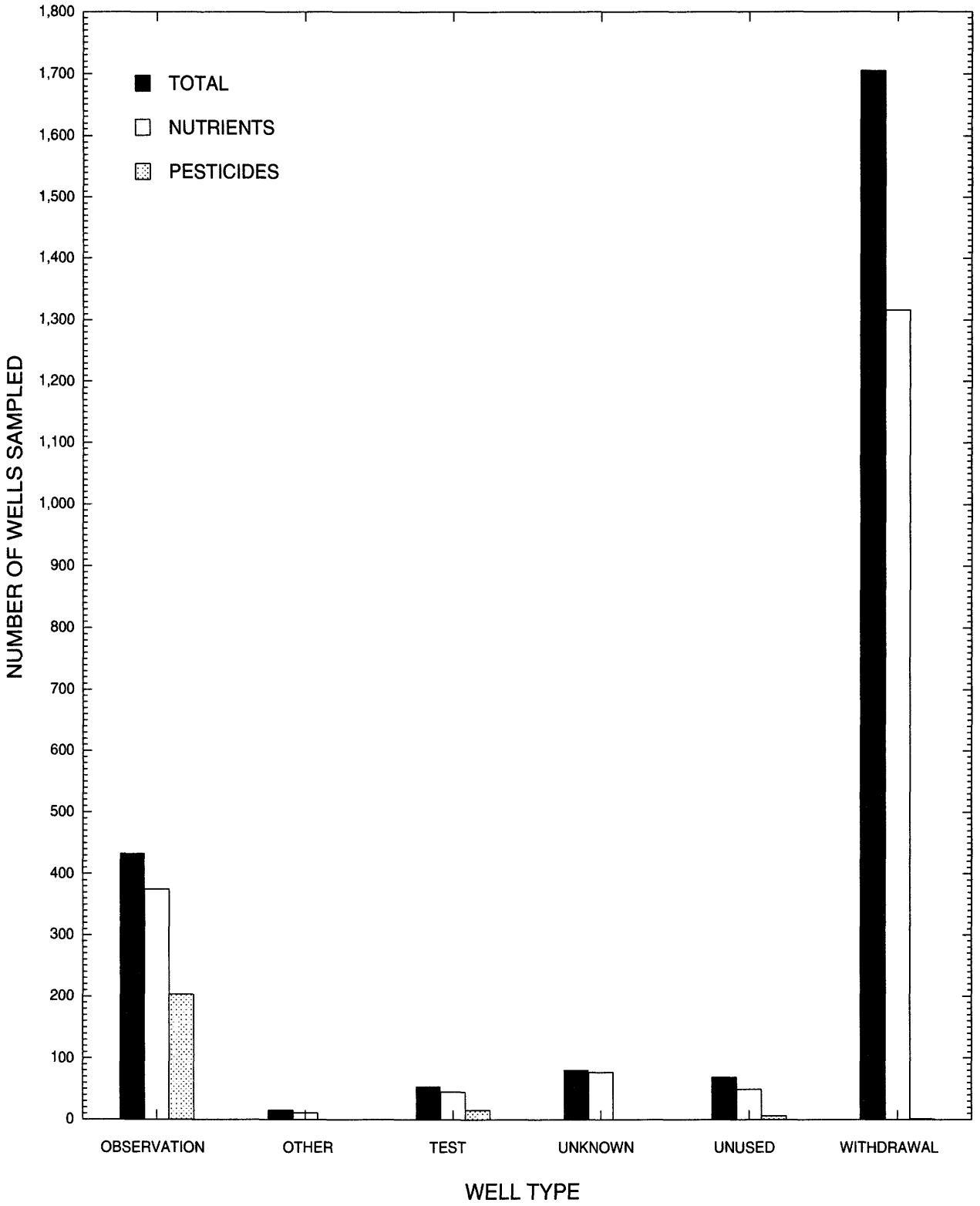


Figure 14. Number of wells sampled for nutrients and pesticides by well type, 1915-93. (U.S. Geological Survey's National Water Information System data base.)

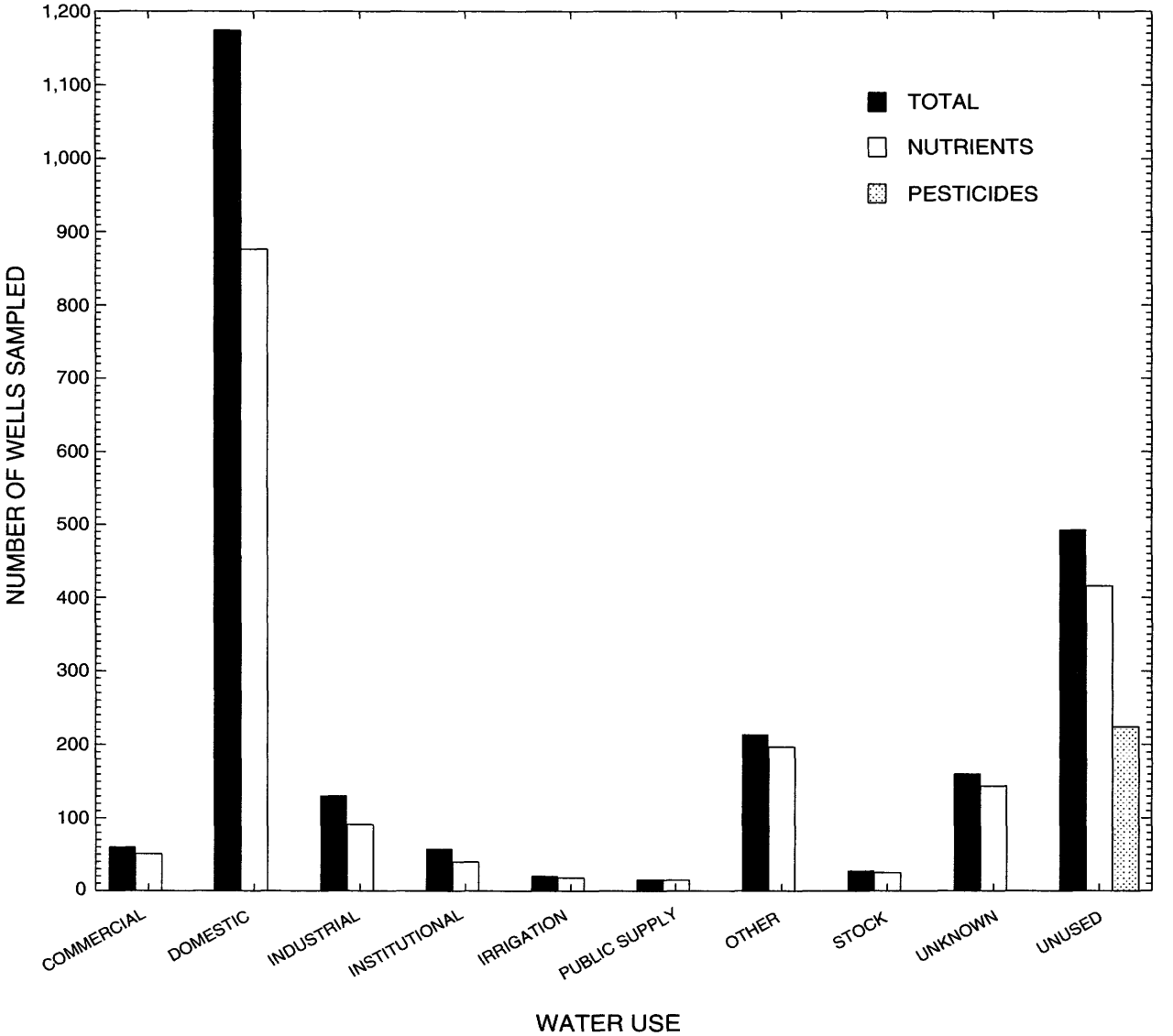


Figure 15. Number of wells sampled for nutrients and pesticides by water use, 1915-93. (Data from the U.S. Geological Survey's National Water Information System data base.)

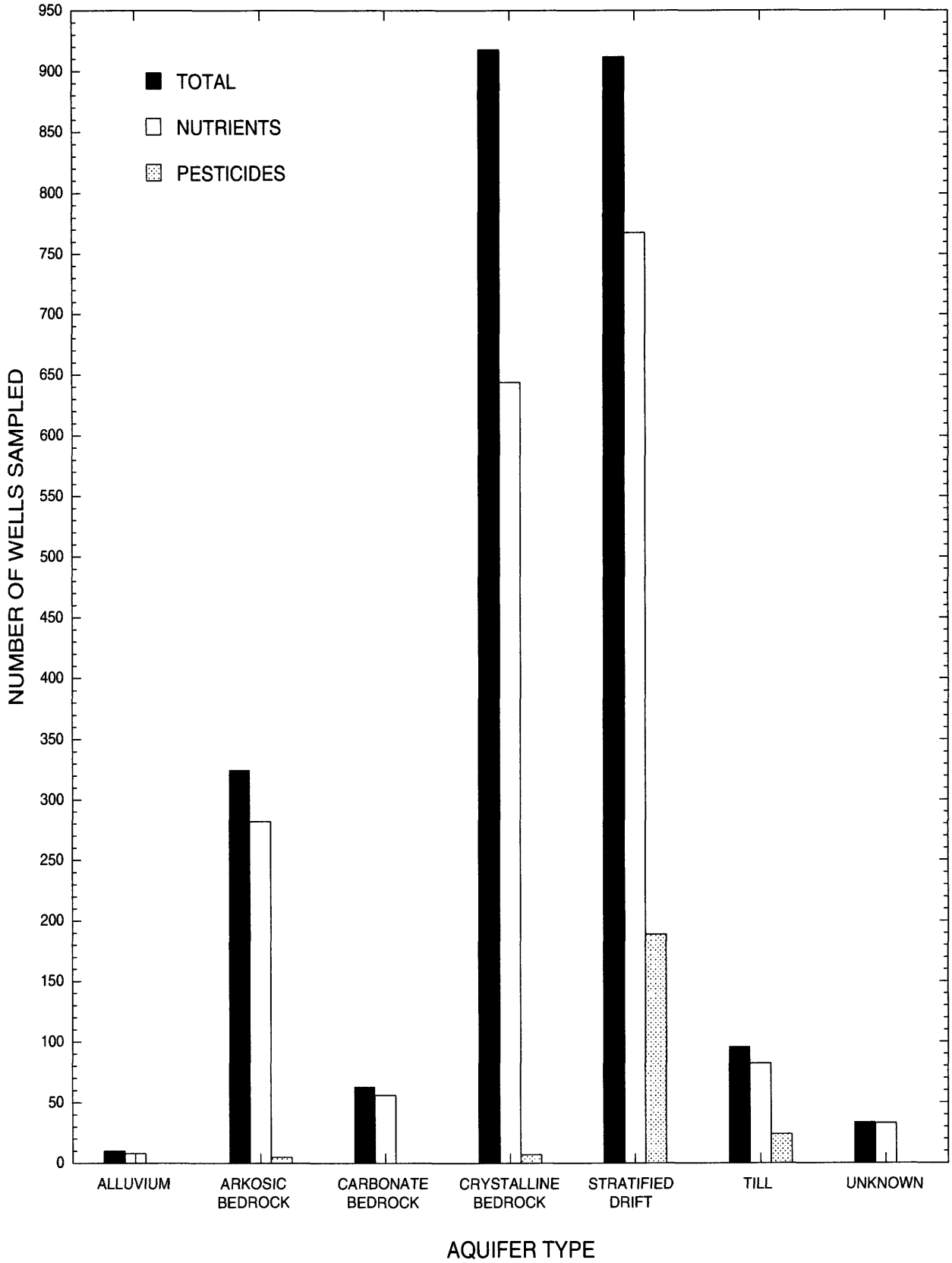


Figure 16. Number of wells sampled for nutrients and pesticides by aquifer type, 1915-93. (Data from the U.S. Geological Survey's National Water Information System data base.)

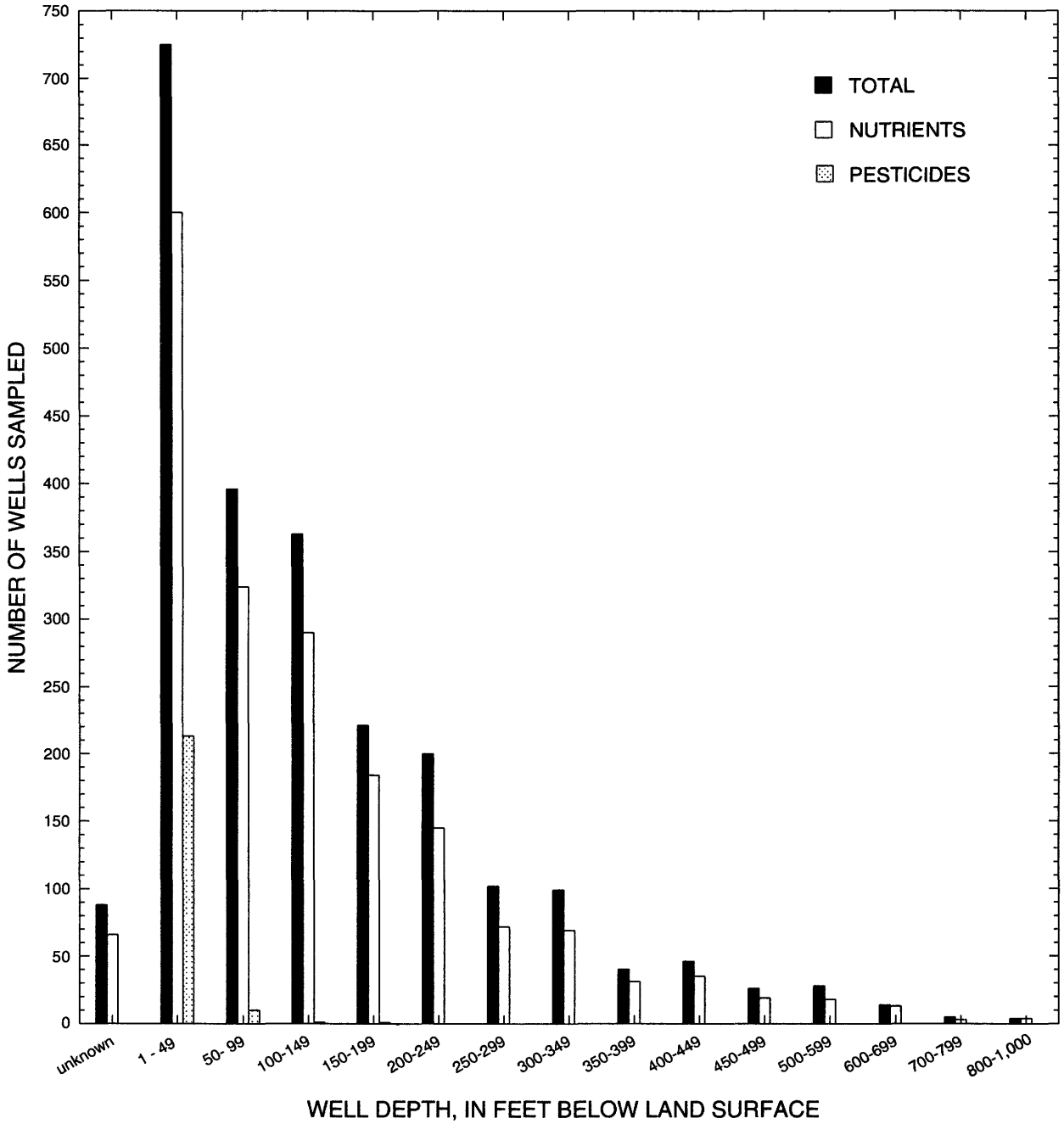


Figure 17. Number of wells sampled for nutrients and pesticides by well depth, 1915-93. (Data from the U.S. Geological Survey's National Water Information System data base.)

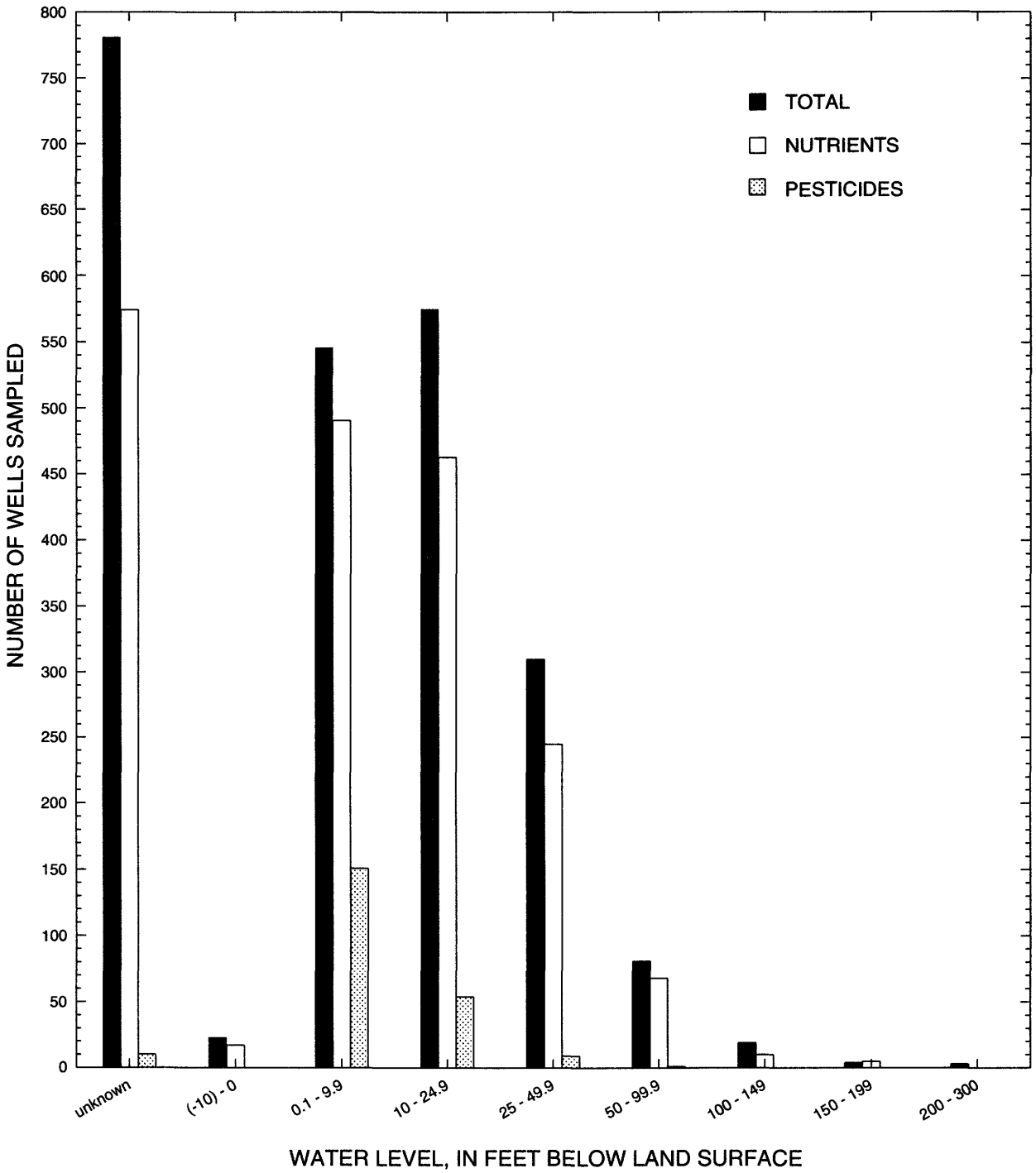


Figure 18. Number of wells sampled for nutrients and pesticides by depth to the water table, 1915-93. (Data from the U.S. Geological Survey's National Water Information System data base.)

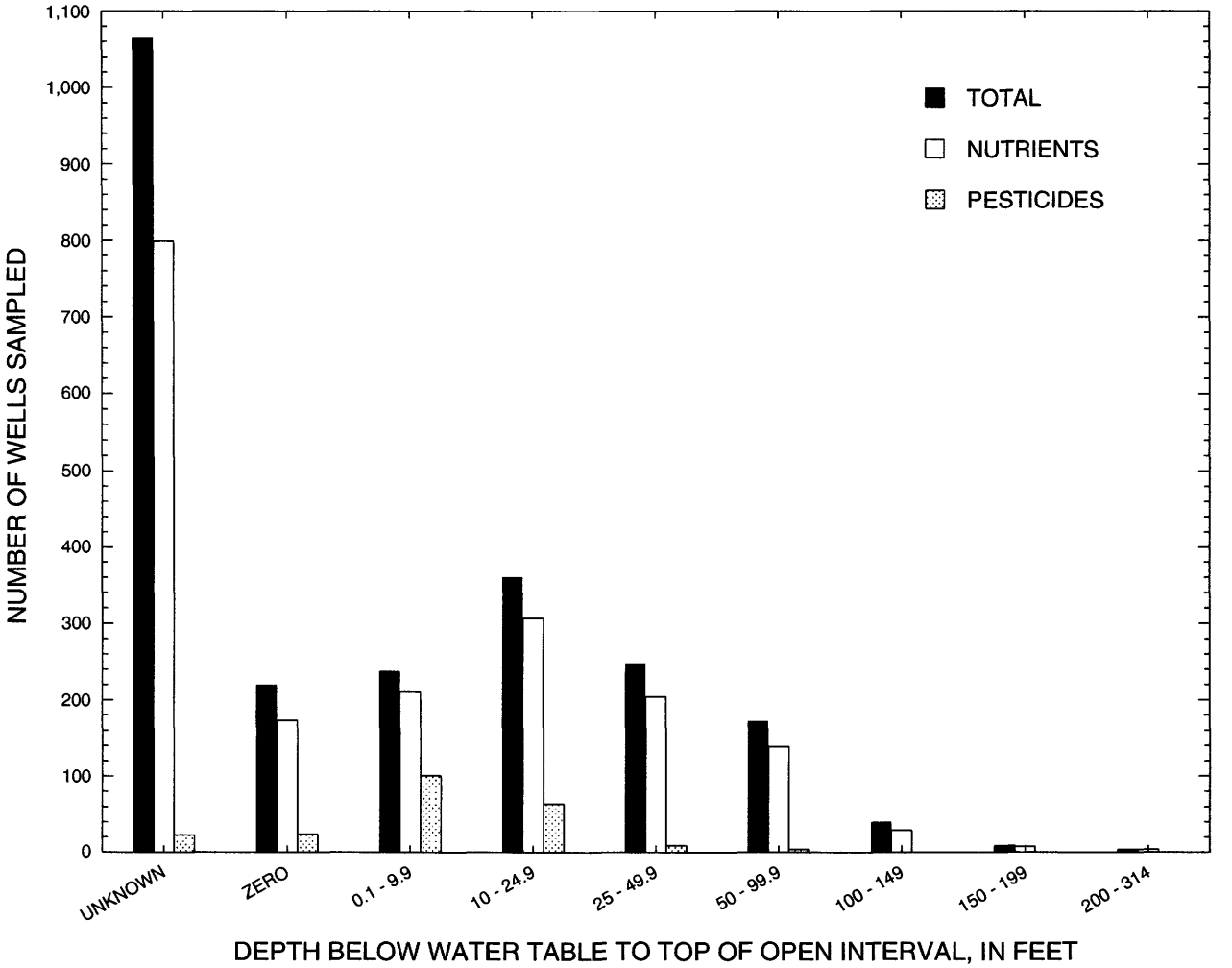


Figure 19. Number of wells sampled for all constituents, nutrients, and pesticides by depth below water table to top of open interval, 1915-93. (Data from the U.S. Geological Survey's National Water Information System data base.)

Pesticides

Data on the occurrence and distribution of pesticides in the study unit were much more limited than nutrient data. Only 90 streamflow-gaging stations (streams) in NWIS, or about 7 percent of the total number of NWIS streamflow-gaging stations with water-quality data (table 5), were sampled for pesticides. Although many of these stations were sampled more than once, few were sampled on more than three or four occasions, and only about 1 percent of all surface-water records included in NWIS had any pesticide data. Twelve NWIS estuary stations and 22 lake stations had some pesticide data. In addition to the NWIS stations, the STORET data base had some limited information on pesticides, including data for 20 stream and 2 lake stations.

Pesticide data were associated with more NWIS ground-water sites, including 225 wells and 8 springs, than streamflow-gaging stations (table 5); however, most wells were sampled only once. No suitable STORET data were available to enhance the NWIS ground-water pesticide data base used in this analysis. Additional data were compiled from CTDOHS monitoring records for community water supplies to augment the pesticide data available from NWIS and STORET. CTDOHS data were obtained for 75 surface-water sources (primarily reservoirs) and 238 public-supply wells.

Chemical analyses for 81 different pesticide compounds or their metabolites in water and bed-sediment samples are included in the combined NWIS and STORET data (table 24, at back of report). In addition to synthetic organic compounds used exclusively as pesticides, a number of volatile and semi-volatile compounds (mostly halogenated aliphatic, monocyclic, and polycyclic aromatic hydrocarbons) are included among the active ingredients of several pesticide compounds, or have subsidiary pesticide uses (Smith and others, 1988). NWIS data are more comprehensive, as STORET data include only 20 of the 81 compounds. The number of sites and samples varies widely for each compound and for the same compound in different sample media. At streamflow-gaging stations, organochlorine insecticides are included in more samples (water column and bed sediments) from more stations than other pesticide types. In ground water, by contrast, triazine and related herbicides as well as organochlorine insecticides are sampled most.

Surface Water

Evaluation of the spatial and temporal distributions of pesticide data for streamflow-gaging stations reveals the limitations inherent in using these data to assess pesticide occurrence and distribution in the study unit. Recognition of these limitations and identification of the scope of pesticide data needed to properly conduct such an assessment are principal objectives of this analysis.

Spatial Distributions

NWIS and STORET streamflow-gaging stations (streams, estuaries, and lakes) where water or bed-sediment samples were collected for pesticide analysis are concentrated in the southern part of the study unit (figs. 20, 21, 22). Only seven streamflow-gaging stations were sampled for pesticide analysis of bed sediments outside of Connecticut. The distribution was nearly as skewed for water-column stations. In addition, a cluster of stations on the Black River, a tributary to the Connecticut River in southeastern Vermont, was sampled during an appraisal of water-quality conditions prior to construction of a hydroelectric power impoundment (Toppin, 1983).

Many of the stations with pesticide data are on the lower reaches of the principal rivers of the study unit, the Connecticut, Housatonic, and Thames Rivers, or their major tributaries. Thus, the stations sample large, complex drainage areas and integrate numerous point and nonpoint sources of contamination. Consequently, there are few locations where pesticides in surface water can be used to relate pesticide occurrence and distribution to specific land uses. In a few places, most notably the Black River in Vermont and the Pomperaug River in Connecticut, a group of closely spaced stations may allow more detailed data analysis (fig. 20).

CTDOHS has monitored pesticides at 75 surface-water sources (reservoirs and some stream diversions) for community water supplies (fig. 22). These locations represent a subsample of about 240 reservoirs and streams that are or have been used for public drinking-water supplies. CTDOHS analytical coverage generally is limited to fewer than 16 pesticide compounds.

Temporal Distributions

During 1969-92, 411 NWIS surface-water (stream, lake, and estuary) samples were collected for pesticide analysis, including 196 samples for analysis of water, 174 samples for analysis of bed sediments, and 41 samples for which both water and sediment were analyzed (fig. 23). About 85 percent of the samples were collected from water years 1972 to 1980. These samples were analyzed mostly for organochlorine and organophosphorus insecticides and chlorophenoxy-acid herbicides. Few additional samples were collected until water years 1991 and 1992, when 13 stations (impoundments) were sampled for polycyclic aromatic hydrocarbons in bottom material, and three streams in the Scantic River Basin in Connecticut were sampled repeatedly during a spring runoff event in 1992 for dissolved triazine and related herbicides as part of this NAWQA study.

The monthly distribution of pesticide samples from NWIS streamflow-gaging stations shows that most samples were collected during the summer months and few samples were collected during the winter months (fig. 24). These data do not include the spring 1992 runoff-event sampling for triazine herbicides in the Scantic River Basin. Water-column samples were collected in 11 of 12 months, with highest numbers of samples in July and the lowest (none) in February. Bed-sediment samples were infrequently collected from November to May when sampling would be inhibited by ice cover or high

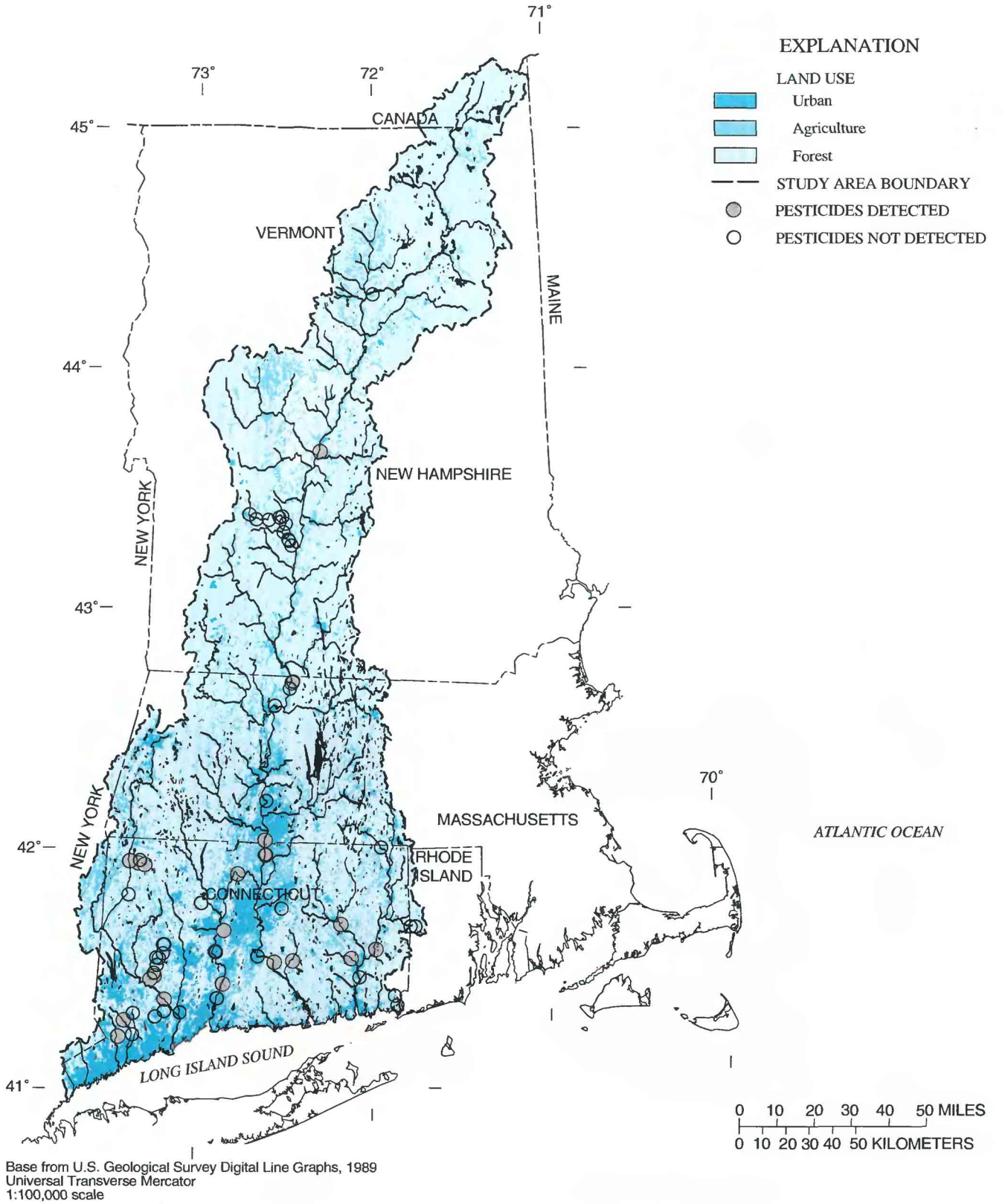


Figure 20. Areal distribution of pesticides at surface-water sites in the Connecticut, Housatonic, and Thames Rivers study unit, 1969-92. [Data from the U.S. Geological Survey's National Water Information System (NWIS) data base and the U.S. Environmental Protection Agency's Storage and Retrieval System (STORET) data base.]

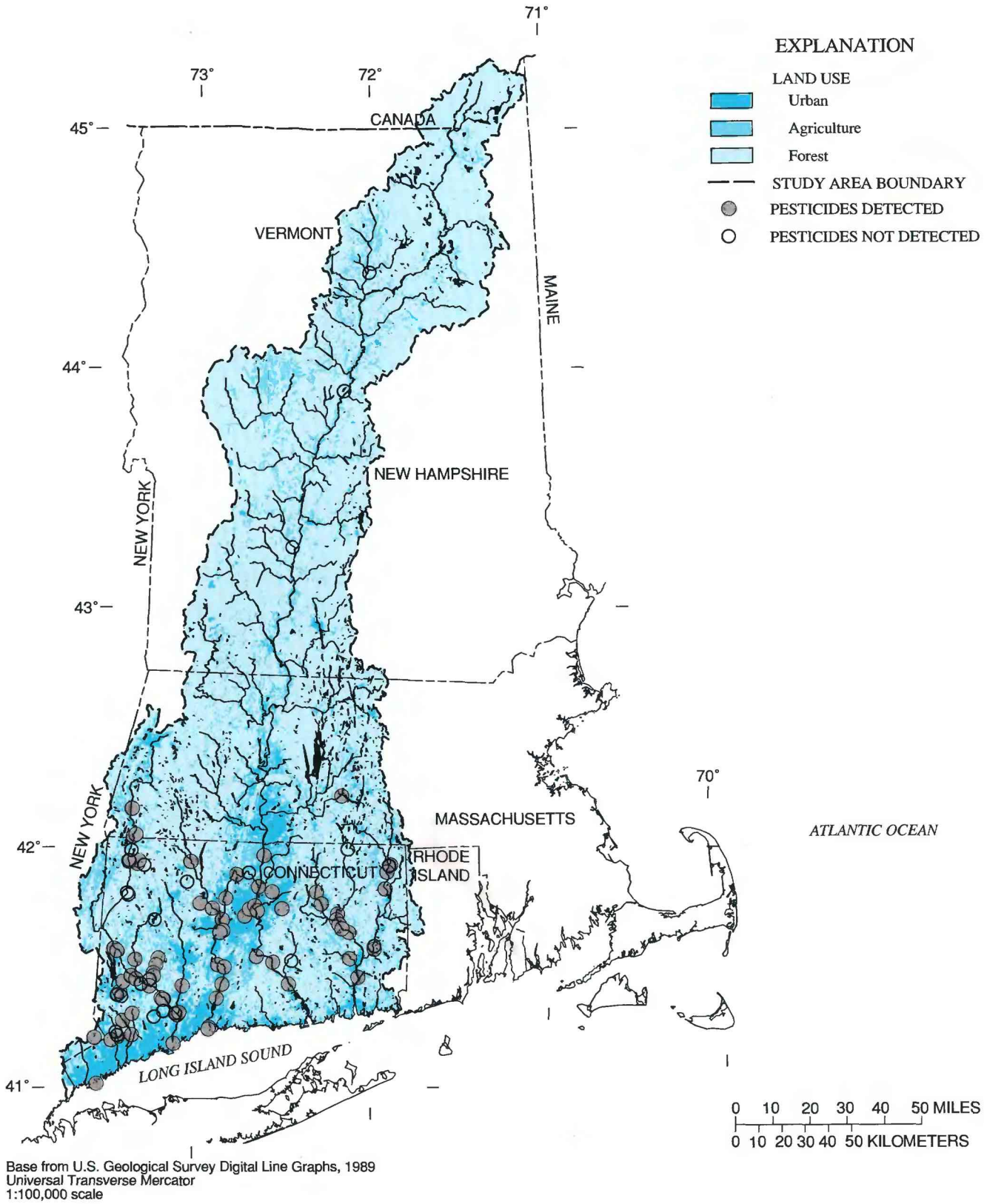


Figure 21. Areal distribution of pesticides in bed sediment in the Connecticut, Housatonic, and Thames River Basins study unit, 1969-92. [Data from the U.S. Geological Survey's National Water Information System (NWIS) data base and the U.S. Environmental Protection Agency's Storage and Retrieval System (STORET) data base.]

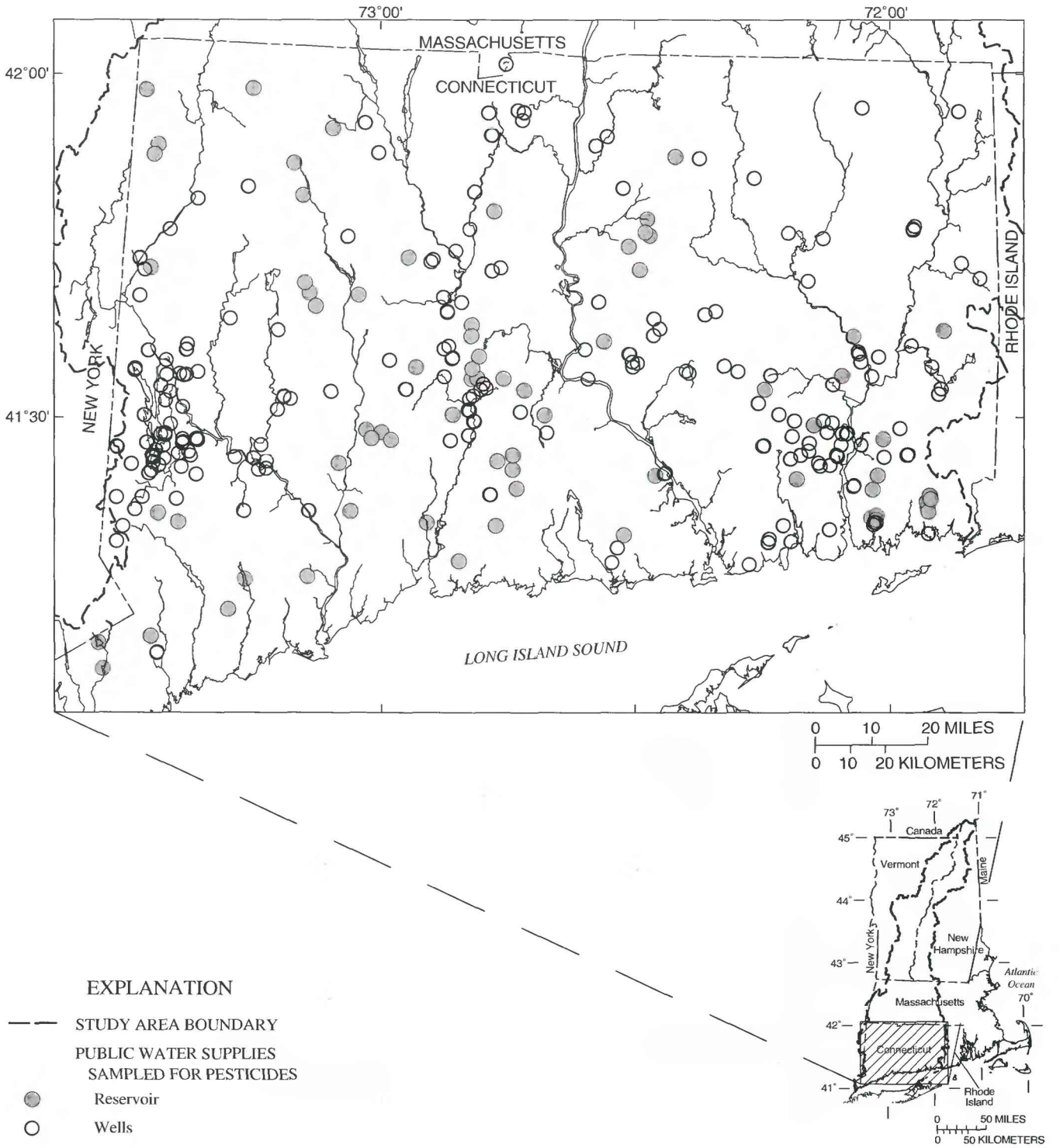


Figure 22. Areal distribution of pesticide sampling in public water supply from surface-water and ground-water sources, 1979-92. (Data from Connecticut Department of Health Services, written commun., 1992.)

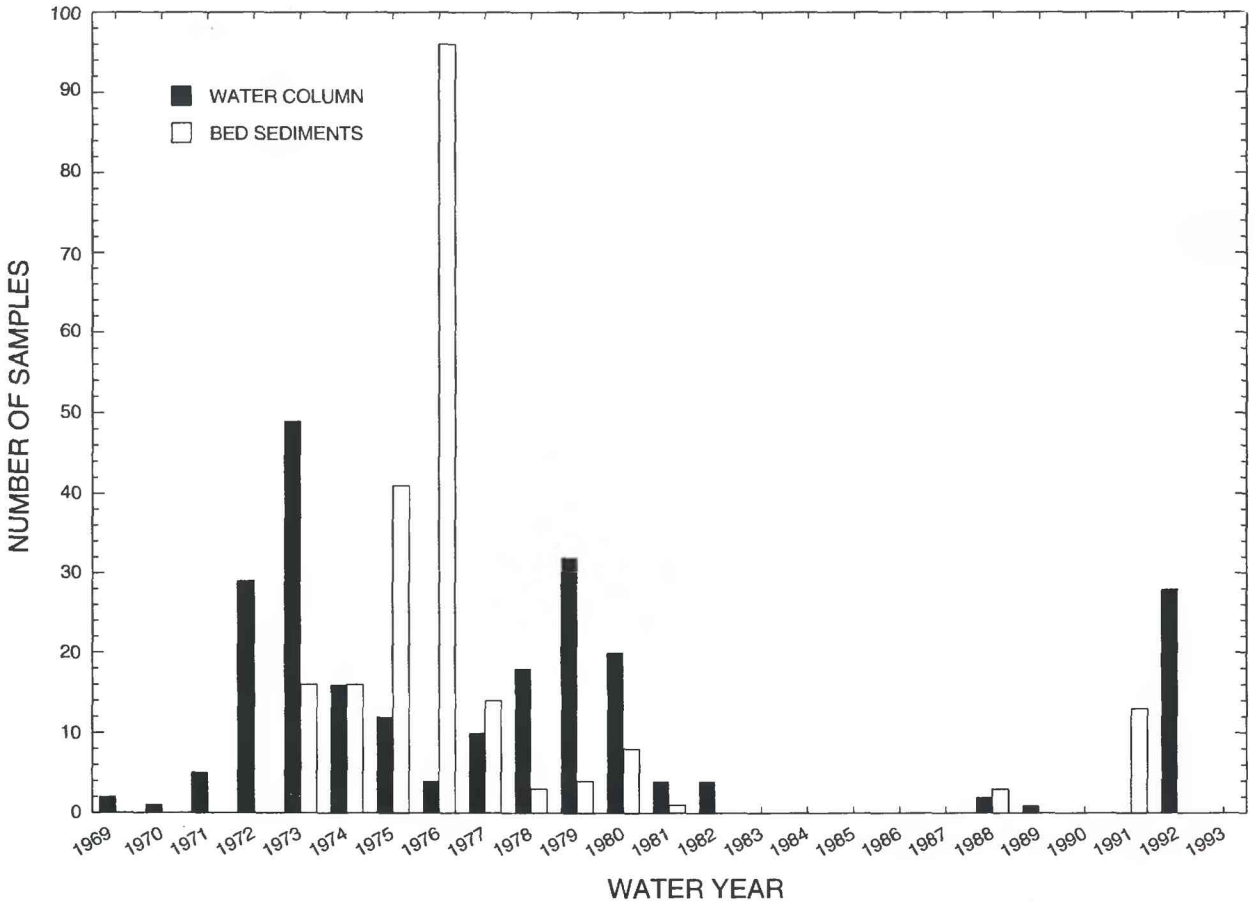


Figure 23. Number of surface-water column and surface-water bed-sediment samples analyzed for pesticides by year for water years 1969-92. (Data from the U.S. Geological Survey's National Water Information System data base.)

flows. Most samples were collected in summer when low-flow conditions prevailed. The small number of samples collected in September and the subsequent large number collected in October likely reflected differences in availability of funds just prior to and following the start of new fiscal years.

Ground Water

In terms of numbers of sites, samples, and types of analyses performed, more data are available for pesticides in ground water than in surface water. Several ground-water studies in Connecticut have targeted pesticide compounds (Grady and Weaver, 1988; 1989; Mullaney and others, 1991), or have at least included pesticides in more general water-quality sampling (Handman and Bingham, 1980; Grady and Handman, 1983; Bingham, 1991). Nevertheless, significant deficiencies in the spatial and temporal coverage limit a comprehensive assessment of pesticide occurrence and distribution in ground water in the study unit.

Spatial Distributions

All but 11 of the 225 wells sampled for at least one pesticide compound were in Connecticut (fig. 25), where they were distributed unevenly. Wells in north-central and southwestern Connecticut were clustered in four particular stratified-drift aquifers studied by Grady and Weaver (1988, 1989), whereas individual wells distributed throughout much of the State were installed, for the most part, for a reconnaissance of pesticide occurrence in ground water beneath agricultural areas and golf courses (Mullaney and others, 1991). An additional 21 wells sampled for pesticides in Connecticut were installed for other studies (Handman and Bingham, 1980, Grady and Handman, 1983;

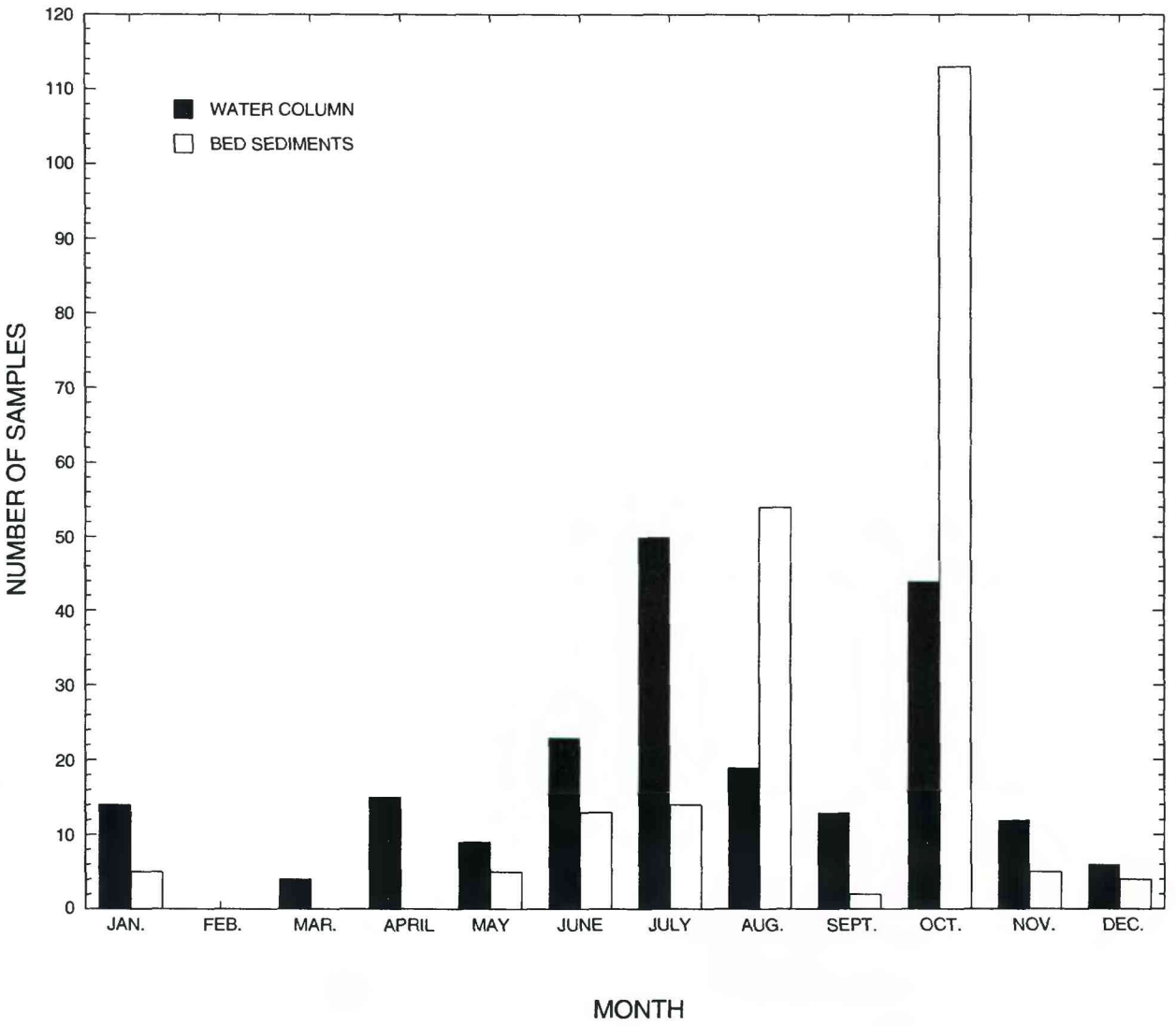


Figure 24. Number of surface-water column and surface-water bed-sediment samples analyzed for pesticides by month, 1969-92. (Data from the U.S. Geological Survey's National Water Information System data base.)

Bingham, 1991). Outside of Connecticut, only 11 wells (fig. 25) and 8 springs in New Hampshire were sampled for volatile organic compounds, including some of the compounds listed in table 24 (Moore and others, 1994).

Ground-water pesticide data are not available in NWIS for large parts of the study unit, including Massachusetts and Vermont, most of New Hampshire, parts of northwestern Connecticut, and the New York drainage basins. Previous pesticide studies or monitoring conducted by or for the States of Massachusetts (Massachusetts Interagency Pesticide Task Force, 1986; Stone & Webster Engineering Corporation, 1988), New Hampshire (New Hampshire Division of Public Health Services, 1986) and Vermont (Vermont Department of Agriculture, 1988) are the only known sources of pesticide information for those areas. In Connecticut, information on public-supply wells sampled for a limited number of pesticides is used to augment NWIS data. About 1,600 public-supply wells in Connecticut are subject to some pesticide monitoring and 237 were inventoried for this analysis (fig. 22).

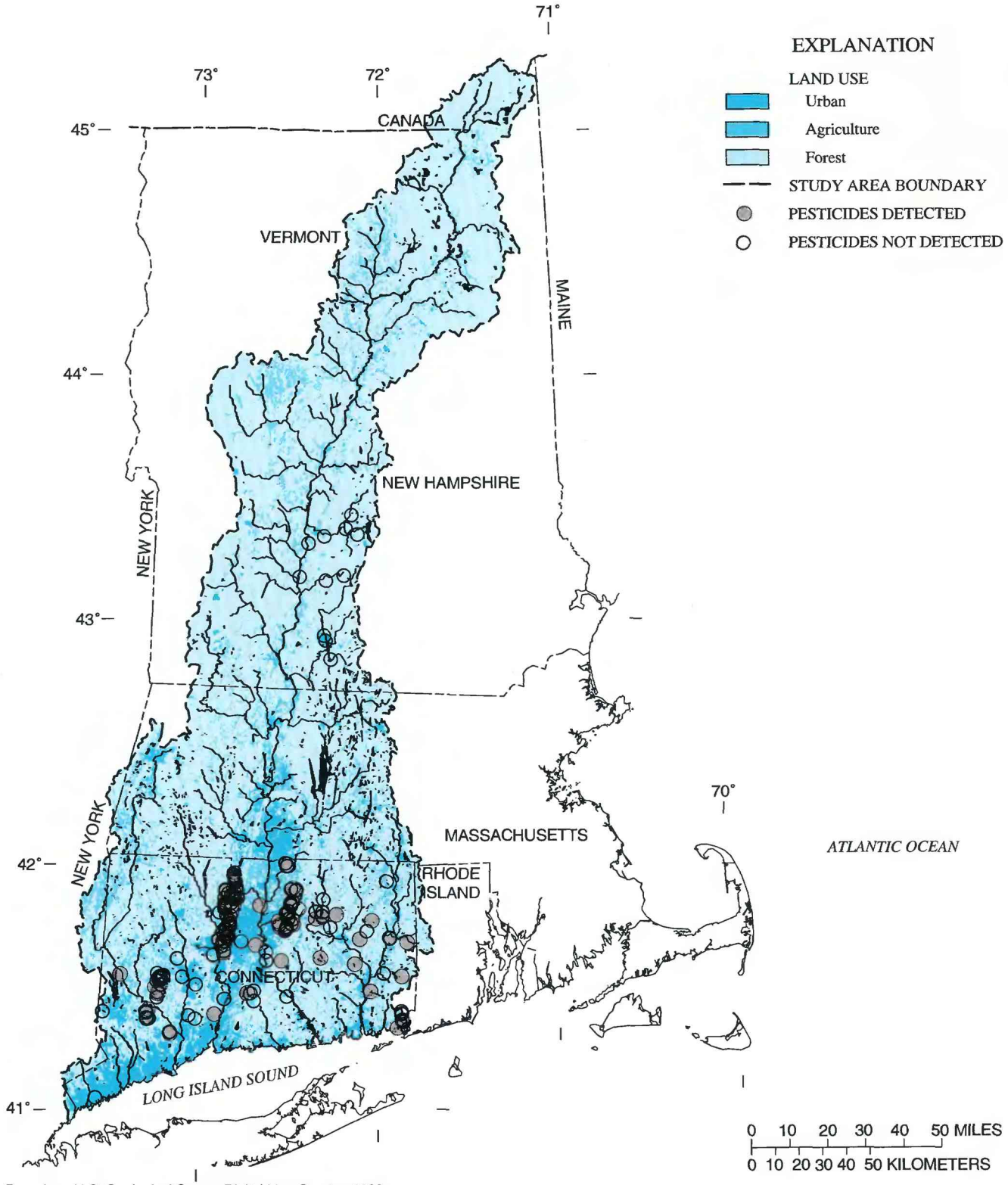


Figure 25. Areal distribution of wells sampled for pesticides in the Connecticut, Housatonic, and Thames River Basins study unit from the U.S. Geological Survey's National Water Information System data base, water years 1978-89.

Temporal Distributions

Essentially all ground-water pesticide data in NWIS were collected during a 5-year period (fig. 12) beginning in 1985, peaking in 1988 when 192 samples were collected, and ending in 1989. Pesticide samples were not collected by the USGS in the study unit after 1989.

Unlike the distribution of samples for nutrients or any other constituents, few pesticide samples were collected in the spring (fig. 13). Most samples were collected during the 3 summer months when essentially no recharge occurs. Also, samples were not collected during the 3 winter months.

Distribution by Well Type and Water Use

Essentially all the NWIS wells sampled for pesticides were USGS observation wells (fig. 14). Excluding wells coded as "test" wells and "unused" wells, which are likely to be observation wells, only one well with pesticide data was coded as a withdrawal well. This single withdrawal well was a public-supply well (fig. 15). All other NWIS pesticide samples came from observation and kindred wells where the water was not used.

Distribution by Aquifer Type

Clearly, most pesticide data (84 percent) were from wells completed in stratified-drift aquifers (fig. 16). About 11 percent are screened in till aquifers; hence 95 percent of the NWIS ground-water pesticide data represent shallow, unconsolidated, glacial aquifers. Only 12 NWIS wells sample the arkosic and crystalline bedrock aquifers, and the carbonate bedrock aquifer has not been sampled at all for pesticides.

Distribution by Well Depth

The bulk of the NWIS ground-water pesticide data was collected from shallow wells with even shallower water levels (figs. 17, 18). Sixty-five percent of the wells are less than 25 ft deep, and 95 percent do not extend to 50 ft below land surface. Of the 225 NWIS wells with pesticide data, 193 were installed for the purpose of sampling shallow ground water beneath agricultural or other land-use areas for pesticides and other potential contaminants. Depth to the water table is less than 10 ft in 66 percent of the wells, and less than 25 ft in more than 90 percent. The sample depth, or depth from the water table to the top of the open interval, is consequently shallow (fig. 19) for most of these sites.

SPATIAL AND TEMPORAL DISTRIBUTIONS OF NUTRIENTS, SEDIMENT, AND PESTICIDES

Information presented in the previous chapters provides the foundation for data presented here on concentrations of nutrients, suspended sediment, and pesticides found throughout the study unit. In particular, the data in this chapter show the relations between environmental factors and water quality.

Nutrients and Suspended Sediment

Surface Water

Although nitrogen and phosphorus are essential nutrients for aquatic plant growth, high concentrations of these constituents can adversely affect water quality through eutrophication, toxicity to aquatic life, and toxicity to warm-blooded animals that drink the water. The most readily absorbed, soluble forms of these nutrients are nitrate (NO_3^-), ammonium (NH_4^+), and orthophosphate (PO_4^{-3}). Other forms of these elements commonly reported individually or in combination include nitrite, total nitrogen, Kjeldahl nitrogen, dissolved phosphate, and total phosphorus. Hem (1985) provides an excellent, concise summary of the biogeochemistry of nitrogen and phosphorus in freshwater environments.

Nitrogen can exist in many forms in natural waters depending on source and degree of decomposition. Sources of nitrogen include atmospheric deposition, nitrate and ammonium agricultural fertilizers, organic wastes (sewage and industrial effluents, decaying plants, and animal wastes) and, to a lesser degree, weathering of certain types of igneous rocks and minerals in soil. Nitrogen availability rarely limits aquatic plant growth in freshwater.

In surface water, phosphorus generally occurs in sufficiently low concentrations to limit plant growth. Meybeck (1982) estimates that, globally, orthophosphate is naturally present in concentrations less than 0.010 mg/L. For this reason, phosphorus is recognized as the single nutrient most likely to limit primary production in fresh water. Sources of phosphorus in surface water include the breakdown and erosion of phosphorus-bearing minerals in soils, decaying vegetation, phosphate fertilizers and detergents, sewage effluent, and metabolic wastes from animals.

Suspended-sediment movement in streams is an important factor in the transport and fate of chemicals in the environment. Many surface-water-quality constituents, such as trace metals, organic compounds, indicator bacteria, and nutrients, especially phosphorus, are transported by suspended sediment (Rinella and others, 1992). Flowing streams almost always carry some sediment, but the highest concentrations and greatest loads are carried by storm runoff. Suspended-sediment sources include erosion due to natural geological conditions; erosion resulting from land-use practices, such as tilled agriculture, silviculture, and construction; and storm runoff from urbanized areas where storm sewers carry material washed in from roads. Because undeveloped areas in the study unit are heavily forested, erosion rates are low, and suspended-sediment concentrations in streams draining these areas are likely to be low. The highest concentrations of suspended sediment are likely to occur in urbanized and agricultural areas, where the land surface is disturbed and vegetative cover removed.

Previous Investigations

Water-quality studies in the Connecticut, Housatonic, and Thames River Basins began in the early 1900's. Southern basins, mainly in Connecticut, have received more attention than the northern basins. This dichotomy resulted, in part, from differences in population density and associated problems of deteriorating water quality. Previous studies were done by the USGS, primarily in Connecticut, and by State governments, which have published reports in all States, but most extensively in Massachusetts. Most water-quality studies have not examined suspended-sediment issues.

Historical Conditions

A series of water-resources investigations in Connecticut in the early 1960's included water-quality information (Pauszek, 1961; Pauszek and Edmonds, 1965; Randall and others, 1966; Ryder and others, 1970; Cervione and others, 1972; Hansen and others, 1973; Wilson and others, 1974; Mazzaferro and others, 1979; and Ryder and others, 1981). Although pollution received some notice at that time, most concerns about water quality in these studies focused on the effects of hardness, iron, and manganese on the use of the water for domestic and industrial uses. However, Randall and others (1966, p. 41), reporting on the water resources of the Quinebaug Basin, stated that the major rivers in the basin were primarily "used to transport industrial waste. Industrial wastes discharged into the streams of the Quinebaug River Basin in Massachusetts and Connecticut include cyanide, copper, nickel, chromium, grinding rouge (iron oxide), bleaches, dyes, soap, and acids and alkalis. Organic wastes, including sugar, starch, pulp fibers, blood, feathers, grease, and domestic sewage are also present." Most rivers in the basin were classified as unsuitable for bathing, public-water supply, some agricultural uses, and recreation.

By the mid-1960's, water-quality studies reported nitrate as a pollutant in surface and ground water. Many of the streams and rivers had nitrate concentrations exceeding 2 mg/L (as N), especially those in urbanized and heavily farmed areas (Mazzaferro and others, 1979; Ryder and others, 1981). Dissolved-oxygen concentrations commonly were less than 20 percent of saturation in many streams (Pauszek, 1961, Randall and others, 1966, Wilson, and others, 1974). In the mid-1960's to early 1970's, water-quality degradation was particularly noted in the Naugatuck, Still (near Danbury, Conn.), Quinnipiac, and Quinebaug River Basins, as well as the Connecticut River and some of its smaller tributaries. Water quality in less urbanized areas of the Housatonic, Thames, and Deerfield River Basins was not as noticeably affected by human activities.

Few early reports discussed phosphorus or suspended sediment. However, Cervione and others (1972) reported high phosphate concentrations in the upper Housatonic River Basin, where phosphate concentrations in some streams ranged from 0.23 mg/L at high flows to 0.38 mg/L at low flows. That study also reported turbidity being troublesome at high flows on the Still River due to construction and, on the Housatonic River near New Milford, Conn., due to extensive clearing and tilling of agricultural land upstream. Ryder and others (1981) reported differences in turbidity in streams in the Connecticut River drainage basin in northern Connecticut. In farmed and rural areas, streams running through soils developed over fine-grained glacial-lake deposits were highly turbid at times.

Although no reports exist on historical water-quality conditions in New Hampshire and Vermont, water quality in rivers in those areas probably was affected by unregulated industrial and municipal waste discharges. However, because of lower population densities and less extensive industrial areas in those states, the problems may have been less widespread.

Current Conditions

Since the adoption of the Clean Water Act of 1972, major improvements in water quality were achieved. Organic pollution in the Connecticut, Housatonic, and Thames River Basins was reduced and the percentage of major stream miles meeting water-quality standards increased.

States must now submit evaluations of water quality in their rivers and lakes to comply with the Clean Water Act of 1972, under the USEPA's 305-B program. These evaluations generally are qualitative descriptions containing little data and they are not specifically studies of nutrients and suspended sediment. Water-quality conditions are described in terms of "designated" and "attained" uses for stream reaches, lakes, ponds, and reservoirs. Although each State defines its own classification system of designated use, most have adopted the following system: Class A water bodies are suitable for public-drinking-water supply; Class B streams are suitable for fishing and bathing; and Class C streams are suitable for fishing, but not for bathing. States are primarily interested in whether or not a stream or river reach meets the classification criteria.

About 3 percent of the total stream miles in Vermont are classified as A (not including all waters recently classified A at altitudes of 2,500 ft above sea level); 91 percent of Vermont's streams are classified as B; and 6 percent of the streams are classified as C. An analysis of the rivers indicates that 60 percent of classified stream miles support designations, 23 percent are in danger of violating their designations, 14 percent partially support designations, and only 4 percent do not support classification designations. The most common pollutant concerns of those stream miles not achieving their classification goals in the Connecticut and Deerfield River Basins are nutrients, siltation, and organic enrichment. These water-quality problems are linked primarily to nonpoint sources and agriculture (Vermont Department of Environmental Conservation, 1990).

Toppin (1983) reported mean concentrations of total nitrogen ranging from 0.31 to 0.61 mg/L along a 26-mile reach of Vermont's Black River. Mean organic nitrogen ranged from 0.16 to 0.27 mg/L. Ammonia concentrations were low, with the highest mean concentration reported for any station of less than 0.08 mg/L, despite three sewage-treatment plants operating along the reach. Concentrations of mean total phosphate ranged from 0.014 to 0.112 mg/L (as P); the high mean concentration was associated with a sampling site just downstream from the largest sewage-treatment plant. Most stations had mean concentrations of total phosphate ranged from 0.02 to 0.03 mg/L for most stations.

Of the 14,544 river miles in New Hampshire, 99 percent are classified at least as good as class B (New Hampshire Department of Environmental Services, 1990). About 477 river miles of a total of 3,484 in the Connecticut River Basin have been evaluated and 58 river miles do not support their classification criteria; 200 river miles partly comply; 219 river miles fully meet the criteria, although all 477 of these river miles are in danger of violating their designated classifications. The major causes of nonattainment are heavy usage by 25 sewage-treatment plants and nonpoint-source contamination from failing septic systems, road-salt storage and use, and landfill leachate.

The major water-quality problems in Massachusetts are urban and suburban runoff, combined sewer outfalls, and contaminated sediments in the Housatonic and Connecticut Rivers and most of the Connecticut's major tributaries. About 10 percent of streams in the Connecticut, Housatonic, and Thames River Basins in Massachusetts are class A streams and the remaining 90 percent are class B streams (Massachusetts Department of Environmental Protection, 1990c). In 1989, 35 percent of the river miles in Massachusetts fully met their designated classification criteria. In addition, 29 percent partially met, and 36 percent did not meet their criteria. Water-quality problems caused by ammonia and suspended solids decreased substantially from 1979 to 1989 and violations resulting in nonattainment for nutrients decreased from 41 percent in 1979 to 24 percent of classified river miles in 1989 (Massachusetts Department of Environmental Protection, 1989).

Water-use classification goals for the State of Connecticut put all of the State's streams and rivers in the class A or B categories—suitable for fishing and bathing. As of 1990, 68 percent of Connecticut's rivers and streams fully complied with their water-quality standards. Of the remainder, 26 percent partially complied with their goals and the remaining 6 percent did not support most water uses (Connecticut Department of Environmental Protection, 1990). Common reasons for nonattainment of designated-use goals included pathogen indicators, low dissolved-oxygen concentrations accompanied by high organic enrichment, high concentrations of ammonia and metals, and the occurrence of PCBs. In Connecticut,

major water-quality concerns include combined sewer overflows, municipal and industrial discharges, landfill leachates, agricultural runoff, and contaminated stream sediments.

Trends in Water Quality

Trends in water quality were analyzed for 33 stream sites in Connecticut for periods of record in water years 1969-88 (E.C.T. Trench, USGS, written commun., 1993). The combined drainage area for this monitoring network encompassed most of the NAWQA study unit. Preliminary trend results from the Connecticut study indicated statewide decreases in some suspended constituents in surface water and increases in dissolved constituents. Turbidity and concentrations of total phosphorus and total organic carbon decreased and specific conductance and concentrations of dissolved magnesium and chloride increased.

The following trends in nutrients were detected for water years 1974-88 period: (1) increasing concentrations of total nitrogen and total organic nitrogen at most stations; (2) increasing concentrations of total nitrite-plus-nitrate at slightly less than one-half of the stations; (3) decreasing concentrations of total phosphorus and total organic carbon at most stations. The number of stations with increasing concentrations exceeded the number of stations with decreasing concentrations for five out of six nitrogen water-quality constituents analyzed for trends during water years 1981-88. However, the percentage of stations with increasing concentrations of nitrogen water-quality constituents was much smaller for water years 1981-88 than for water years 1974-88. The number of stations with decreasing concentrations of total ammonia nitrogen exceeded the number of stations with increasing concentrations during water years 1981-88.

Previous National Studies

Smith and others (1987) discussed national trends in river water quality for water years 1974-81. They found decreases in total phosphorus associated with improvements in advanced wastewater treatment following passage of the Clean Water Act of 1972. National trends showed that increased nitrogen fertilizer use and increased atmospheric deposition of nitrogen, two nonpoint sources, correlated with increases in total nitrogen in surface water.

A USEPA study (Omernik, 1977) of 928 streams with nonpoint sources only, sampled from 1972 to 1974, indicated a correlation between land use and nonpoint source nutrient concentrations. Nationally, nutrient concentrations were considerably higher in streams draining agricultural drainage basins than in streams draining forested drainage basins. This USEPA study also indicated that total phosphorus concentrations in the study unit did not exceed the 0.1 mg/L concentration limit recommended to inhibit excessive growth of aquatic plants in flowing water.

Spatial Distributions

Examination of surface-water-quality data in the NWIS data base indicates that median concentrations for most nitrogen and phosphorus constituents generally are higher in the southern part of the study unit than in the northern part of the study unit (table 25, at back of report; figs. 26 and 27). Median concentrations are high for all constituents at the Connecticut stations, and low at the Vermont and New Hampshire stations. These latitudinal variations in concentration correspond generally to the overall increasing population density and urban land-cover patterns southward in the study unit (figs. 2 and 3). However, most of the stations used in this analysis are in Connecticut where water-quality issues have received more attention; Massachusetts has the fewest stations.

Unlike the NWIS data, STORET nitrogen data-collection stations primarily are in Vermont and New Hampshire (table 26, at back of report; fig. 28). Like the NWIS data, STORET data indicate that median concentrations of total nitrate in surface water do not approach the USEPA's maximum contaminant level (MCL) of 10 mg/L as nitrogen (44 mg/L as nitrate). Stations with low concentrations are in the far northern part of the study unit. However, the geographic coverage of the STORET data is quite limited; large parts of the study unit are not represented.

Much of the analysis in this report is based on 18 major long-term NWIS water-quality stations and the characteristics of their respective basins (tables 9 and 27, at back of report). The Connecticut River at Thompsonville, Conn., drains the largest basin (9,660 mi^2), comprising more than one-half of the area of the study unit. The smallest basin studied, Burlington Brook near Burlington, Conn., is a primarily rural forested site, with some low-density

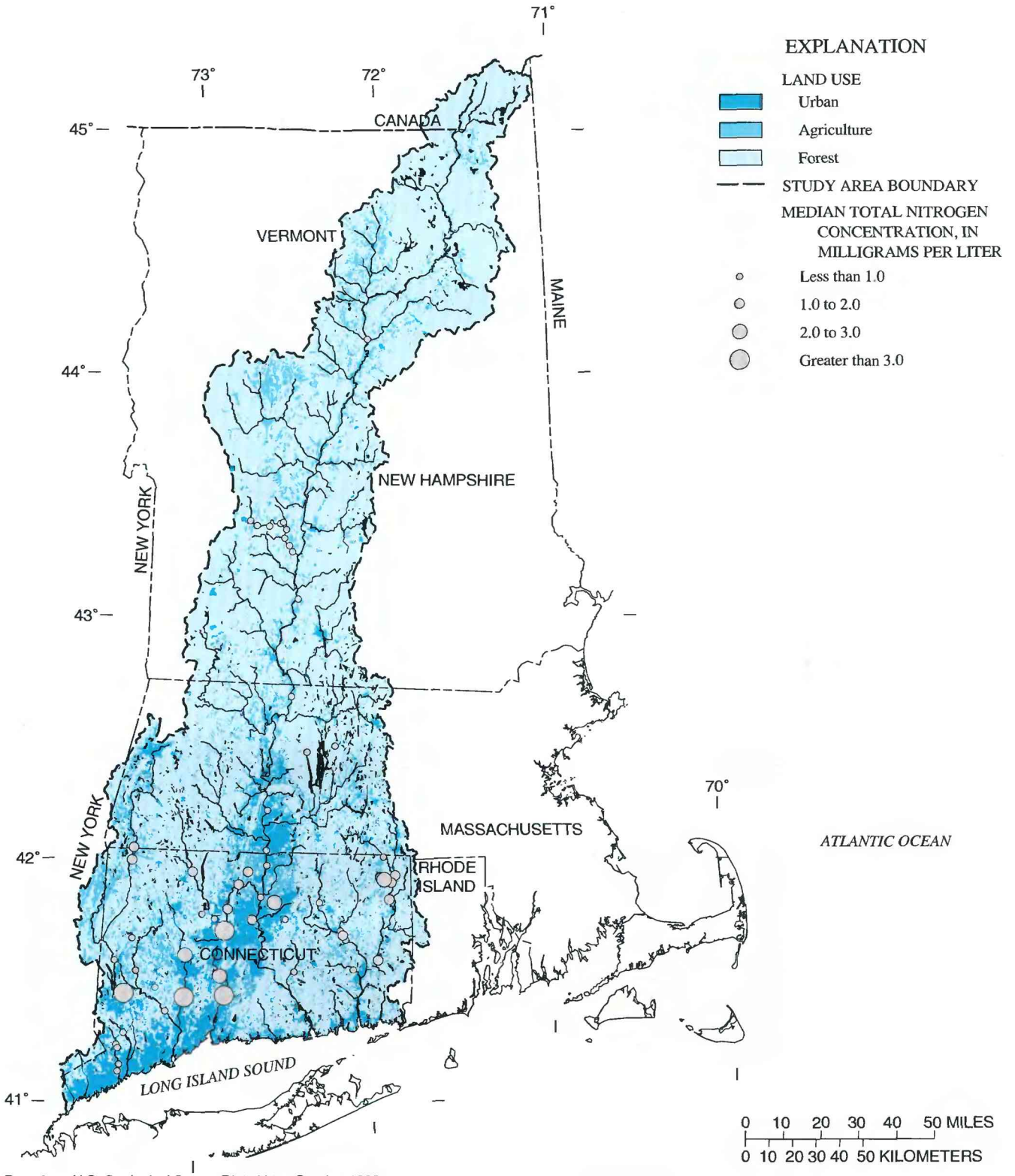


Figure 26. Areal distribution of median concentrations of total nitrogen at selected streamflow-gaging stations in the Connecticut, Housatonic, and Thames River Basins study unit from the U.S. Geological Survey's National Water Information System data base.

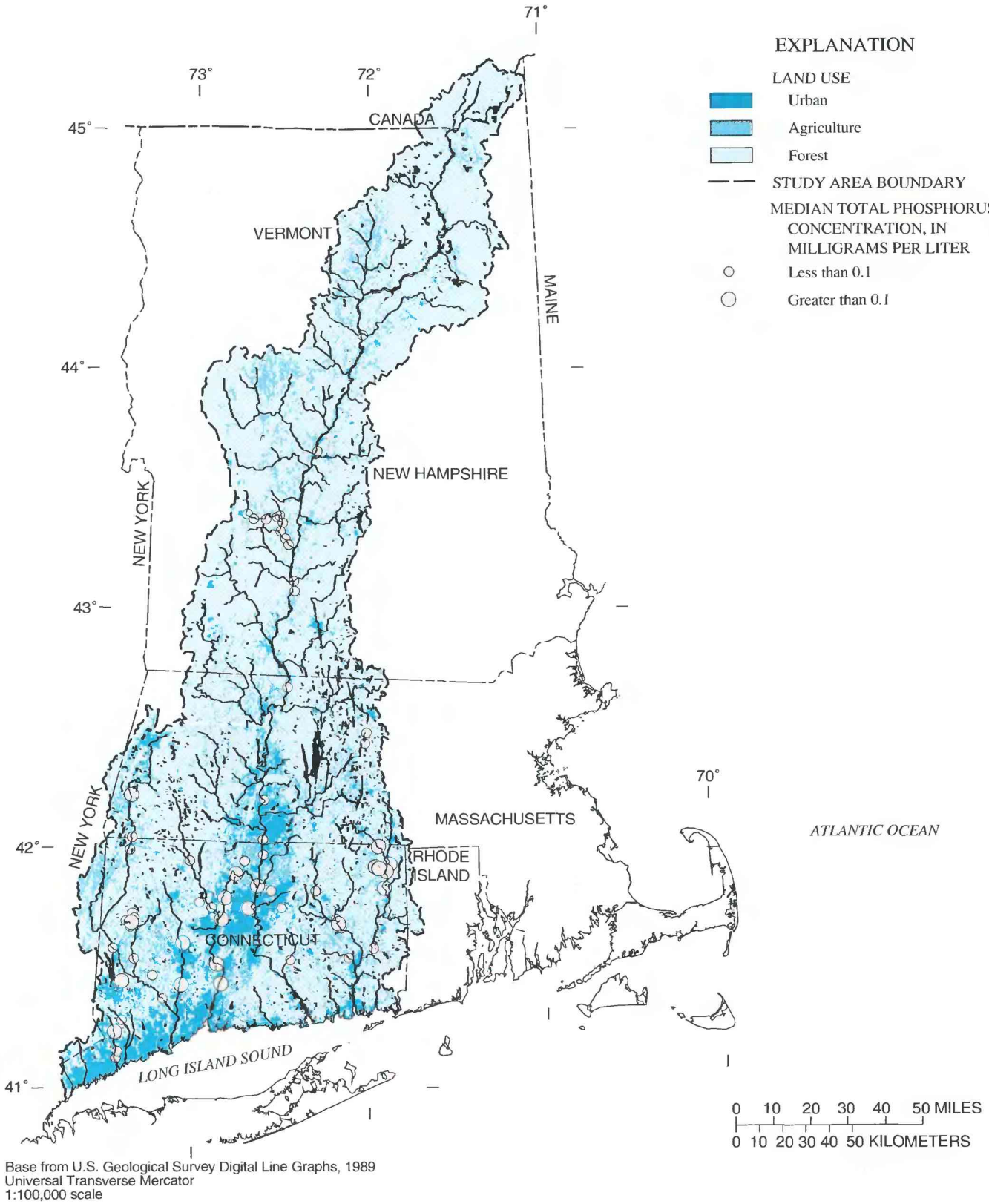


Figure 27. Areal distribution of median concentrations of total phosphorus at selected streamflow-gaging stations in the Connecticut, Housatonic, and Thames River Basins study unit from the U.S. Geological Survey's National Water Information System data base.

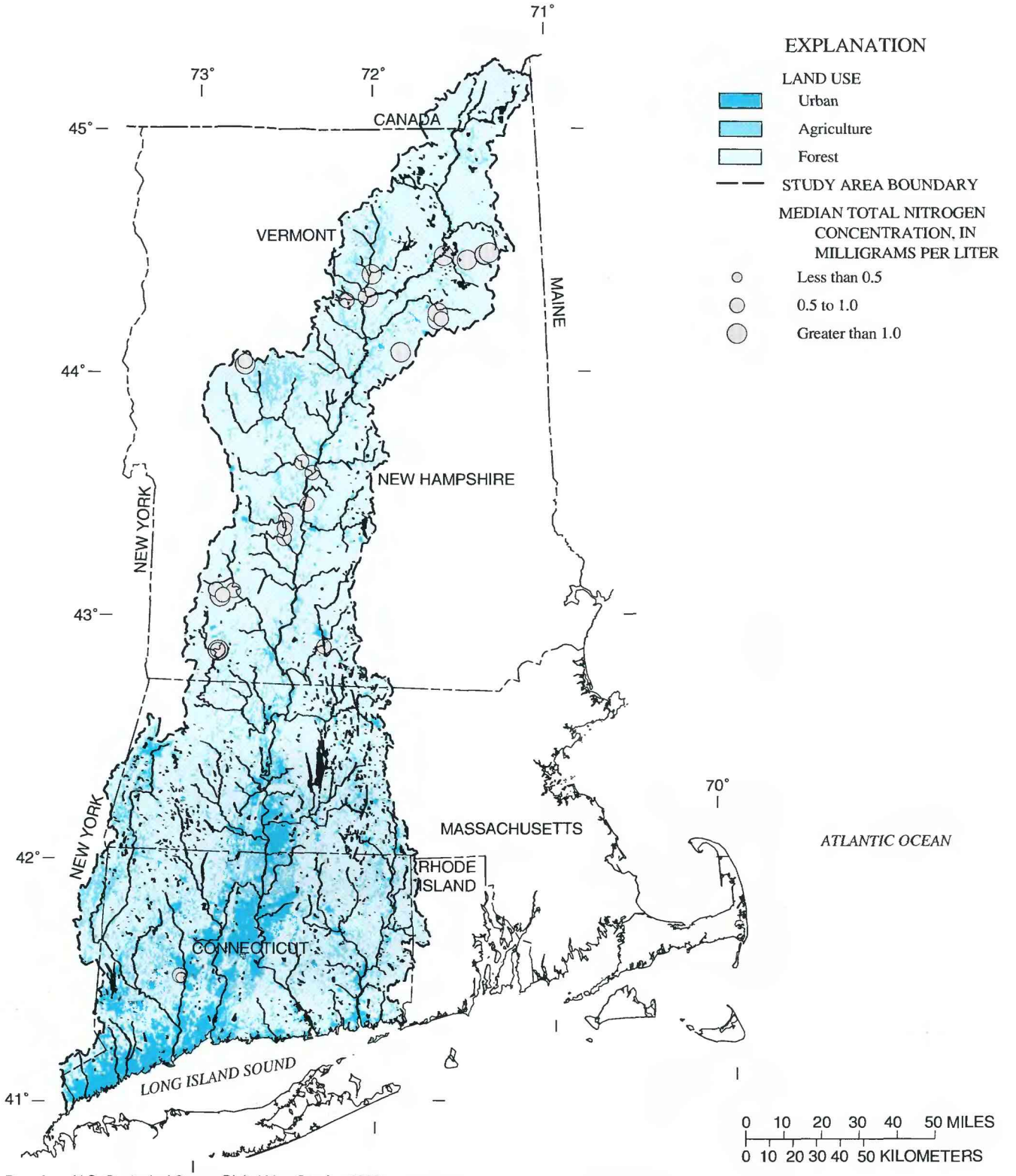


Figure 28. Areal distribution of median concentrations of total nitrogen at selected surface-water sites in the Connecticut, Housatonic, and Thames River Basins study unit from the U.S. Environmental Protection Agency's Storage and Retrieval System data base.

Table 9. Drainage areas, environmental settings, and upstream point sources for selected water-quality-sampling stations in the U.S. Geological Survey's National Water Information System in the Connecticut, Housatonic, and Thames Rivers study unit

[Map No.: See plate 1 for location of water-quality-sampling stations; see table 27 for summary of data. Major point source discharge types and number of upstream dischargers from National Pollution Discharge Elimination System (NPDES) 1990 database. Major point source types: A , wastewater treatment and food processing; C, paper and paperboard processing; E, manufacturing and industry. mi^2, square mile]

Map No.	Station No.	Station name	Drainage area (mi^2)	Environmental settings	Major point source types	Number of upstream dischargers
1	01119375	Willimantic River at Merrow, Conn.	94.0	Forested New England Upland	A	2
2	01122610	Shetucket River at South Windham, Conn.	408	Forested and Urban New England Upland	A,E	5
3	01124000	Quinebaug River at Quinebaug, Conn.	155	Forested and Urban New England Upland	A,E	9
4	01127000	Quinebaug River at Jewett City, Conn.	713	Forested, Agricultural, and Urban New England Upland	A,E	45
5	01154500	Connecticut River at North Walpole, N.H.	5,490	Forested and Agricultural New England Upland; Mountain Upland	A,C	56
6	01184000	Connecticut River at Thompsonville, Conn.	9,660	Forested, Agricultural, and Urban New England Upland; Urban and Agricultural Connecticut Valley Lowland; Mountain Upland	A,C,E	183
7	01186800	Still River at Riverton, Conn.	86.2	Forested New England Upland	E	10
8	01188000	Burlington Brook near Burlington, Conn.	4.10	Forested New England Upland	--	0
9	01188085	Farmington River at State Highway 4, at Unionville, Conn.	378	Forested, Agricultural, and Urban New England Upland	E	15
10	01189030	Pequabuck River at Farmington, Conn.	57.2	Urban New England Upland, Urban Connecticut Valley Lowland	A,E	22
11	01189995	Farmington River at Tariffville, Conn.	577	Forested, Agricultural, and Urban New England Upland; Forested, Agricultural, and Urban Connecticut Valley Lowland	A,E	39
12	01193500	Salmon River near East Hampton, Conn.	100	Forested New England Upland	E	1
13	01196500	Quinnipiac River at Wallingford, Conn.	115	Urban, Agricultural, and Forested Connecticut Valley Lowland	A,E	36
14	01200600	Housatonic River near New Milford, Conn.	1,022	Forested, Agricultural, and Urban Housatonic Valley Lowland; Forested New England Upland	A,C	27
15	01205500	Housatonic River at Stevenson, Conn.	1,544	Forested, Agricultural, and Urban Housatonic Valley Lowland; Forested New England Upland	E	63
16	01208500	Naugatuck River at Beacon Falls, Conn.	260	Forested, Agricultural, and Urban New England Upland	A,E	68
17	01208990	Saugatuck River near Redding, Conn.	21.0	Forested Coastal Lowland	--	0
18	01209710	Norwalk River at Winnipauk, Conn.	33.0	Urban Coastal Lowland	E	4

residential development. Many of the basins have significant numbers of upstream point-source discharges. Of these 18 basins, the Quinnipiac River at Wallingford, Conn., and the Naugatuck River at Beacon Falls, Conn., have the most dischargers per unit area in their basins; Burlington Brook near Burlington, Conn., the Salmon River near East Hampton, Conn., and the Saugatuck River near Redding, Conn., have the fewest. Data presented for each basin include the major environmental setting, or settings if the basin encompasses more than one setting or has mixed land uses. For example, the drainage basin of the Shetucket River at South Windham, Conn., consists primarily of forested areas, with several smaller urban areas in the New England Uplands. Detailed land-use data (Anderson Level II; Anderson and others, 1976) in each basin are from the mid-1970's (table 27).

Total Nitrogen

Analyses for total nitrogen include organic and inorganic forms of dissolved and suspended nitrogen including nitrite, nitrate, and ammonia species. Median total nitrogen concentrations for 60 NWIS surface-water sampling stations (table 25) ranged from 0.31 (Black River at Weathersfield, Vt.) to 4.2 mg/L as N (Pequabuck River at Farmington, Conn.). Median concentrations of total nitrogen for 34 STORET water-quality sampling stations (table 26) ranged from 0.19 (West Branch Upper Ammonoosuc River at Berlin, N.H.) to 1.30 mg/L as N (confluence of Hop and Shattuck Brooks at Waterbury, Conn.)

Ammonia

Ammonia is generated as the primary nitrogenous end product of decomposition of organic matter, and to a lesser degree, as an excretory product of aquatic animals. In most fresh water of the study unit, ammonia occurs primarily as the ammonium ion (NH_4^+), a nutrient form readily absorbed by aquatic plants. Under conditions of high pH, however, the change in chemical equilibrium may result in a substantial proportion of ammonia existing as un-ionized NH_3, a form toxic to freshwater aquatic life, especially to fish. At a pH of 9.3, and a temperature of 25 °C, about one-half of the ammonia is un-ionized NH_3 (D.R. Helsel, U.S. Geological Survey, written commun., 1993).

Minimum ammonia concentrations that are toxic to invertebrates and fish vary among animal species; for invertebrates, these minimum concentrations range from 0.12 to 22.8 mg/L as NH_3 (or 0.10 to 18.7 mg/L as N) and for fish, the range is 0.083 to 4.60 mg/L as NH_3 (or 0.068 to 3.77 mg/L as N) (U.S. Environmental Protection Agency, 1986). The USEPA's criteria for NH_3 for protection of aquatic life vary with temperature and pH, according to different formulae based on duration of exposure for salmonids or other fish species.

Ammonia concentrations in this report cannot be directly related to USEPA thresholds of toxicity to aquatic life because of reporting conventions for this nitrogen species. According to established nomenclature conventions in the U.S. Geological Survey and other State and Federal agencies, the sum of the ammonium ion [NH_4^+] concentration and the solvated (un-ionized) ammonia [NH_3 (aq)] concentration determined for a natural water sample is reported as 'ammonia' (D.A. Rickert, U.S. Geological Survey, written commun., 1993). Ammonia [NH_3 (aq)] is the constituent of concern in terms of toxicity to aquatic life. However, the proportion of ammonium ions [NH_4^+] in reported ammonia concentrations is likely to be high under most conditions in the study unit. Equilibrium calculations for individual samples, incorporating an equilibrium constant and information on temperature and pH of the sample at the time of collection, would be necessary to estimate concentrations of [NH_3 (aq)] and [NH_4^+] from samples reported as 'ammonia'.

Of 48 NWIS stations reporting 15 or more dissolved ammonia samples (table 25), median concentrations range from less than 0.01 mg/L as N (West Branch Swift River near Shutesbury, Mass.) to 1.9 mg/L as N (Pequabuck River at Farmington, Conn.). Only 11 of the stations are north of Connecticut.

Total ammonia concentrations are greater than the detection limit for 91 sampling stations in STORET. Concentrations at those stations range from 0.004 mg/L (as N) at South Peacham Brook at South Peacham,Vt., to 0.55 mg/L (as N) at Otter River at Baldwinsville, Mass. Ammonia concentrations are high at NWIS and STORET stations in urbanized settings with point-source discharges.

Nitrate or Nitrite Plus Nitrate

Ammonium ions in aerobic soil or water undergo bacterial oxidation to nitrite and then nitrate. In addition, certain prokaryotes can fix atmospheric nitrogen

(N_2) and convert it into nitrate. Conversely, in anaerobic soils and water, through the process of denitrification, bacteria may reduce nitrite and nitrate to nitrogen gas (N_2).

Examination of data in the NWIS data base indicates that at most locations in the study unit, concentrations of nitrite are negligible most of the time. Thus, little or no difference exists in the data for total nitrate and the data for total nitrite plus nitrate. For that reason, these constituents are discussed together here, and one can assume that general statements about nitrite plus nitrate concentrations also are valid for nitrate at most times and locations in the study unit. However, nitrite occasionally exceeds negligible concentrations in many streams that receive effluent from sewage-treatment plants. The streams with the highest concentrations of nitrite and the greatest frequency of high concentrations are all in Connecticut—the Pequabuck River, the downstream end of the Farmington River, the Quinnipiac River, the Still River (at Brookfield Center), and the Naugatuck River. Other unmonitored stream reaches in the study unit also may have high nitrite concentrations under some conditions.

High concentrations of nitrate, nitrite, or nitrite plus nitrate in a water supply may harm warm-blooded animals by causing methemoglobinemia, a condition in which red blood cells are unable to carry oxygen. The greatest human health risk is to infants less than 1 year of age. The MCL for nitrate (as N) in domestic drinking-water supplies is 10 mg/L (U.S. Environmental Protection Agency, 1986). Data for the 228 NWIS and STORET stations listed in table 23 were examined to determine if any stations had nitrite plus nitrate (for NWIS) or nitrate (for STORET) concentrations that equaled or exceeded the USEPA's drinking water MCL. One NWIS station had two samples that equaled the MCL, and eight STORET stations each had one sample that exceeded the MCL. All these samples were collected during water years 1974-76.

High concentrations of nitrite plus nitrate in the presence of other nutrients also help to stimulate undesirable plant growth in streams. However, a nitrite plus nitrate criterion for controlling these growths has not been established.

Median concentration of nitrite plus nitrate in United States rivers was 0.36 mg/L from 1974 to 1981 (Smith and others, 1987). Median concentrations of total nitrite plus nitrate in water from 61 NWIS water-quality stations (table 25) ranged from 0.01 at the East Branch Fever Brook near Petersham, Mass., to 1.9 mg/L at the Podunk River at South Windsor, Conn. Median concentrations of total nitrate from 56 STORET water-quality stations (table 26) ranged from 0.07 at the Quinebaug River at Holland, Mass., to 0.89 mg/L at the Miller River below Birch Hill Dam, Royalston, Mass.

Nitrate data from the ARMP served to augment the otherwise sparse stream water-quality data for Massachusetts. Nitrate (as N) concentrations were calculated for the 185 ARMP sites (fig. 9) for the monitoring period, 1984-93, and 62 percent were less than the detection limit of 0.2 mg/L (table 28, at back of report). About 79 percent of the median concentrations were less than 0.2 mg/L and 17 percent of the median concentrations ranged from 0.2 to 1.0 mg/L. Only 3.2 percent were greater than 1.0 mg/L. The maximum nitrate concentration for any site was 9.63 mg/L as N at Wellington Brook at Oxford, Mass., slightly less than the USEPA's MCL of 10 mg/L for drinking water.

Total Phosphorus

Total phosphorus concentrations include the organic and inorganic forms of dissolved and suspended phosphorus. The dynamic physical, biological, and geochemical factors affecting phosphorus concentrations in rivers may result in analytical concentrations of dissolved species varying from concentrations less than detection limits to concentrations equal to the total analytical concentration (Livingstone, 1963).

Phosphorus concentrations determined in the study unit from 1972 to present resemble concentrations measured in many rivers in the United States from 1974 to 1981 (Smith and others, 1987). Median concentrations of total phosphorus in United States rivers was 0.13 mg/L (as P). Median concentrations of total phosphorus for 67 NWIS streamflow-gaging stations (table 25) ranged from 0.01 (9 stations) to 0.80 mg/L (as P) for the Pequabuck River at Farmington, Conn., which has four upstream sewage-treatment plants. Median concentrations of total phosphorus for the 94 STORET streamflow-gaging stations (table 29, at back of report) ranged from less than 0.003 mg/L (as P) (Hancock Branch tributary to White River at Hancock, Vt., and Clark Brook at Granville, Vt.) to 0.365 mg/L (as P) (Otter River at Baldwinsville, Mass.)

A comparison of figures 26 and 27 shows that, for NWIS data, areas of the study unit with high median concentrations of total nitrogen also tend to have median concentrations of total phosphorus greater than 0.1 mg/L (as P). Most of the stream reaches with high median concentrations of total phosphorus (fig. 27) are receiving water for major municipal or industrial discharges.

Dissolved Orthophosphate

Median concentrations of dissolved orthophosphate for 23 NWIS streamflow-gaging stations (table 25) ranged from 0.001 mg/L (as P) (Schroeder Brook at South Marlborough, Conn.) to 0.28 mg/L (as P) (Naugatuck River at Beacon Falls, Conn.) Median concentrations of dissolved orthophosphate concentrations for 33 STORET streamflow-gaging stations (table 29) ranged from 0.004 (as P) (Connecticut River at Cornish, N.H.) to less than 0.30 mg/L (as P) (six stations). The upper median concentration (<0.3 mg/L) was a censored value reflecting multiple detection limits used by different laboratories.

In addition to contributing to eutrophic conditions, orthophosphate (as P) concentrations greater than 0.1 mg/L may interfere with coagulation during the removal of fine-grained sediment at water-treatment plants (U.S. Environmental Protection Agency, 1976). Median concentrations of dissolved orthophosphate greater than 0.1 mg/L (as P) were reported in water from three NWIS streamflow-gaging stations: a tributary to Peake Brook at Harrisville, Conn. (downstream from a dairy barn), the Quinnipiac River at Wallingford, Conn. (three upstream sewage-treatment plants), and the Naugatuck River at Beacon Falls, Conn. (downstream from the city of Waterbury and three sewage-treatment plants). Median concentrations of dissolved orthophosphate greater than or equal to 0.1 mg/L (as P) also were reported from three STORET stations—the Quinebaug River at Dudley, Mass., at Thompson, Conn., and at Fabyan, Conn.

Eutrophication

Examination of recent data from the NWIS and STORET data bases (tables 25, 29; pl. 1; figs. 27, 29) indicates several locations where median concentrations of total phosphorus exceed 0.1 mg/L. These data indicate the potential for eutrophication in the following Connecticut River subbasins—Ashuelot, Farmington, Millers, Otter, Park, and Pequabuck Rivers; in the Naugatuck and Still Rivers, which drain into the Housatonic River; and in the following Thames River subbasins: French, Quinebaug, and Shetucket Rivers; and in the coastal Quinnipiac River (pl. 1). Most of these streams receive substantial point discharges and some receive urban nonpoint runoff.

Suspended Sediment

From a purely physical standpoint, heavy suspended-sediment loads can have detrimental effects on aquatic stream life. Sediment, on settling out, can bury rocky streambeds, obliterating macroinvertebrate and fish spawning habitats. Large concentrations of suspended material create turbidity that may inhibit photosynthesis. Suspended sediment with a high organic content can create a significant oxygen demand in the water column, thereby depleting dissolved-oxygen concentrations.

The predominance of forest cover throughout the drainage basins, the relatively insoluble, crystalline bedrock, and the widespread deposits of compact, erosion-resistant glacial till tend to limit sediment erosion in the study unit. Sediment erosion mainly occurs where small tributary streams locally incise the terraces along the study unit's three main rivers and their major tributaries. These terraces are formed from stratified gravel, sand, silt, and clay deposited in proglacial lakes and streams. Sediment erosion also can occur in lowland areas where intensive agricultural activity is undertaken with little or no erosion control (Fay and Downer, 1971). Erosion of sediment also is likely in urban or developing areas where roads or buildings are being constructed.

Most rivers in New England have been dammed to provide power and to control floods. The study unit rivers are no exceptions. Much of the sediment load may settle in the impoundments behind these dams. Because significant fractions of the total phosphorus load may be associated with particulate material, the controlled releases from dams tend to diminish or attenuate natural transport of nutrients and suspended sediment and to alter natural patterns of nutrient loads. Sediments and associated nutrients trapped behind the dams and reservoirs may not leave the basin and, therefore, would not be reflected in samples collected downstream. However, the biogeochemical processes of nitrification and denitrification, for example, may mobilize some portion of the nutrients.

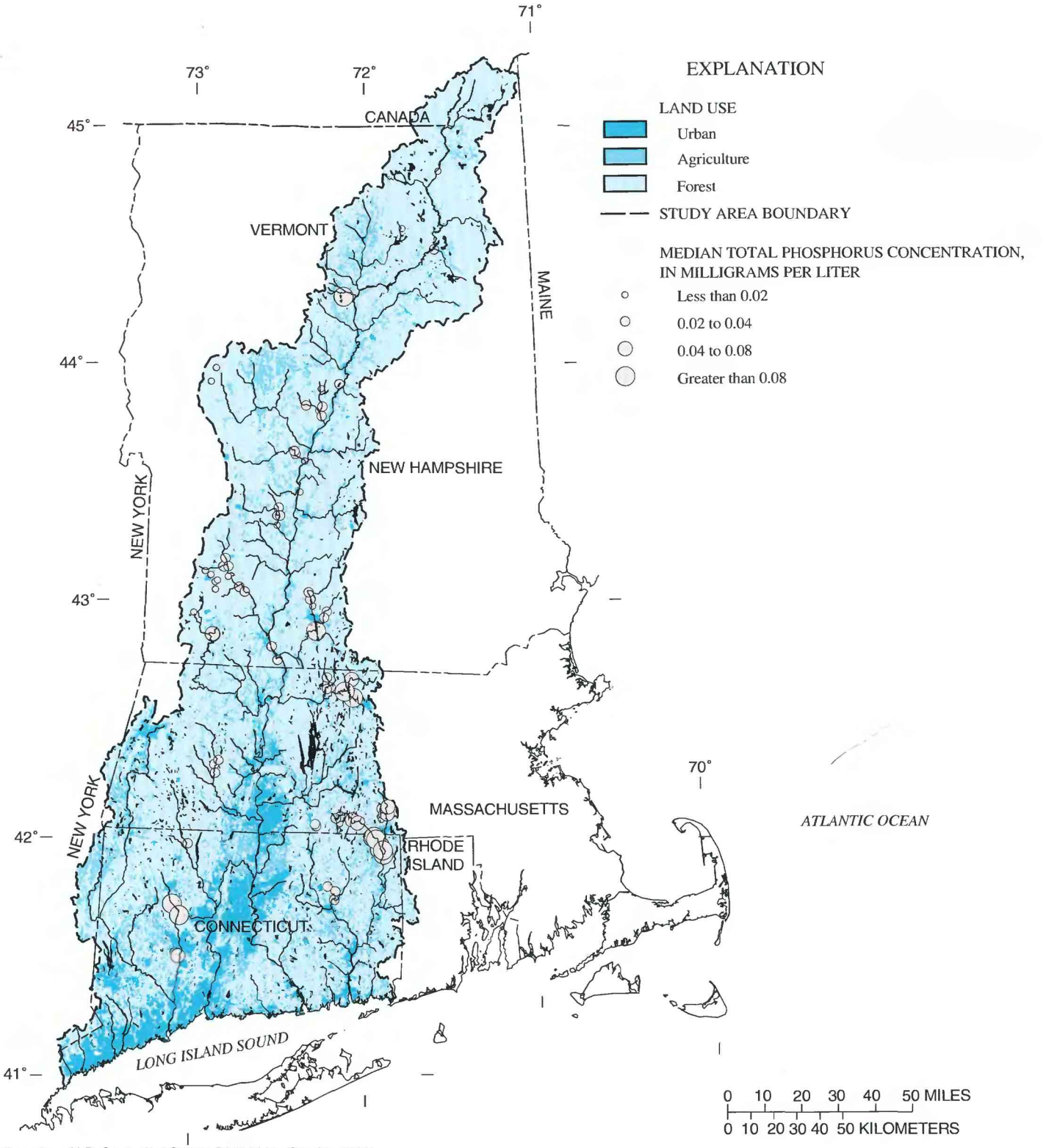


Figure 29. Areal distribution of median concentrations of total phosphorus at selected surface-water sites in the Connecticut, Housatonic, and Thames River Basins study unit from the U.S. Environmental Protection Agency's Storage and Retrieval System data base.

Generally speaking, excessive erosion and subsequent suspended-sediment transport and deposition in streambeds and lake or reservoir bottoms are not the problems in the study unit that they are elsewhere in the Nation (Briggs and Ficke, 1977). Only two sites, a tributary to Peake Brook at Harrisville, Conn., and a tributary to Mill Brook at South Woodstock, Conn. (table 25) had median concentrations of suspended sediment that exceeded the median concentration of 66.8 mg/L for many rivers in the United States for water years 1974-81 (Smith and others, 1987). In addition to geologic and climatic factors, two other reasons may account for the low median concentrations of suspended sediment reported: (1) samples from the 30 NWIS surface-water sites with suspended-sediment data (table 25, fig. 30) may have been collected during normal to low-flow periods when suspended-sediment concentrations are low; and (2) many of the sampling sites are affected by the numerous dams and reservoirs located throughout the study unit, especially along the Connecticut, Housatonic, and Thames Rivers.

Comparisons Among Environmental Settings

Most of the NWIS stations with the highest median concentrations of nitrogen and phosphorus constituents (table 25) are in environmental settings characterized by a large percentage of urbanized area, including major point discharges. The drainage basins of some of these stations also contain a substantial amount of agricultural land.

More detailed examination of nutrient data from eight representative stations in the study unit (fig. 31) illustrates how nutrient concentrations in urban environmental settings differ from concentrations in forested basins or integrator basins. Median concentrations for all nutrient water-quality constituents at the two urban stations—the Quinnipiac River at Wallingford, Conn., and the Naugatuck River at Beacon Falls, Conn.—substantially exceed medians for the other stations. In addition, median nutrient concentrations for these two stations exceed median concentrations for selected urban stations (fig. 32) examined in a nationwide survey (D.A. Helsel, U.S. Geological Survey, written commun., 1992).

The Connecticut River at North Walpole, N.H., which drains the northern part of the study unit, has the largest percentage of forested land and the smallest percentage of urbanized land in the basins associated with the stations shown in figure 31. Among the four integrator basins evaluated for nutrients and the five integrator basins evaluated for suspended sediment, this station consistently has the lowest median concentrations.

The lowest median concentrations and smallest interquartile ranges for nutrients are consistently seen in the data for the Connecticut River at North Walpole, N.H., the Salmon River near East Hampton, Conn., and the Saugatuck River near Redding, Conn. (fig. 31). These three drainage basins include the following environmental settings (fig. 4)—Mountain Upland, New England Upland, and Coastal Lowland. Land-use comparisons alone (table 27) are inadequate to explain the differences between these three basins and the five basins characterized by high nutrient concentrations. For example, the Saugatuck River near Redding, Conn., which has low nutrient concentrations, and the Naugatuck River at Beacon Falls, Conn., which has high nutrient concentrations, have about 22 percent of their drainage basins in urbanized land uses. Hypothetically, the important factors in explaining major differences in nutrient concentrations may be the type and intensity of urban land uses, in addition to the overall percentage of urbanized land; the number and size of point discharges, or absence of point discharges; and the distribution of urbanized land and point discharges in the drainage basin, relative to the monitoring location.

Seven of the eight stations for which nutrient concentrations are shown in figure 31 have greater than 10 percent of their drainage basins in agricultural land (table 27). Possible effects of agricultural fertilizer applications are not discernible at this level of analysis. Four of these seven stations monitor large integrator basins where high discharges may dilute local agricultural nutrient sources, and two of the stations monitor smaller basins where the effects of urban nutrient sources may mask agricultural effects.

The shortage of long-term water-quality stations in primarily agricultural basins makes it difficult to assess the effects of farming practices on water quality in any of the environmental settings in the study unit or in comparison to agricultural land nationally. The mixture of urban and agricultural land use in many individual basins in the study unit further exacerbates this analysis problem.

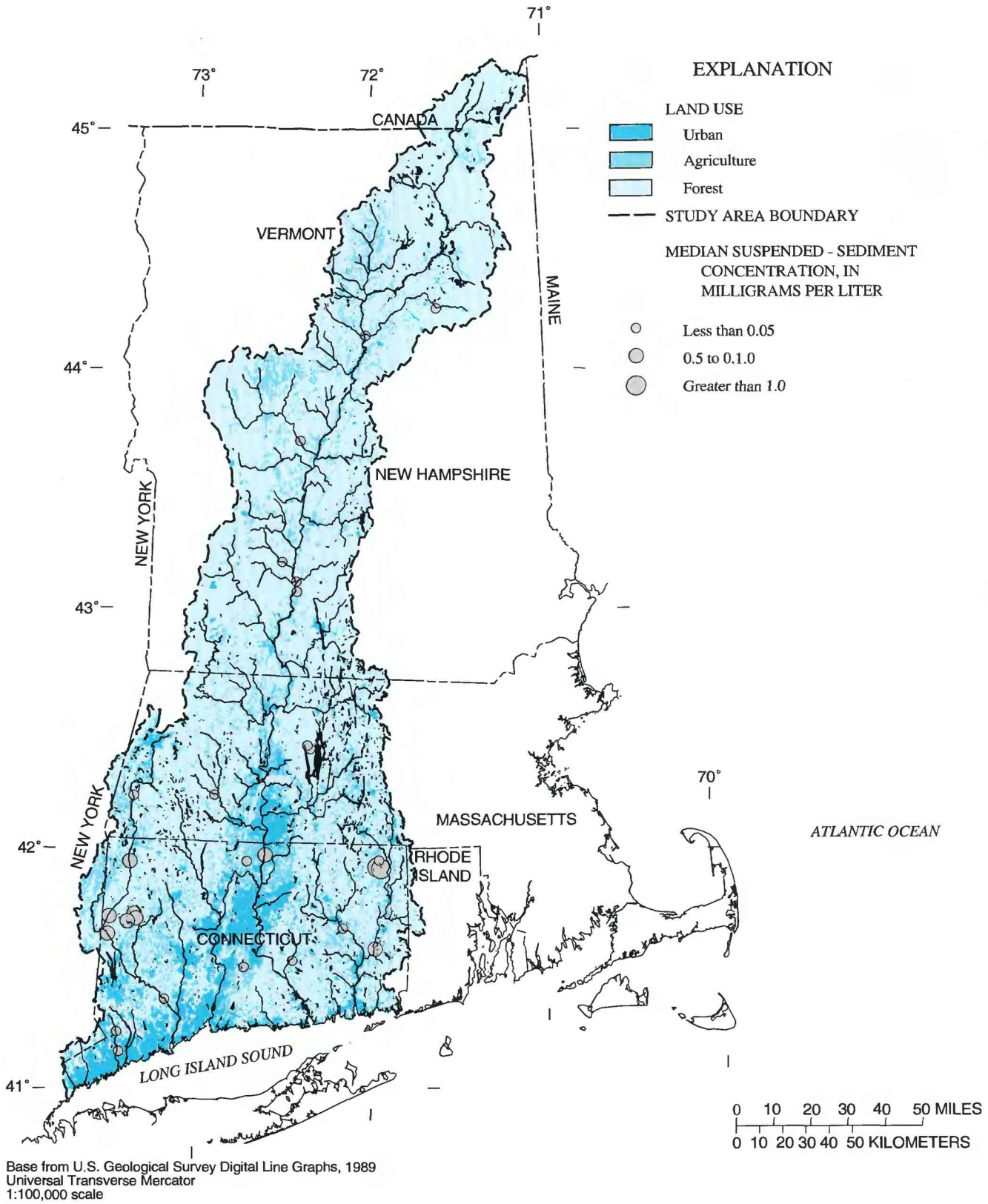


Figure 30. Areal distribution of median concentrations of suspended sediment at selected streamflow-gaging stations in the Connecticut, Housatonic, and Thames River Basins study unit from the U.S. Geological Survey's National Water Information System data base.

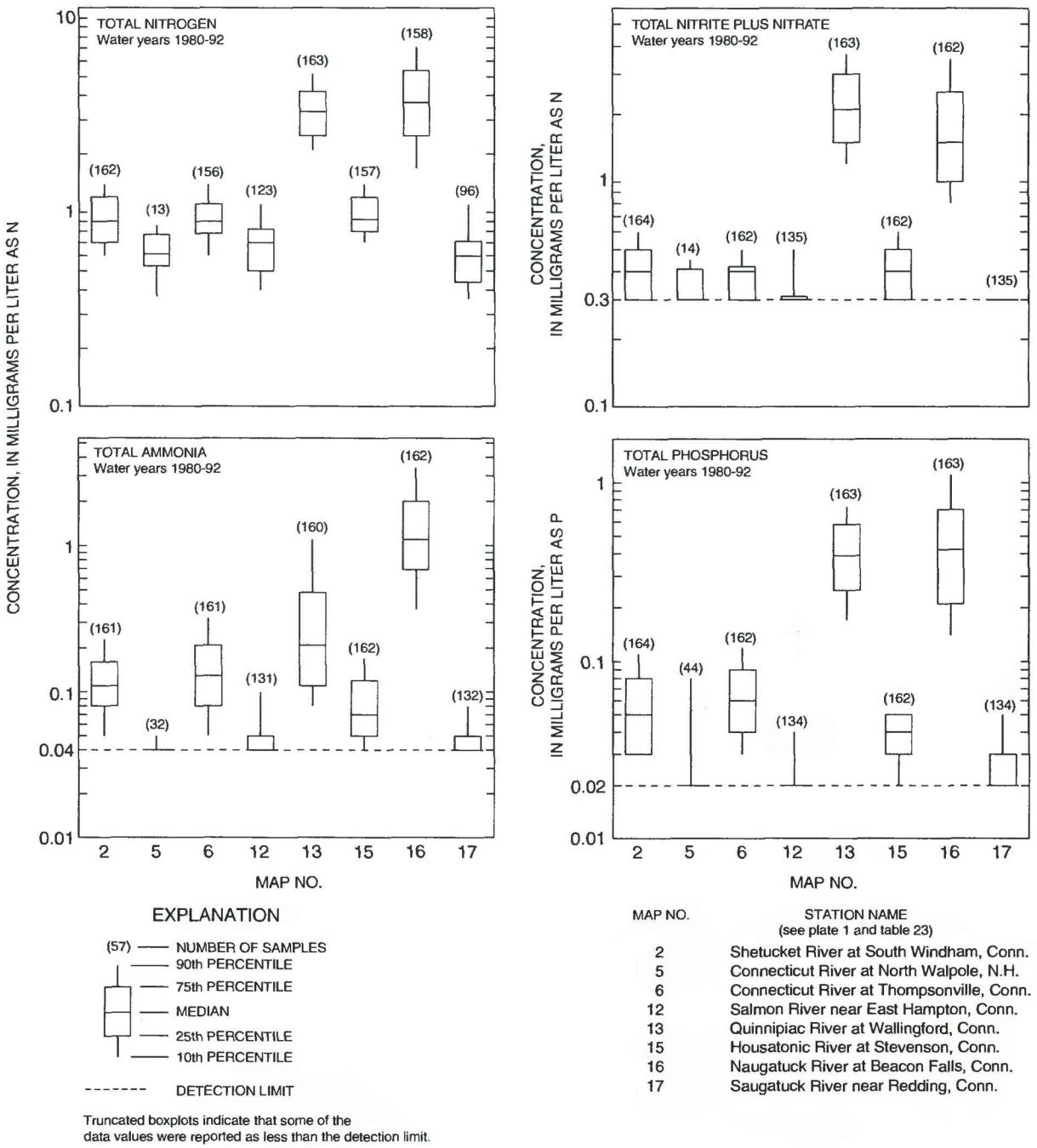


Figure 31. Concentrations of total nitrogen, nitrite plus nitrate, total ammonia, total phosphorus, and suspended sediment at selected streamflow-gaging stations in the Connecticut, Housatonic, and Thames River Basins study unit. (Data from U.S. Geological Survey's National Water Information System data base.)

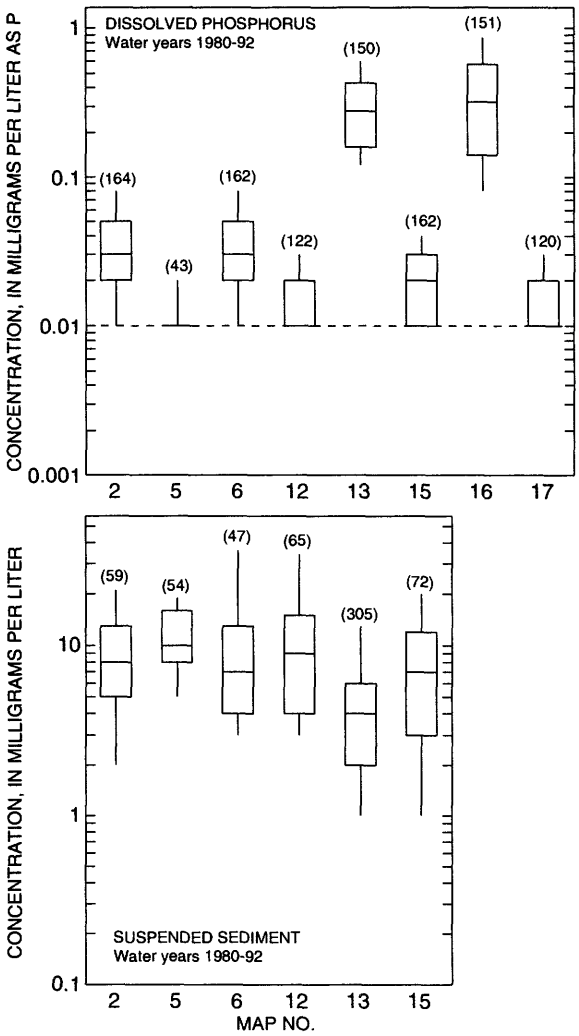


Figure 31. —*Continued.*

Median concentrations of suspended sediment for six stations in the study unit (fig. 31) are less than the national median for urban land-use settings and similar to the national median for forested basins (fig. 32). These six stations include five integrator basins with varying degrees of urban development, and one small, primarily forested basin. Four of the six drainage basins are underlain primarily by crystalline bedrock of the New England Upland and Mountain Upland environmental settings (fig. 4, table 9). One

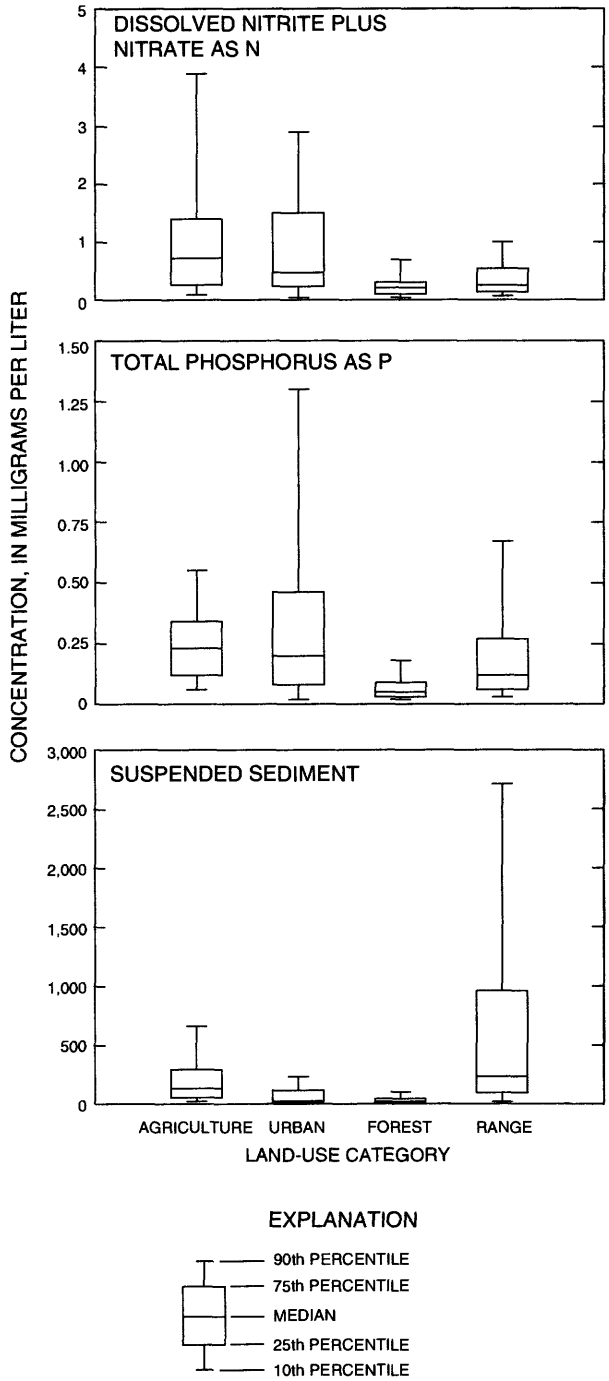


Figure 32. Concentrations of nitrite plus nitrate, total phosphorus, and suspended sediment in streams characteristic of four land-use categories nationwide, 1980-89. (Data from Dennis Helsel, U.S. Geological Survey, written Commun., 1992.)

station, the Connecticut River at Thompsonville, Conn., also includes arkosic bedrock of the Connecticut Valley Lowland in its drainage basin. The Housatonic River at Stevenson, Conn., includes carbonate bedrock of the Housatonic Valley Lowland environmental setting within its drainage basin. Land uses in these six basins are too varied, and the number of basins is too small to discern differences in suspended-sediment concentration that may be attributable to differences in geology. This is a potential area for further study.

In order to provide some insight into factors controlling nitrate concentrations, the analysis of data from the 185 ARMP sites considered correlation of nitrate concentrations with population density, land use, and stream order. Distribution of sites by town population density is as follows: low population-density towns (towns with less than 25 persons per mi^2) accounted for 20 percent of the sites; low to moderate density towns (25 to 150 persons per mi^2) had 49 percent; moderate to high density towns (151 to 500 persons per mi^2) contained 20 percent; and high population-density towns (more than 500 persons per mi^2) had 11 percent. Nitrate concentration correlates strongly with town population density (fig. 33). Median concentrations of nitrate (fig. 9) were high in streams near Northampton and Amherst in central Massachusetts, in the Springfield area, in tributaries to the southern reaches of the Deerfield River, in the Pittsfield area in the upper Housatonic drainage, and in the Quinebaug drainage. These regions encompass urban areas with the highest population density in the Massachusetts section of the study unit.

Sites in low population-density towns tend to have lower nitrate concentrations than sites in moderate to high population density towns, irrespective of existence of upstream dischargers. In towns with moderate to high population density and few point sources, median concentrations of nitrate are high. Low median concentrations of nitrate characterize some sites downstream from sewage-treatment plants in towns with low population density. Thus, much of the high nitrate concentrations in Massachusetts streams may result from nonpoint-source pollution. Sources of this high nitrate may include a combination of agriculturally derived nitrogen, leaky (or nonexistent) septic tanks, nonpoint urban runoff, and undocumented industrial or municipal discharges.

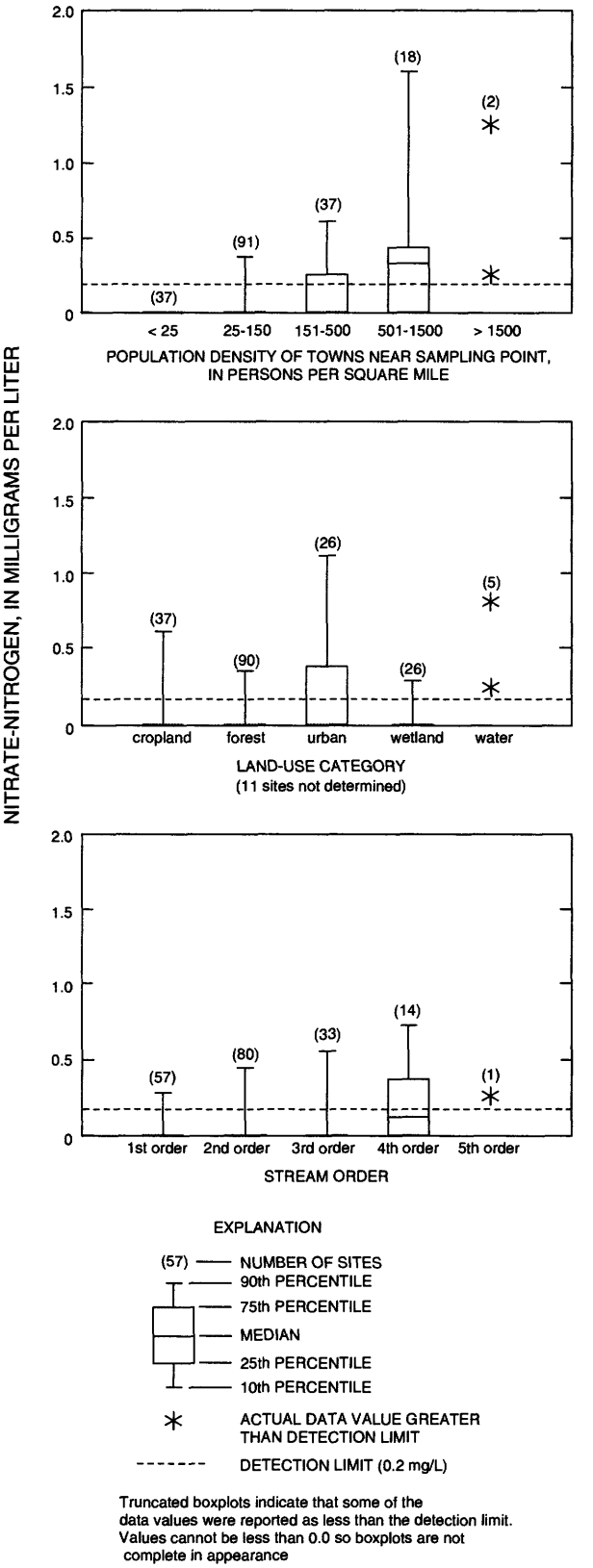


Figure 33. Median concentrations of nitrate as nitrogen in selected Massachusetts streams differentiated by population density, land use, and stream order, water years 1984-91.

The ARMP sites were located on a land-use map compiled in the mid-1970's to compare effects of land use on nitrate concentrations, which is the most recent land-use data available for the whole study unit (fig. 3). Twenty percent of the ARMP sites were in cropland areas, 49 percent in forested areas, 14 percent in urban areas, and 14 percent in wetlands. An additional 3 percent were classified as "water" because, at the scale of resolution of the data, the site was located on a significant water body. Although land use has changed in some areas since the period of land-use data compilation, the general categorization of ARMP sites by land use is believed to be accurate.

Relating land use to nitrate concentration indicates that median concentrations of nitrate were highest in urban areas (fig. 33). Forested sites generally have low nitrate concentrations and cropland and wetlands have intermediate nitrate concentrations. For example, median concentrations of nitrate are high at Hinsdale and Stafford Brooks, two tributaries to the Green River (0.61 mg/L and 0.62 mg/L, respectively); the predominant land use is agriculture in these basins. Median concentrations of nitrate are similar in the Swift and Ware Rivers, which drain primarily undeveloped, forested areas, and supply drinking water to metropolitan Boston (less than 0.2 mg/L). Nitrate concentrations are low (table 28, fig. 9) in the Millers River, which has many water-quality problems related to point sources (G.G. Girouard, U.S. Geological Survey, oral commun., 1992); the predominant land uses are forests and wetlands in this basin.

Of the 185 sites, first-order streams accounted for 31 percent, second order 43 percent, third order 18 percent, and fourth order or higher 8 percent. In general, stream order does not appear correlated with median nitrate concentrations for first, second, or third order streams; however, fourth and higher order streams, which are larger streams and may occur on more urbanized settings, consistently have higher median concentrations of nitrate (fig. 33).

Trend patterns

Trends in water-quality-constituent concentrations can indicate long-term improvement or deterioration in stream quality and may be caused by various conditions within a stream's drainage basin. Trend tests were performed for total nitrogen, total ammonia, total nitrite-plus-nitrate, total phosphorus, and dissolved phosphorus at 18 stations (table 10). Suspended-sediment trend results do not appear in table 10, since only two stations, the Housatonic River at Stevenson, Conn., and the Connecticut River at North Walpole, N.H., had adequate suspended-sediment data to perform trend tests and no trends were detected at these stations. Among the eight stations selected for graphical presentation, trends in flow-adjusted concentrations may not be readily apparent because of the scales of the graphs (fig. 34). However, where the trends are pronounced, such as at the Quinnipiac River at Wallingford, Conn., the LOWESS (LOcally Weighted Scatterplot Smoothing) curves (Cleveland, 1979) plotted on these graphs may be quite informative, because they can show points in time when trends developed or changed.

Trend analyses for the 18 stations indicated (table 10) either no trends or downward trends in nutrients at most stations during water years 1980-92. At most stations, total and dissolved phosphorus trends were downward. These downward trends may be due to improvements in wastewater treatment, changes in agricultural fertilization practices, or the elimination of phosphate from some laundry detergents.

The bias in phosphorus data discussed earlier in this report is assumed to have had a negligible effect on the analyzed trends, because of the low concentrations of phosphorus and suspended sediment at most stations. The change in NWQL methodology may result in higher reported phosphorus concentrations after May 1, 1990, than those reported before May 1 for some environmental settings. However, data for all stations evaluated indicate either a downward trend or no trend, indicating that the method change has not produced any false positive trends. The preponderance of downward trends in phosphorus also is consistent with preliminary results of a study of stream-quality trends in Connecticut for water years 1974-88 (E.C.T. Trench, U.S. Geological Survey, written commun., 1993).

The only upward trend in total nitrogen is at the Willimantic River at Merrow, Conn. Upward trends determined for total nitrite-plus-nitrate were associated with either a downward trend in total ammonia or an upward trend in total nitrogen. Most stations with increases in nitrite-plus-nitrate either represent urban settings or integrate large drainage basins with multiple environmental settings. This land-use association suggests interpretations related to wastewater treatment.

Table 10. Summary of trends in flow-adjusted concentrations of nutrients at selected stations in the Connecticut, Housatonic, and Thames Rivers study unit, water years 1980-92

[Map No.: See plate 1 for location of water-quality sampling stations. All trend tests were monthly, except as noted. Trends were considered significant for $p<0.05$. --, insufficient data to determine trends]

Map No.	Station name	Trends				
		Nitrogen, total	Ammonia, total	Nitrite plus nitrate, total	Phosphorus, total	Phosphorus, dissolved
1	Willimantic River at Merrow, Conn.	Upward	No	Upward	Downward	Downward
2	Shetucket River at South Windham, Conn.	No	No	No	Downward	Downward
3	Quinebaug River at Quinebaug, Conn.	Downward	Downward	No	Downward	Downward
4	Quinebaug River at Jewett City, Conn.	No	Downward	No	Downward	Downward
5	Connecticut River at North Walpole, N.H.	--	--	--	No[1]	No[2]
6	Connecticut River at Thompsonville, Conn.	No	No	No	Downward	Downward
7	Still River at Riverton, Conn.	No	Downward	No	Downward	Downward
8	Burlington Brook near Burlington, Conn.	--	--	--	--	--
9	Farmington River at State Highway 4, at Unionville, Conn.	No	Downward	No	Downward	Downward
10	Pequabuck River at Farmington, Conn.	No	Downward	Upward	Downward	Downward
11	Farmington River at Tariffville, Conn.	No	Downward	Upward	Downward	Downward
12	Salmon River near East Hampton, Conn.	No	No	No	No[2]	No[2]
13	Quinnipiac River at Wallingford, Conn	No	Downward	Upward	Downward	Downward
14	Housatonic River near New Milford, Conn.	No	Downward	No	Downward	Downward
15	Housatonic River at Stevenson, Conn.	No	Downward	Upward	Downward	Downward
16	Naugatuck River at Beacon Falls, Conn.	No	No	No	No	No
17	Saugatuck River near Redding, Conn.	No	Downward	No	No	No[2]
18	Norwalk River at Winnipauk, Conn.	Downward	Downward	Downward	Downward	No

[1]Quarterly data.

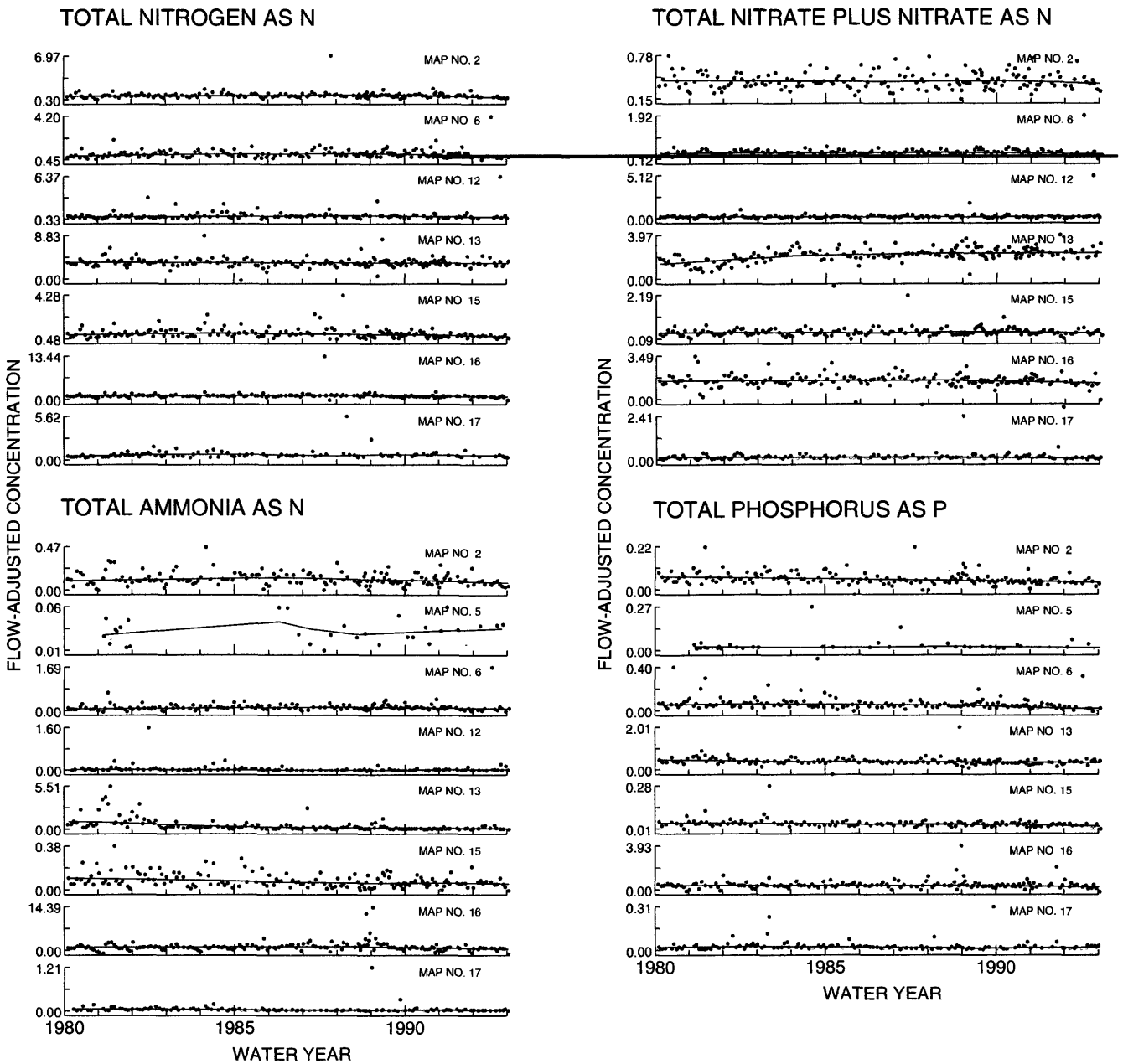
[2]Concentration trend test only, no flow adjustment.

Improvements in municipal wastewater treatment are likely to be reflected in downward trends in phosphorus and ammonia, as evidenced here. The concurrence of upward trends in concentrations of nitrite plus nitrate with downward trends in ammonia further reflects these improvements; that is, as sewage is treated, reduced nitrogen compounds are oxidized and more of the discharged nitrogen occurs in the form of nitrate.

Additional data for dissolved oxygen, evaluated as part of this study, but not presented in this report, showed widespread increasing trends in dissolved-oxygen concentrations for water years 1972-92, but few increasing trends during water years 1980-92. This ancillary information supports an inference that the major improvements to water quality in these basins were achieved in the 1970's.

At primarily forested, low population-density sites, such as the Salmon River near East Hampton, Conn., and the Saugatuck River near Redding, Conn., which have no major point or nonpoint sources, concentrations of nitrogen and phosphorus are historically low and there are no significant phosphorus trends. The Saugatuck River near Redding, Conn., has a downward trend in ammonia.

One station affected by major urban areas, the Naugatuck River at Beacon Falls, Conn., exhibits no trends in nitrogen or phosphorus while also having some of the highest nitrogen and phosphorus concentrations of any station examined. The lack of trends in nutrients during water years 1980-92 may be explained by recognizing that major improvements in waste treatment in that basin were achieved in the previous decade. No major municipal wastewater treatment upgrades took place during the period covered by the trend analysis (W.R. Hogan, Connecticut Department of Environmental Protection, oral commun., 1993). All



EXPLANATION

(see plate 1 and table 10)

MAP NO.	STATION NAME
2	Shetucket River at South Windham, Conn.
5	Connecticut River at North Walpole, N.H.
6	Connecticut River at Thompsonville, Conn.
12	Salmon River near East Hampton, Conn.
13	Quinnipiac River at Wallingford, Conn.
15	Housatonic River at Stevenson, Conn.
16	Naugatuck River at Beacon Falls, Conn.
17	Saugatuck River near Redding, Conn.

—— CONCENTRATION TREND

CENSORED VALUE

• FLOW ADJUSTED VALUE

Figure 34. Relations between selected nutrient water-quality constituents and time using smoothed, flow-adjusted, LOWESS analysis at selected stations, 1980-92.

the municipal sewage-treatment plants that discharge to the Naugatuck River achieved secondary level waste treatment by 1976; advanced treatment projects are planned for completion in the late 1990's (Connecticut Department of Environmental Protection, 1992, p. 47-48). This basin contains the highest density of dischargers in the study unit and further substantial reductions in nutrients may be difficult to achieve.

The trend analyses presented here further support the need for additional data collection from primarily agricultural basins in the study unit. Agricultural areas, with their less easily managed nonpoint sources of nutrients, have received inadequate study to determine their contributions to water-quality conditions, trends, and nutrient loads in the study unit's streams.

Ground Water

Nitrogen, in the form of nitrate (NO_3^-), probably is the most common contaminant in ground water (Freeze and Cherry, 1979, p. 413). Under oxidizing conditions, nitrate usually is the most prevalent and stable form of nitrogen in ground water. However, nitrogen also may occur as ammonium, ammonia, nitrite, nitrogen gas, and organic nitrogen. Although there are no solubility constraints on the concentration of nitrate in ground water, chemical and biological processes control the speciation of nitrogen and direct its transport between soil, water, plants and animals, and the atmosphere. In the absence of human sources, nitrate concentrations in ground water are typically low, less than 2.0 mg/L as nitrogen (Feth, 1966, p. 49). Nonpoint sources of nitrogen in ground water include agricultural use of organic and inorganic fertilizers, subsurface disposal of sewage, application of lawn and garden fertilizer, domestic animal wastes, leakage from sewer lines, and atmospheric deposition (Feth, 1966; Ragone and others, 1980). Nitrate concentrations in excess of 10 mg/L as N (44 mg/L as nitrate) in drinking water pose a health hazard and may cause methemoglobinemia in infants and, consequently, the U.S. Environmental Protection Agency (1975) has set the drinking-water regulation at this concentration.

Phosphorus is a relatively abundant element in soil and rock, but the low solubilities of its common inorganic compounds control its concentration in natural ground water to a few tenths of a milligram per liter (Hem, 1985, p. 126). Although phosphorus can occur in a number of oxidation states, the only phosphorus species present in natural ground waters under oxidizing conditions is the fully oxidized form—phosphate. Anthropogenic phosphorus sources include fertilizers, and domestic and industrial sewage effluents. Phosphate was a principal component of many household detergent formulations in the 1950's and 1960's, but increasing awareness of the role of phosphorus enrichment as the primary factor in cultural eutrophication, led to greatly restricted use in detergents. Some insecticides (malathion, parathion, ethion, and diazinon among other "organophosphorus" insecticides) incorporate reduced forms of phosphorus. Generally, the low solubility and high sorption of most of these compounds enhance the likelihood of their chemical and biological degradation and limit their environmental persistence (Smith and others, 1988, p. 36-39), except in some reducing environments (Hem, 1985, p. 127).

Previous Studies

High nitrate concentrations in ground water were noted as early as 1954 in the study unit, when nitrate concentrations were as high as 86 mg/L (19.4 mg/L as N) in wells completed in stratified drift in north-central Connecticut (Pauszek, 1961, p. 38-40). These excessive nitrate concentrations were attributed to surface contamination. Subsequently, the Connecticut water-resources-inventory studies conducted from 1963 to 1972 listed 18 wells, mostly completed in bedrock, but also in stratified drift and till, that had nitrate concentrations greater than 45 mg/L (10 mg/L as N), with the highest reported concentration being 171 mg/L (38.6 mg/L as N) (Randall and others, 1966; Thomas and others, 1967, 1968; Ryder and others, 1970, 1981; Cervione and others, 1972; Wilson and others, 1974; Mazzaferro and others, 1979; Weiss and others, 1982; Handman and others, 1986). Nitrate concentrations were highest in samples from the upper 75 ft of stratified-drift aquifers in north-central Connecticut (Ryder and others, 1981, p. 63).

The Connecticut water-resources-inventory studies reported that nitrate concentrations in 8 to 46 percent of the wells sampled in each basin exceeded the 2.2 mg/L (as N) "background concentration" perceived to represent natural conditions. The Quinnipiac River Basin, which is among the most highly urbanized of the large drainage basins in the study unit, had the highest frequency of wells with nitrate concentrations exceeding 2.2 mg/L (as N) (46 percent) (Mazzaferro and others, 1979). Melvin and others (1988, p. 192-194) reported median nitrate concentrations less than

1.0 mg/L (as N) in bedrock and stratified-drift aquifers in Connecticut and only one percent of all wells had nitrate concentrations exceeding the MCL of 10 mg/L (as N). By 1990, nitrates from agricultural practices, waste disposal, or urbanization were identified as the cause of 6.1 percent of the 1,368 incidents of contaminated wells on file at the Connecticut Department of Environmental Protection (1990, p. 65).

More recently, other studies were conducted to relate anthropogenic sources of contaminants, including nutrients, to shallow ground-water quality in Connecticut. Statistical analysis of new data on nutrient concentrations in ground water has identified relations between water quality and common land uses or agricultural practices. Grady and Weaver (1988, p. 25), working with a small sample size, did not discern statistically significant differences for nitrate and ammonia concentrations in ground water beneath agricultural, residential, and mixed industrial and commercial land-use areas, although the median concentrations were highest in residential areas. Subsequently, after augmenting the number of wells sampled and including wells in undeveloped reference sites, Grady and Weaver (1989, p. 24-25) reported significantly higher ($p = 0.0002$) nitrite plus nitrate concentrations in all three developed land-use categories than in the undeveloped reference sites. After installing a network of 116 observation wells in four stratified-drift aquifers in Connecticut, Grady (1993) reported significantly higher nitrite-plus-nitrate concentrations in ground water in tilled agricultural, sewered and unsewered residential, and commercial land-use areas than in ground water in undeveloped areas ($p = 0.0001$) (fig. 35). The median concentration of nitrite plus nitrate for ground-water samples from the undeveloped areas, 0.11 mg/L (as N), represented essentially natural (unaffected by human activity) water-quality conditions. Median concentrations of nitrite plus nitrate were highest (4.7 mg/L as N) in ground-water samples from tilled agricultural areas where silage corn was the principal crop. High concentrations of nitrite plus nitrate in water from the shallow, stratified-drift aquifers beneath tilled agricultural areas were assumed to result from the use of organic and inorganic fertilizers. A significantly higher frequency of ammonia was detected in tilled agricultural, sewered residential, commercial, and industrial areas than in the undeveloped areas ($p = 0.0415$).

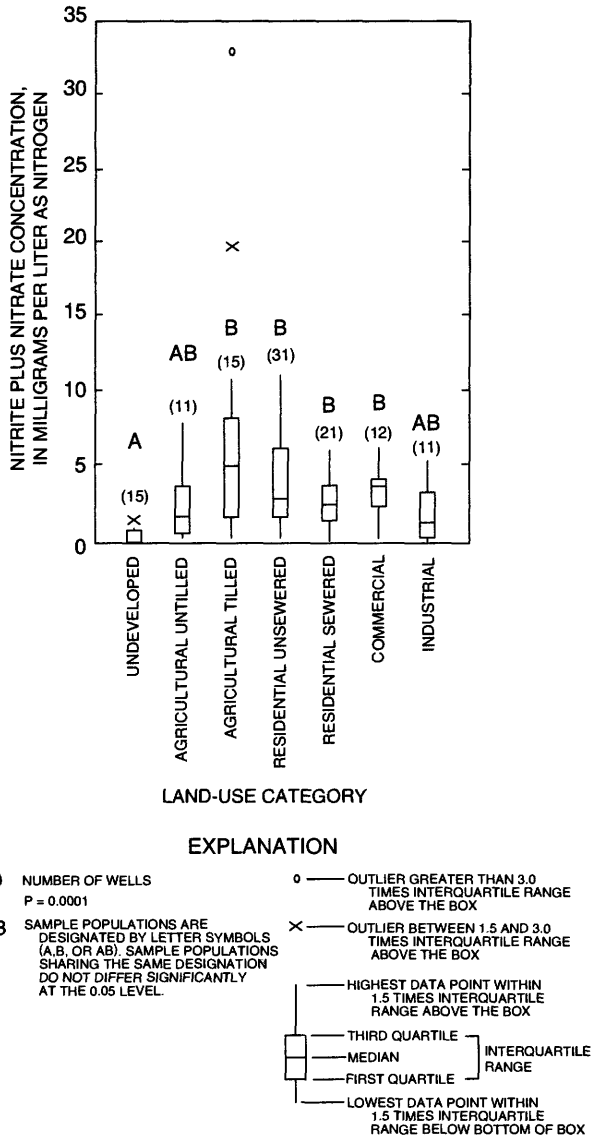


Figure 35. Relations between land use and nitrate concentrations in stratified-drift aquifers in Connecticut, 1985-89 (from Grady, 1993).

Mullaney and others (1991, p. 18-20) statistically compared concentrations of nitrite plus nitrate in ground water from agricultural and nonagricultural sites where pesticides were applied. Agricultural sites included tilled fields with corn, potato or vegetable cultivation, orchards, and ornamental plant nurseries. Nonagricultural sites included residential and commercial areas and recreational fields and golfcourses. Ground-water samples were collected primarily from stratified-drift aquifers but shallow till and bedrock

(1988) and Cotton (1988) summarized ground-water-quality conditions for New Hampshire and Vermont, respectively, using analyses of samples from public-supply wells. Morrissey (1988, p. 363-364) used analyses of 57 samples collected from 1980 to 1985 to compare water quality between stratified-drift and crystalline-bedrock aquifers in New Hampshire. Median nitrate concentrations were less than 1 mg/L as N for both aquifers and the maximum concentration in the State was 3.4 mg/L (Morrissey, 1988, p. 364). Median nitrate concentrations in Vermont were less than 0.5 and 0.6 mg/L as N, respectively, for crystalline-bedrock and stratified-drift aquifers, based on 218 samples collected from 1979 to 1986 (Cotton, 1988, p. 502-504). Nitrate concentrations were less than 2.0 mg/L (Cotton, 1988, p. 504) in 90 percent of the samples from public-supply wells in stratified-drift aquifers in Vermont and 75 percent of the samples in crystalline bedrock were less than the detection level of 0.5 mg/L.

Analysis of Information

Analysis of NWIS data on nutrient concentrations in ground water was summarized for wells and springs in the study unit. Summary statistics present analyses for 19 nutrient constituents (table 11). Sample size was too small (less than 10 observations) to provide percentile values for 12 constituents in springs and 5 constituents in wells. Median concentrations for all nutrients except total nitrite plus nitrate were less than 1.0 mg/L. High concentrations of nitrite plus nitrate (46 mg/L as N, total; 60 mg/L as N, dissolved) and phosphorus (23 mg/L as P, total) were measured in some wells in the study unit. Whether point or nonpoint sources caused high nutrient concentrations generally is unknown; hence, no effort was made for this analysis to screen any well from the data base on the basis of measured concentration.

In addition to constraints imposed on the analysis of data by small sample size, sample population distributions were severely censored. That is, 50 percent or more of the observations were less than detection limits for nearly one-half of the nutrient constituents (table 11) sampled in wells and springs.

Data for total and dissolved nitrite plus nitrate were among the most numerous, least censored, and most recent (period of record is water years 1973-93) of the nutrient constituents. Consequently, aquifers also were sampled. Concentrations of nitrite plus nitrate were significantly higher in samples from the agricultural sites than in samples from the nonagricultural sites ($p = 0.001$). Median concentrations of nitrite plus nitrate was 3.90 mg/L (as N) for samples from the stratified-drift aquifer beneath agricultural sites, but was 9.30 mg/L for samples from shallow till and bedrock aquifers beneath agricultural sites. Median concentrations were significantly lower for stratified-drift aquifers (2.85 mg/L) and till or bedrock aquifers (1.95 mg/L) beneath nonagricultural areas than for agricultural areas. Concentrations of nitrite plus nitrate also were related to the detection of pesticides at agricultural sites. Where pesticides were detected, median concentrations of nitrite plus nitrate were significantly higher ($p = 0.003$) in agricultural sites than in nonagricultural sites.

Published information on the occurrence and distribution of nutrients in ground water for areas of the study unit outside of Connecticut was largely limited to monitoring programs of public supply wells. In Massachusetts, Trombley (1992) assembled and analyzed data from public-supply wells for 1975-86. Trombley (1992, p. 53-55) reports summary statistics for nitrate, nitrite, and ammonia concentrations in ground water in wells in eight subbasins in the study unit in Massachusetts. Median nitrate concentrations for the eight basins were less than 1.0 mg/L (as N) and ranged from less than 0.1 to 0.7 mg/L. The maximum concentration for any Massachusetts public-supply well sampled in the study unit was 9.3 mg/L (as N). Median nitrite and ammonia concentrations were less than 0.0002 and 0.02 mg/L as N, respectively, in wells in six of the eight basins. Exceptions were the Millers and Quinebaug River Basins. Trombley further reports (1992, p. 36) that nitrate and ammonia concentrations were significantly correlated to basin population density ($p = 0.01$). Trends in nitrate concentrations decreased ($p = 0.1$) for the Chicopee River and Quinebaug River Basins, whereas trends increased in wells in the French River Basin (Trombley, 1992, p. 39).

Published information on the occurrence and distribution of nutrients in ground water for the northern part of the study unit was even more limited and, where available, has not been compiled by drainage basin. Hence, available data were difficult to relate directly to conditions in the study unit. Morrissey

Table 11. Summary of nutrient concentrations in ground water in the Connecticut, Housatonic, and Thames River Basins study unit

[Data from the U.S. Geological Survey's National Water Information System (NWIS). Ground-water data include samples from wells (water years 1917-93) and springs (water years 1951-93); reporting level is the least concentration detectable by the analytical methods used during the period of record. mg/L, milligram per liter; percentile values less than the reporting level were estimated using the method described by Helsel and Cohn (1988). --, insufficient data to estimate percentile value due to small sample size (less than 10 samples). na, not applicable; <, actual value is less than value shown]

Parameter code	Nutrient	Reporting level (mg/L)	Number of samples	Number of detections (percentage detected)	Minimum detected value (mg/L)	Maximum (mg/L)	Concentration at indicated percentile (mg/L)				
							10	25	50	75	90
				Wells							
91003	Nitrogen, nitrate, dissolved (as N)	0.05	3	1 (33)	na	0.38	--	--	--	--	--
00615	Nitrogen, nitrite, total (as N)	.001	104	69 (66)	0.001	.2	0.002	0.004	0.007	0.01	0.016
00613	Nitrogen, nitrite, dissolved (as N)	.001	216	59 (27)	.001	.4	.0009	.002	.004	.008	.013
00630	Nitrogen, nitrite plus nitrate, total (as N)	.01	367	320 (87)	.01	46	.03	.143	1.9	6.2	12.2
00631	Nitrogen, nitrite plus nitrate, dissolved (as N)	.01	576	439 (76)	.01	60	.026	.101	.59	3.2	6.1
00610	Nitrogen, ammonia, total (as N)	.001	265	124 (46)	.004	4.8	.0003	.001	.007	.03	.162
00608	Nitrogen, ammonia, dissolved (as N)	.01	513	346 (67)	.01	5.8	.002	.006	.02	.06	.21
00625	Nitrogen, ammonia plus organic, total (as N)	.01	93	83 (89)	.02	6.5	.045	.1	.29	.72	3.22
00623	Nitrogen, ammonia plus organic, dissolved (as N)	.1	181	107 (61)	.11	4.8	.056	.109	.2	.44	.8
00665	Phosphorus, total (as P)	.001	160	122 (76)	.002	23	.002	.009	.04	.12	.65
00650	Phosphate, total (as PO_4)	.01	205	180 (88)	.01	7.9	.008	.03	.06	.1	.23
00666	Phosphorus, dissolved (as P)	.001	257	130 (50)	.001	2.3	.001	.002	.007	.016	.06
70507	Phosphorus, orthophosphate, total (as P)	.001	316	163 (52)	.002	.65	.001	.003	.008	.02	.05
00671	Phosphorus, orthophosphate, dissolved (as P)	.01	516	172 (33)	.01	6.6	.0003	.001	.004	.015	.051
00669	Phosphorus, hydrolyzable, total (as P)	.01	2	1 (50)	<.01	.03	--	--	--	--	--
00672	Phosphorus, hydrolyzable, dissolved (as P)	.01	1	1 (100)	.05	.05	--	--	--	--	--
00678	Phosphorus, hydrolyzable, plus orthophosphate, total (as P)	.01	5	5 (100)	.04	.09	--	--	--	--	--
00677	Phosphorus, hydrolyzable, plus orthophosphate, dissolved (as P)	.01	36	20 (55)	.01	.34	.002	.005	.01	.038	.06
91004	Phosphorus, orthophenol, dissolved (as P)	.06	3	0 (0)	na	<.06	--	--	--	--	--

Table 11. Summary of nutrient concentrations in ground water in the Connecticut, Housatonic, and Thames River Basins study unit—*Continued*

Parameter code	Nutrient	Reporting level (mg/L)	Number of samples	Number of detections (percentage detected)	Minimum detected value (mg/L)	Maximum (mg/L)	Concentration at indicated percentile (mg/L)				
							10	25	50	75	90
	Springs										
91003	Nitrogen, nitrate, dissolved (as N)	0.05	0	0 (0)	--	--	--	--	--	--	--
00615	Nitrogen, nitrite, total (as N)	.001	2	2 (100)	0.001	0.005	--	--	--	--	--
00613	Nitrogen, nitrite, dissolved (as N)	.011	13	2 (15)	.01	.01	--	--	--	--	0.01
00630	Nitrogen, nitrite plus nitrate, total (as N)	.01	10	9 (90)	.01	1.0	0.009	0.023	0.067	0.305	.97
00631	Nitrogen, nitrite plus nitrate, dissolved (as N)	.01	14	7 (50)	.13	4.3	.008	.027	.096	.36	2.7
00610	Nitrogen, ammonia, total (as N)	.01	1	0 (0)	na	<0.01	--	--	--	--	--
00608	Nitrogen, ammonia, dissolved (as N)	.01	23	12 (52)	.01	.34	.0008	.003	.01	.06	.21
00625	Nitrogen, ammonia plus organic, total (as N)	.01	8	8 (100)	na	1.2	--	--	--	--	--
00623	Nitrogen, ammonia plus organic, dissolved (as N)	.1	14	3 (21)	.19	.2	.183	.186	.191	.195	.199
00665	Phosphate, total (as P)	.01	11	9 (82)	.03	.1	.014	.03	.05	.06	.092
00650	Phosphate, total (as PO_4)	.01	0	0 (0)	--	--	--	--	--	--	--
00666	Phosphorus, dissolved (as P)	.01	14	6 (43)	.01	.04	.003	.005	.008	.013	.035
70507	Phosphorus, orthophosphate, total (as P)	.001	2	0 (0)	--	--	--	--	--	--	--
00671	Phosphorus, orthophosphate, dissolved (as P)	.01	5	3 (60)	.01	.033	--	--	--	--	--
00669	Phosphorus, hydrolyzable, total (as P)	.01	0	0 (0)	--	--	--	--	--	--	--
00672	Phosphorus, hydrolyzable, dissolved (as P)	.01	0	0 (0)	--	--	--	--	--	--	--
00678	Phosphorus, hydrolyzable, plus orthophosphate, total (as P)	.01	0	0 (0)	--	--	--	--	--	--	--
00677	Phosphorus, hydrolyzable, plus orthophosphate, dissolved (as P)	.01	0	0 (0)	--	--	--	--	--	--	--
91004	Phosphorus, orthophenol, dissolved (as P)	.06	0	0 (0)	--	--	--	--	--	--	--

nitrite-plus-nitrate data were examined in this analysis for potential relations to the study unit's hydrogeologic and land-use characteristics. In total, 565 NWIS wells had one or more sample for either total or dissolved nitrite-plus-nitrate. Data for total and dissolved nitrite-plus-nitrate were combined, and median concentrations were determined when more than one sample was reported for any well.

Nitrite-plus-nitrate data were not evenly distributed by aquifer type (fig. 36). Stratified-drift aquifers, with 69 percent of the wells, were sampled more frequently than other aquifers in the study unit, reflecting the importance of the stratified-drift aquifers for water supply. Among the bedrock aquifers, the carbonate bedrock (two wells) was effectively not sampled for nitrite plus nitrate and the number of wells from the arkosic bedrock (22 wells, or 4 percent) appeared inadequate in light of the large variance in concentrations. Only five wells in alluvium had nitrite-plus-nitrate data. Alluvium is not a major water-supply aquifer in the study unit, and commonly is difficult to distinguish from stratified-drift aquifers composed of sand and gravel.

Concentrations of nitrite plus nitrate generally are low in all aquifers (fig. 36). Median concentrations are less than 1.0 mg/L (as N) in all aquifers except the till, where the median is 1.5 mg/L. About 20 percent of the samples from wells in till aquifers exceed the 10 mg/L (as N) MCL for nitrite plus nitrate. This apparent association of high concentrations of nitrite plus nitrate with the till aquifers may be an artifact of the relatively small sample size together with a biased distribution of till wells with respect to land use. More than one-half (26) of the till wells sampled for nitrite plus nitrate are in agricultural or nonagricultural sites where agricultural chemicals (pesticides and fertilizers) are known to be used. Other reasons for high nutrient concentrations in till aquifers include shallow water tables (Melvin and others, 1992b, p. 25) and the presence of near-surface features (fractures, macropores, and lenses of stratified drift) that enhance contaminant migration from the surface (Melvin and others, 1992a, p. 16-17; 1992b, p. 31-32).

Median concentrations of nitrite plus nitrate are less than 1.0 mg/L (as N) for all well-depth categories except the shallowest (less than 50 ft) where the median concentration is 1.2 mg/L (fig. 37). All but 1 of the 28 wells with concentrations of nitrite plus nitrate greater

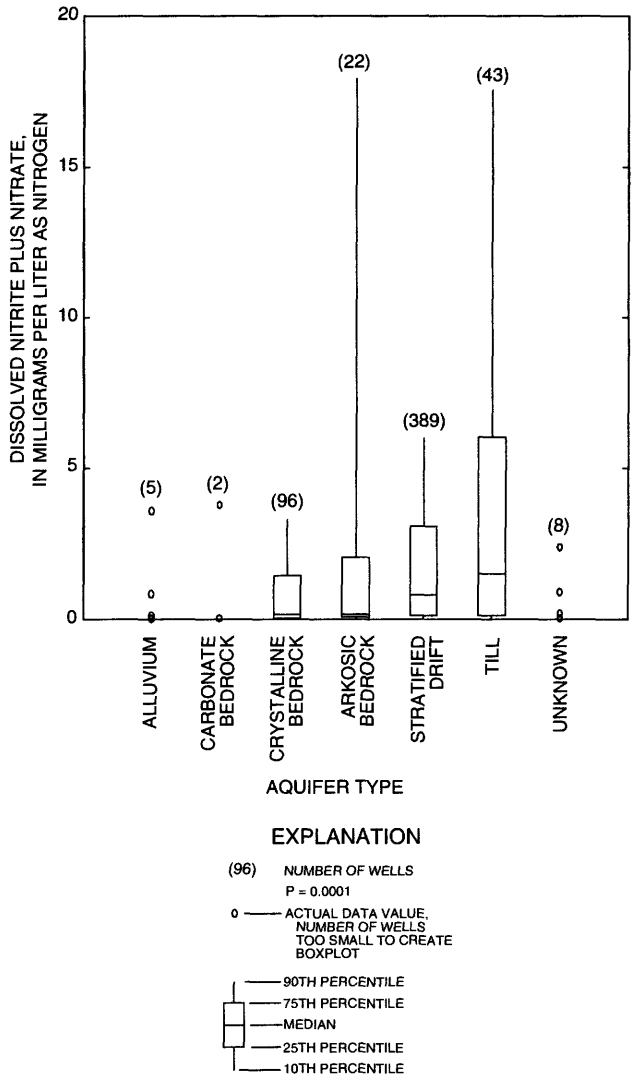


Figure 36. Concentrations of dissolved nitrite plus nitrate in ground water by aquifer type in the Connecticut, Housatonic, and Thames River Basins study unit, 1973-93. (Data from U.S. Geological Survey's National Water Information System data base.)

than 10 mg/L are shallower than 50 ft. High nitrogen concentrations in the shallowest wells may reflect: (1) the surface and shallow subsurface contributions of nitrogen from agricultural and lawn fertilizers and sewage disposal through septic tanks, (2) vertical transport to the water table with infiltrating recharge from precipitation, snowmelt, or irrigation, (3) lateral movement in the upper part of the ground-water-flow system with little mixing, dispersion or diffusion, or (4) less frequent sampling of deeper wells.

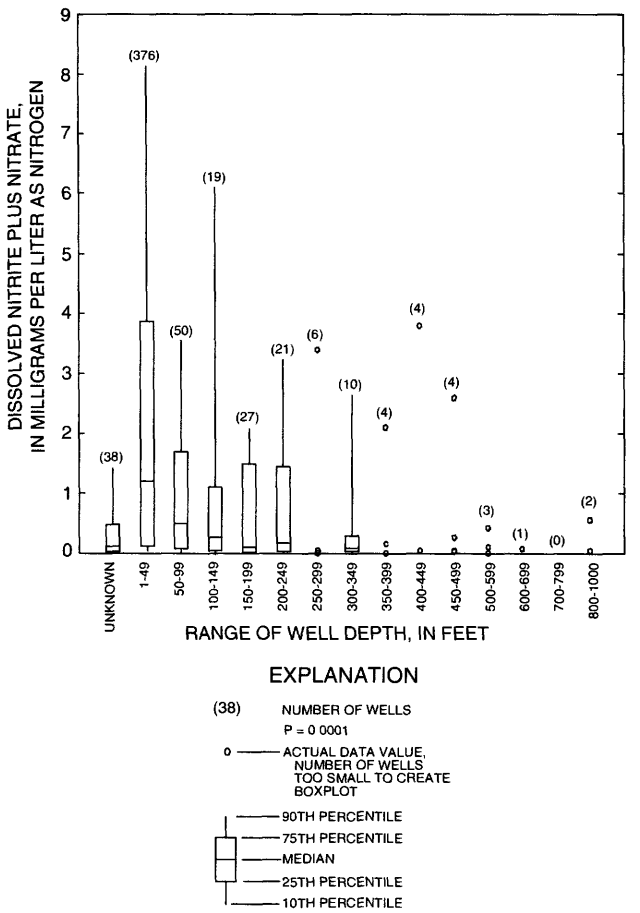


Figure 37. Concentrations of dissolved nitrite plus nitrate by well depth in the Connecticut, Housatonic, and Thames River Basins study unit, 1973-93. (Data from U.S. Geological Survey's National Water Information System data base.)

In order to evaluate concentrations of nitrite plus nitrate in ground water in relation to land use, the 565 NWIS wells sampled were assigned the USGS's Level I land-use categories (Anderson and others, 1976) based on Geographic Information Retrieval and Analysis System (GIRAS) coverages (Mitchell and others, 1977). A number of factors complicated the use of the GIRAS coverage for evaluating the effects of land use on ground-water quality. First, the 1970's GIRAS land-use coverage represented land-use conditions existing as much as two decades prior to the installation and sampling of some wells. Furthermore, the land-use coverage was produced at a scale of 1:250,000, a scale unlikely to accurately designate land use where wells were in areas of heterogeneous land use or on boundaries between land-use categories. Lastly, categorizing land use at wellhead sites did not necessarily reflect all important land-use impacts within the contributing area of the well.

All but seven wells with nitrite-plus-nitrate data were assigned to Level I urban, agricultural, forest, and wetland land-use/land-cover categories (fig. 38). A Kruskal-Wallis test was applied to test the null hypothesis that concentrations of nitrite plus nitrate were not significantly different for any of the land-use categories. Median concentrations of nitrite plus

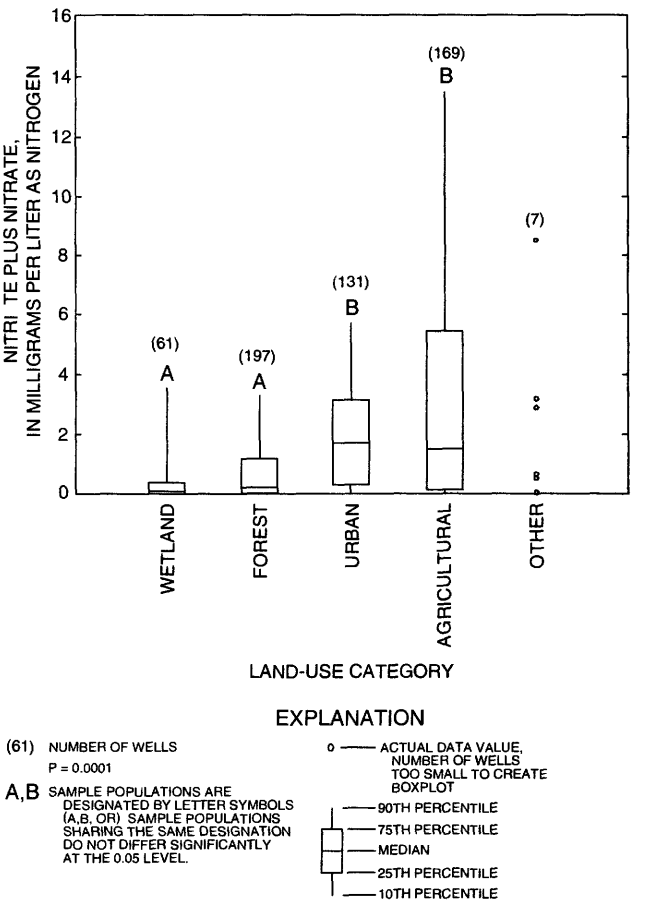


Figure 38. Concentrations of dissolved nitrite plus nitrate by land use in the Connecticut, Housatonic, and Thames River Basins study unit, 1973-93. (Data from U.S. Geological Survey's National Water Information System data base.)

nitrate were significantly greater ($p < 0.001$) for the urban (1.7 mg/L) and agricultural (1.5 mg/L) wells than the forest (0.24 mg/L) and wetland (0.10 mg/L) wells.

The wells listed in NWIS were sampled too infrequently to evaluate long-term trends for nutrient concentrations in ground water. However, as part of an evaluation of land-use effects on ground-water quality in Connecticut (Grady and Weaver, 1988, 1989), seven stratified-drift wells were sampled repeatedly for nitrite plus nitrate from June 1987 to July 1988. Four of the wells were in agricultural areas; two were in urban settings; and one was in a forested, undeveloped area (table 12).

The variance in concentrations of nitrite plus nitrate for five of the wells was quite small (less than 1.0), even though they included sites clearly affected by nitrogen sources (wells V 89, F 324, and V 96 with median concentrations of nitrite plus nitrate greater than 1.0 mg/L), as well as sites with no apparent enrichment of nitrite plus nitrate from anthropogenic sources (wells NT 100 and EL 81 with median concentrations less than 1.0 mg/L). The small variance in concentrations of nitrite plus nitrate in these five wells indicated that sources of nitrogen, or its transport to the wells, was relatively constant for the 1-year period of sample collection. The two wells with the highest median concentrations of nitrite plus nitrate (SB 37 and EL 85), however, also had highest variances (13.8 and 436, respectively). The high variance in concentrations of nitrite plus nitrate in these two wells may have reflected variable sources of nitrogen or intermittent transport to the wells over the sampling period.

Table 12. Median nitrite plus nitrate concentrations and variance for ground-water samples from seven wells in stratified-drift aquifers in Connecticut, June 1987 to July 1988

[Data from U.S. Geological Survey's National Water Information System. Wells listed in order of increasing variance. mg/L, milligram per liter]

Local well No.	Land use adjacent to or upgradient of well	Number of samples	Median nitrite plus nitrate concentrations (mg/L as N)	Variance
NT 100	Agricultural, hay	12	0.58	0.025
EL 81	Forest, undeveloped	8	.73	.044
V 89	Urban, residential	11	3.5	.577
F 324	Agricultural, corn	12	5.6	.241
V 96	Urban, commercial	11	5.8	.875
SB 37	Agricultural, berries	12	11	13.8
EL 85	Agricultural, corn	11	33	436

Pesticides

The number and type of synthetic organic compounds used to control pests have increased sharply since they were first introduced in the 1940's. Because of changing agronomic practices, State and Federal regulatory actions restricting or eliminating use of specific pesticides, as well as increased immunity of targeted organisms to some widely used formulations, the pesticide industry continually produces and introduces new products.

The occurrence and distribution of pesticides in surface water, streambed sediments, and aquifers in the study unit are functions of their usage and chemical characteristics, as well as hydrologic factors. The fate of various compounds and their metabolites in aquatic environments depends greatly on the degree to which they are soluble in water, their tendency to be sorbed onto sediment or other organic particles, and the extent to which they are degraded by physical, chemical, and biological processes. The movement of pesticides from areas of application to surface and ground water also depends on the timing and intensity of precipitation, the quantity and pathways of runoff from agricultural fields and urban areas, the type, thickness, and permeability of soils, and depth to the water table.

Soluble pesticide compounds such as atrazine tend to move readily from areas of application by surface runoff, or to infiltrate the water table where they move with the ground-water circulation. Hydrophobic compounds that degrade slowly, such as DDT, and that have not been used extensively in the study unit for decades, can persist in river sediments, where they may remain until resuspended during floods or ingested by aquatic organisms (Gilliom, 1985).

The use of pesticides in the study unit is extensive, considering the region's limited and declining agricultural economy. The most productive farmlands, most notably in the Connecticut Valley Lowland, are in the densely populated, southern half of the study unit, where the cultivation of vegetables, berries, and ornamental plants uses more varied and heavier applications of pesticides than commonly is associated with silage corn production for dairy farms. In addition, the rate of application of pesticide active ingredients in urban areas of the study unit may equal or exceed agricultural uses. Commercial and domestic applicators tend to use a broader spectrum of pesticide compounds than do farmers. Surface-water and bed-sediment samples have not been analyzed for many of the compounds used in the study unit.

Surface Water and Bed Sediment

Only 11 pesticide compounds or other metabolites were detected in NWIS surface-water samples collected in the study unit (table 13). Four of those were acetamide and triazine herbicides detected in storm runoff samples in one of the most heavily agricultural drainages, the Scantic River Basin in Connecticut. This basin was the only one in the study unit sampled for this class of pesticides. Trace-level concentrations of three organochlorine insecticides, one organophosphorus insecticide, and three chlorophenoxy-acid herbicides were reported for surface-water samples from 17 stations. Forty-eight NWIS stations in the study unit were sampled for these three classes of pesticides, although not every compound identified with each class was included in all analyses (see table 24). Most of the stations sampled were in Connecticut and almost all the detections were in that State. Pesticides were detected in surface-water samples at only two of the NWIS stations outside of Connecticut—the Connecticut River at Wilder, Vt., and the Connecticut River at Agawam, Mass. (fig. 20).

Total 2,4-dichlorophenoxy acetic acid (2,4-D) was the most commonly detected pesticide compound in the NWIS surface-water data, both in the number of sites (15) with detections and the percentage of samples (22) with detections, followed closely by the related compound total silvex. 2,4-D is a general purpose herbicide with agricultural, commercial, and residential uses and is among the 10 most heavily applied pesticides (tables 2 and 3) in the study unit. 2,4-D was detected at stations on many of the major streams in Connecticut, including the Shetucket, Quinebaug, and Yantic Rivers in the east, the Connecticut, Farmington, Salmon, and Quinnipiac Rivers in the central part of the State, and the Housatonic, Pomperaug, Saugatuck, and Norwalk Rivers in the west. This compound also was detected at two NWIS stations on the Connecticut River north of Connecticut (Agawam, Mass., and Wilder, Vt.). 2,4-D and the other chlorophenoxy-acid herbicides are relatively soluble, and rapidly degraded by biological, chemical, and physical processes (Smith and others, 1988, p. 40-41), and do not persist in the environment. The maximum concentration measured for NWIS surface-water samples from the study unit was 0.05 microgram per liter (µg/L); 2,4-D in surface water were not detected in the STORET data (table 14). The period of record for 2,4-D sampling was water years 1971-82, with most samples dating to 1973 or earlier.

Other pesticide compounds detected at more than one NWIS station (table 13) were the insecticides dieldrin and diazinon. STORET listed no diazinon data but dieldrin was detected in nearly two-thirds of the 21 STORET surface-water samples (table 14). Like the other organochlorine insecticides, dieldrin was not used extensively after the mid-1960's (Smith and others, 1988, p. 27). Low solubility and a high affinity for sediment particles generally limit dieldrin and other organochlorine insecticide concentrations in the water column to trace levels. Dieldrin concentrations in surface-water samples from the study unit were less than or equal to 0.03 µg/L. All dieldrin data for water-column samples from NWIS surface-water stations were from water year 1982 or earlier, whereas the STORET data were for the mid-1960's. Diazinon, a general purpose, organophosphorus insecticide and nematocide used heavily by commercial applicators in Connecticut (table 3), was detected at trace concentrations (less than or equal to 0.03 µg/L) in the Connecticut and Quinnipiac Rivers.

The CTDOHS pesticide data for surface-water sources (mostly reservoirs and a few stream diversions) indicated that pesticide detections were less frequent in sources used for drinking-water supplies. Only 1 of the 145 pesticide analyses inventoried in this study for 75 surface-water sources in Connecticut (fig. 22) reported any pesticides—trace levels of two organochlorine insecticides, methoxychlor (0.09 µg/L) and endrin (0.03 µg/L) measured in a sample collected in May 1992 from a reservoir in south-central Connecticut. Detection of pesticides in reservoirs or other drinking-water sources is less likely than in ambient surface water because large areas of their drainage basins

Table 13. Summary of pesticide detections and concentrations in surface water in the Connecticut, Housatonic, and Thames Rivers study unit from the U.S. Geological Survey's National Water Information System (NWIS) data base, water years 1969-92

[**Reporting level:** Least concentration detectable by analytical methods used during the period of record. **Drinking water MCL references:** U.S. Environmental Protection Agency (Code of Federal Regulations, v. 40, Parts 141, 142, and 143; 1991a; 1992a, 1992b). **Human health references:** U.S. Environmental Protection Agency (1991b, 1992c, 1992d). **Acronymns:** MCL, Maximum Contaminant Level; DDT, dichlorodichloroethylene; 2,4-D, (2,4-dichlorophenoxy) acetic acid; 2,4,5-T, (2,4,5-trichlorophenoxy) acetic acid. µg/L, microgram per liter. <, actual value is less than value shown. --, no data]

Pesticide (parameter code)	Reporting level (mg/L)	Number of sites or samples with detections (percentage shown in parentheses)				Range of concentrations detected (µg/L)	U.S. Environmental Protection Agency (USEPA) water-quality criteria		
		Sites		Samples			Drinking water MCL (µg/L)	Human health, ingestion of organisms (µg/L)	Freshwater, aquatic life, acute (µg/L) and USEPA reference
Organochlorine insecticides									
Aldrin, total[1] (39330)	0.01	1	(2)	1	(<1)	0.01	--	[1 2]1.4×10^{-4}	[3]3 (1980a,b,1991b,1992a,d,e)
DDT, total[1 2] (39370)	.01	1	(2)	1	(<1)	0.01	--	[1]5.9×10^{-4}	[3 4]1.1 (1980a,c,1991b, 1992a,c,d,e)
Dieldrin, total (39380)	.01	4	(10)	6	(3)	0.01-0.03	--	1.4×10^{-4}	[3 5]2.5 (1980a,b,1991b,1992c,d,e)
Organophosphorus insecticides									
Diazinon, total (39570)	.01	4	(15)	3	(3)	0.01-0.03	--	--	--
Chlorophenoxy-acid herbicides									
2,4-D, total (39730)	.01	15	(56)	23	(22)	0.01-0.05	70	--	--
2,4-D, dissolved (39732)	.02	1	(14)	1	(8)	0.02	70	--	--
2,4,5-T, total (39740)	.005	2	(7)	2	(2)	0.01-0.05	--	--	--
Silvex, total (39760)	.005	9	(33)	21	(20)	0.01-1.5	50	--	--
Silvex, dissolved (39762)	.005	1	(14)	1	(10)	0.05	50	--	--
Acetamide and triazine herbicides									
Atrazine, dissolved (39632)	.05	3	(100)	17	(61)	0.06-4.6	3	--	--
Desethylatrazine, dissolved (04040)	.05	3	(100)	20	(71)	0.06-0.91	--	--	--
Deisopropylatrazine, dissolved (04038)	.05	1	(33)	1	(4)	0.19	--	--	--
Metolachlor, dissolved (39415)	.05	3	(100)	10	(36)	0.09-0.25	--	--	--

[1] Criteria are based on carcinogenicity (10^{-6} risk). For a risk level of 10^{-5}, move the decimal point in the matrix one place to the right.

[2] Aldrin is converted to dieldrin in fish, so criteria also consider risk resulting from intake of dieldrin in fish due to the presence of aldrin in water, as well as the risk from intake of aldrin itself in fish and/or water. Therefore, criteria for aldrin also depend on the unit cancer potency (17 (milligrams per kilogram of body weight per day)$^{-1}$), derived from a linearized multi-stage model, and the weighted average bioconcentration factor, adjusted to the average percentage of lipids in fish and shellfish consumed by Americans (3 percent), for dieldrin. The intake of dieldrin resulting from aldrin in the water is estimated by assuming that, in the absence of dieldrin, aldrin would bioconcentrate 4,670 times (as dieldrin does), and that since aldrin accumulates 28 times, the remainder of the expected aldrin residues are being stored as dieldrin (4,670 - 28 = 4,642) (U.S. Environmental Protection Agency 1980b).

[3] Acute values shown are Final Acute Values, an instantaneous maximum value based on acute toxicity data for a variety of fish and invertebrate samples.

[4] Criteria apply to DDT and its metabolites.

[5] This is the value published in the 1980 water-quality criteria document (U.S. Environmental Protection Agency, 1980b). The Integrated Risk Information System (U.S. Environmental Protection Agency, 1992a) reported a different value for the acute freshwater criterion (1.0 milligrams per liter).

Table 14. Summary of pesticide detections and concentrations in surface water in the Connecticut, Housatonic, and Thames Rivers study unit from the U.S. Environmental Protection Agency's Storage and Retrieval System (STORET) data base

[DDD, dichlorodiphenyldichloroethane; DDE, dichlorodiphenyl-dichloroethylene; DDT, dichlorodiphenyltrichloroethane; µg/L, microgram per liter]

Pesticide (parameter code)	Period of record (water year)	Number of samples	Number of samples greater than reporting level	Range of concentrations (µg/L)
Organochlorine insecticides				
DDD, total (39360)	1962-67	21	2	0.004 - 0.013
DDE, total (39365)	1962-67	21	1	0.002
DDT, total (39370)	1962-67	21	1	0.005
Dieldrin, total (39380)	1962-67	21	14	0.002 - 0.023
Endrin, total (39390)	1962-76	23	2	0.014 - 0.025
Heptachlor, total (39410)	1962-80	24	1	0.037
Heptachlor epoxide, total (39420)	1962-76	23	1	0.001
Lindane, total (39340)	1962-67	21	2	0.002 - 0.004

generally were preserved in undeveloped conditions, if possible, with access and human activities strictly limited to prevent contamination.

Bed sediments were sampled more frequently for various pesticides, and at more locations in the study unit than surface water (tables 15 and 16; fig. 21). Bed sediments were analyzed for organochlorine insecticides at 82 NWIS streamflow-gaging stations, organophosphorus insecticides and chlorophenoxy-acid herbicides at more than 50 NWIS stations, and semi-volatile organic compounds at 14 stations. Seven organochlorine insecticides, two chlorophenoxy-acid herbicides, and one semi-volatile organic compound were detected at NWIS streamflow-gaging stations (table 15), and detections of most of the same compounds also were listed in the more limited STORET data (table 16). As with water samples, most bed-sediment pesticide samples were collected in Connecticut and most detections for those samples also were in Connecticut.

Organochlorine insecticide residues were detected at more stations and were more frequently detected in river, lake, and estuary sediments (table 15) than in water-column samples (table 13) from NWIS stations in the study unit. Four organochlorine insecticides (chlordane, DDD, DDT, and dieldrin) were detected in bed-sediment samples at more than one-half of the stations, and a fifth (DDE) at nearly one-half (44 percent) of the stations. DDT was the most widely detected pesticide, reported in bed sediments at 58 percent of the streamflow-gaging stations. Dieldrin was the most frequently detected compound, in 45 percent of all bed-sediment samples from all stations; chlordane, DDD, DDE, and DDT were detected in more than 25 percent of the samples. Concentrations ranged from 0.1 to 220 micrograms per kilogram (µg/kg) for some of these insecticides.

Most of the data on organochlorine pesticides in bed material were collected in water years 1973-77 (see fig. 23). Use of these compounds to combat termite infestations and soil insects in corn, vegetable, fruit, and ornamental plant production was banned or severely restricted in the early 1970's.

Six organochlorine insecticides were detected once or twice in bed sediments at STORET surface-water sites in the study unit, but the number of sites and samples were quite limited and reporting levels for some samples were too high to provide useful information (table 16). STORET bed-sediment samples, however, were collected more recently than most of the NWIS data. Concentrations for five of the six organochlorine insecticides in bed sediments from STORET stations were low, less than 1.0 µg/kg, but one sample contained 5.3 µg/kg of heptachlor epoxide.

The pattern of detections of organochlorine compounds has as much to do with chemical properties and analytical capabilities as it does with past usage (Gilliom, 1985, p. 87). Few organochlorine compounds have been detected in water samples because, as a class, they have low solubilities and tend to adsorb onto sediment particles. Some of the DDD and DDE organochlorine compounds detected in bed-sediment samples are metabolic degradation products of other compounds, such as DDT, and were commonly detected although DDD was used on a limited basis and DDE was not used at all. Chlordane, one of the

Table 15. Summary of pesticide detections and concentrations in bed sediments from streams, estuaries, and lakes in the Connecticut, Housatonic, and Thames Rivers study unit from the U.S. Geological Survey's National Water Information System (NWIS) data base, water years 1973-91

[Reporting level is the least concentration detectable by analytical methods used during the period of record. **USEPA draft sediment-quality criteria:** Derived from Final Residue Value, contaminant concentration in water that will protect wildlife that consume aquatic organisms and will not result in tissue concentrations that exceed U.S. Food and Drug Administration action levels. USEPA, U.S. Environmental Protection Agency; DDD, dichlorodiphenlydichloroethane; DDE, dichlorodiphenyl-dichloroethylene; DDT, dichlorodiphenyltrichloroethane; 2,4-D, (2,4-dichlorophenoxy) acetic acid; 2,4,5-T, (2,4,5-trichloro-phenoxy) acetic acid; µg/kg, microgram per kilogram; <, actual value is less than value shown]

Pesticide (parameter code)	Reporting level (µg/kg)	Number of sites or samples with detections (percentage shown in parentheses)		Range of concentrations detected (µg/kg)	USEPA draft sediment-quality criteria (µg/kg)	USEPA reference
		Sites	Samples			
Organochlorine insecticides						
Aldrin, bottom material (39333)	0.1	1 (1)	1 (<1)	0.6	--	--
Chlordane, bottom material (39351)	1.0	42 (51)	76 (38)	1.0-150	309	1990
DDD, bottom material (39363)	.1	46 (56)	87 (44)	0.1-220	--	--
DDE, bottom material (39368)	.1	36 (44)	52 (26)	0.1-81.0	--	--
DDT, bottom material (39373)	.1	48 (58)	72 (36)	0.1-54.0	828	1988a
Dieldrin, bottom material (39383)	.1	43 (52)	91 (45)	0.1-29.0	9,000	1991c
Heptachlor epoxide, bottom material (39423)	.1	1 (1)	1 (<1)	0.8	--	--
Chlorophenoxy-acid herbicides						
2,4-D, bottom material (39731)	.1	2 (4)	2 (2)	0.1-0.2	--	--
2,4,5-T, bottom material (39741)	.1	3 (6)	3 (3)	0.4-13.0	--	--
Semi-volatile organic compounds						
Acenaphthylene, bottom material (34203)	200	2 (14)	2 (12)	210-540	--	--

Table 16. Summary of pesticide detections and concentrations in bed sediments from streams, estuaries, and lakes in the Connecticut, Housatonic, and Thames Rivers study unit from the U.S. Environmental Protection Agency's Storage and Retrieval (STORET) data base

[2,4-D, 2,4-dichlorophenoxyacetic acid; 2,4,5-T, trichlorophenoxyacetic acid; DDD, dichlorodiphenyldichloroethane; DDE, dichlorodiphenyldichloroethylene; DDT, dichlorodiphenyltrichloroethane; µg/kg, microgram per kilogram; <, actual value is less than value shown]

Pesticide (parameter code)	Period of record (water year)	Number of samples	Number of samples greater than reporting level	Range of concentrations detected (µg/kg)
Organochlorine insecticides				
Aldrin, bottom material (39333)	1973-79	4	2	0.2 - <0.06
DDD, bottom material (39363)	1979-80	3	1	0.8
DDE, bottom material (39368)	1979-80	3	1	0.2
DDT, bottom material (39373)	1979-80	3	1	0.1
Heptachlor, bottom material (39413)	1973-79	3	1	0.1
Heptachlor epoxide, bottom material (39423)	1974-79	3	1	5.3
Organophosphorus insecticides				
Methyl parathion, bottom material (39601)	1973	2	1	0.1
Chlorophenoxy-acid herbicides				
2,4-D, bottom material (39731)	1974	1	1	0.05
2,4,5-T, bottom material (39741)	1974	1	1	4.3

most persistent of the organochlorine insecticides, also was one of the most frequently detected organochlorine insecticides in bed sediments, even though its reporting level was higher than those of other compounds detected.

Other than organochlorine compounds, there were a few detects of the chlorophenoxy-acid herbicides 2,4-D and 2,4,5-T in bed-sediment samples from NWIS and STORET streamflow-gaging stations, and a few organophosphorus insecticide detections. Acenaphthylene, a coal-tar derivative with some insecticide and fungicide usage, was detected in bed sediment at two NWIS stations.

Pesticide data for surface water are inadequate to accurately and completely describe the occurrence and distribution of pesticides in the study unit. For much of the study unit, including parts of Massachusetts, New Hampshire, New York, Rhode Island, and Vermont, little or no pesticide data exist for surface water or bed sediments. For some of the study unit, notably the State of Connecticut, the historical data coverage is relatively complete and fairly extensive. However, even in Connecticut, data are collected primarily at stations on large, high-order rivers and their tributaries. These stations commonly are downstream from point-source dischargers, such as sewage-treatment plants or industrial discharges, and generally represent drainage areas that are hundreds of square miles in size with heterogeneous land uses. Data from stations located downstream from point-source dischargers can not be readily used to interpret water-quality effects related to specific land use or human activities in their drainage basins. Furthermore, the pesticide information for these stations generally dates to the early to mid-1970's, with essentially no data available for the last decade.

Although some 57 different pesticide compounds or metabolites were analyzed in surface-water or bed-sediment samples, many of the compounds used extensively in the study unit were never included in pesticide analyses. Other pesticides with unreported or under-reported use may be present in the rivers, sediment, or biota of the study unit as well.

Ground Water

More information is available in the study unit on the occurrence and distribution of pesticides in ground water than in surface water. A number of previous studies by the USGS, State agencies, or consultants have documented the occurrence of various pesticides in surficial and bedrock aquifers in the study unit. Data on pesticides in ground water collected during several of these studies are in the NWIS data base, and these data are sufficient to synthesize new information or reaffirm relations formerly identified between pesticide occurrence and human activities or hydrogeologic factors.

Previous Studies

Connecticut

Before 1983, few instances of ground-water contamination by pesticides were reported in the study unit. Miller and others (1974, p. 283) attributed thallium contamination of wells in the sedimentary bedrock aquifer in Middlefield, Conn., in 1966 to applications of thallium sulphate as an insecticide and rodenticide.

In 1983, the soil fumigant ethylene dibromide (EDB), long used on tobacco fields in north-central Connecticut, was discovered in water from domestic and public-supply wells in 10 Connecticut towns (Frink and Hankin, 1986). By 1986, the State had tested water samples from more than 2,000 domestic and public-supply wells in a 50-square-mile area of north-central Connecticut and EDB was detected in samples from 268 domestic and 54 public-supply wells at concentrations that equaled or exceeded the State's 0.1 mg/L drinking-water standard (Melvin and others, 1988, p. 195). Through the end of 1989, the State had analyzed 3,928 samples from 2,246 domestic wells and 840 samples from 278 public-supply wells in 20 towns in north-central Connecticut (fig. 39). EDB was detected in 575 of the wells (23 percent) at concentrations ranging from 0.02 to 25 µg/L (table 17). Groundwater samples from two-thirds of the wells where EDB was detected, including 330 domestic wells and 56 public-supply wells, contained EDB concentrations equal to or greater than the 0.1 mg/L State drinking-water standard (Connecticut Department of Health Services, written commun., 1992). The largest number of affected wells and the highest concentrations were in the towns of Somers, East Windsor, and Ellington where cultivation of shade tobacco was extensive; EDB also was detected in 12 other Connecticut towns. The CTDOHS continues to monitor public-supply wells in the State for EDB and some additional data are collected annually for their EDB survey.

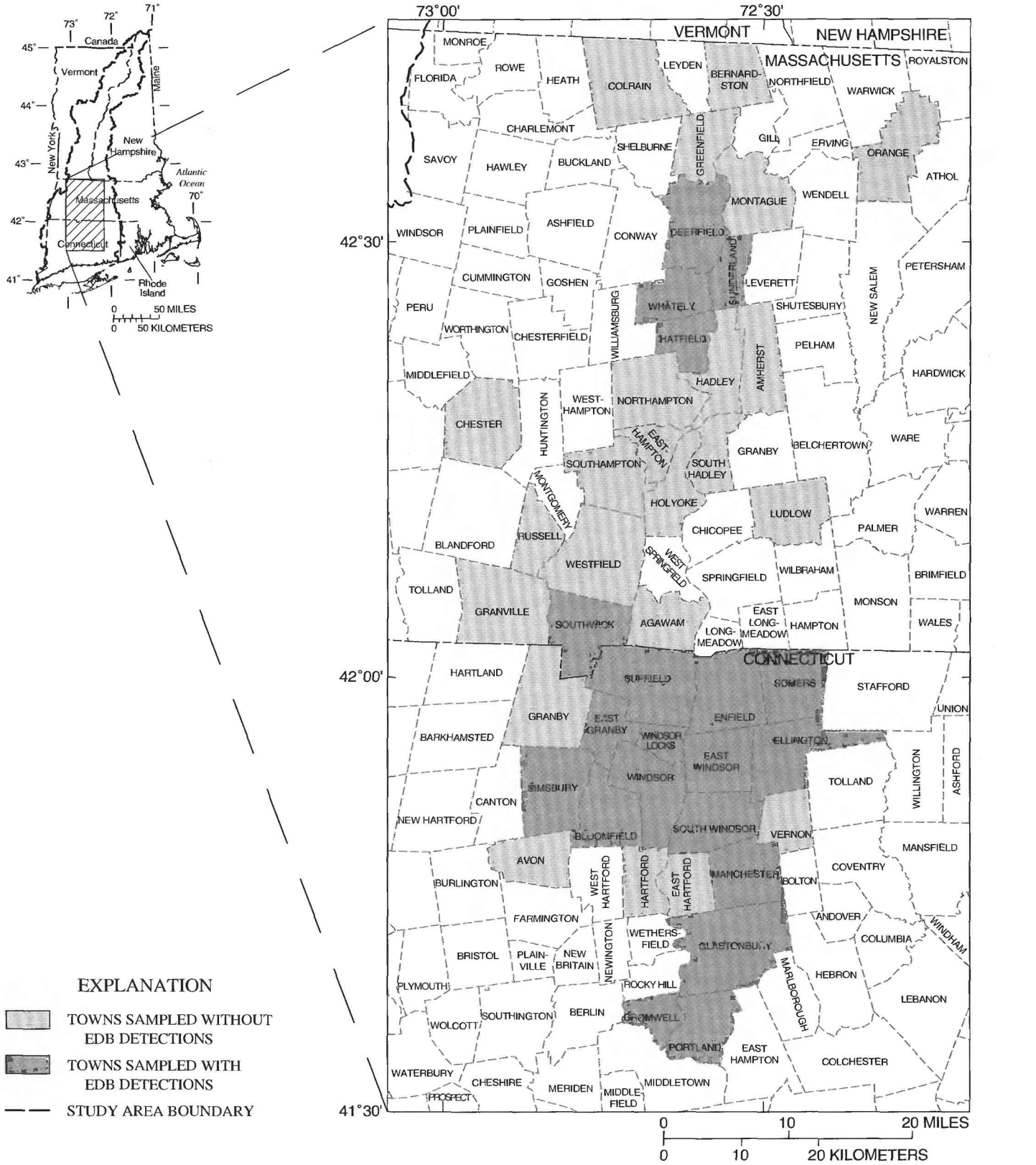


Figure 39. *Towns in present and former tobacco-growing areas of* Connecticut and Massachusetts where wells have been sampled and EDB has been detected in ground water, 1983-86. (Data from Connecticut Department of Health Services, written commun., 1992; Massachusetts Interagency Pesticide Task Force, 1986; Stone and Webster Engineering Corp., 1988.)

Table 17. Summary of ethylene dibromide detections and concentrations in ground-water samples from wells in the Connecticut River Valley and adjacent areas of Connecticut and Massachusetts

[Data for Connecticut from Connecticut Department of Health Services (written commun., 1992). Sampling conducted from March 1984 to December 1989. Data for Massachusetts from Massachusetts Department of Environmental Quality and Engineering (Massachusetts Interagency Pesticide Task Force, 1986; Stone and Webster Engineering Corp., 1988). Sampling conducted from 1983 to 1986. µg/L, microgram per liter; --, no data]

Town	Number of wells sampled	Number of wells with detections	Range of concentrations detected (µg/L)
Connecticut			
Avon	17	0	--
Bloomfield	22	2	0.03
Cromwell	14	2	0.02-0.09
East Granby	219	4	0.05-0.15
East Hartford	10	0	--
East Windsor	383	165	0.01-7.1
Ellington	134	51	0.02-25
Enfield	130	39	0.01-2.1
Glastonbury	64	8	0.02-1.88
Granby	88	0	--
Hartford	2	0	--
Manchester	44	3	0.04-0.41
Portland	29	2	0.03-0.07
Simsbury	276	69	0.02-1.6
Somers	327	170	0.02-2.8
South Windsor	248	39	0.03-1.2
Suffield	249	13	0.01-2.2
Vernon	15	0	--
Windsor	234	6	0.04-0.14
Windsor Locks	19	2	0.02-0.57

Town	Number of wells sampled	Number of wells with detections	Range of concentrations detected (µg/L)
Massachusetts			
Agawam	9	0	--
Amherst	2	0	--
Bernardston	2	0	--
Chester	2	0	--
Colrain	1	0	--
Deerfield	5	2	0.15-7.40
Easthampton	4	0	--
Granville	1	0	--
Greenfield	5	0	--
Hadley	5	0	--
Hatfield	12	1	0.16
Holyoke	2	0	--
Ludlow	1	0	--
Montague	6	0	--
Northampton	2	0	--
Orange	2	0	--
Russell	2	0	--
South Hadley	2	0	--
Southampton	3	0	--
Southwick	26	9	0.05-4.80
Sunderland	22	4	0.08-0.60
Westfield	9	0	--
Whately	132	43	0.02-6.9

In 1985, the Connecticut Agricultural Experiment Station analyzed ground-water samples from 95 public-supply wells (or adjacent surface-water bodies subject to induced infiltration) for a suite of 32 pesticide compounds (Frink and Hankin, 1986). Public-supply wells were selected from the 49 largest community water utilities that rely at least in part on ground-water sources. Wells were selected to include locations where agricultural, urban, and recreational (golf-courses) land uses were present in the recharge areas. The only pesticide detected was the soil fumigant 1,2-dichloropropane. This pesticide was detected in samples from four wells in Cheshire, Conn., at concentrations less than the CTDOHS action level of 1.0 mg/L (Connecticut Department of Health Services, written commun., 1984).

More recently, the USGS conducted studies in Connecticut to assess the occurrence and distribution of a large number of pesticide compounds in ground water. Grady and Weaver (1988, 1989) sampled shallow wells in stratified-drift aquifers in Connecticut to statistically relate water-quality conditions to common land uses. With only a few pesticide samples (13) and limited analytical coverage (26 compounds), Grady and Weaver (1988, p. 31) reported the detections of three compounds—2,4-D, 2,4,5-T, and silvex—in one well and diazinon in a second well in residential areas. Pesticides were not detected in the six agricultural-area wells sampled. In 1989, after expanding the pesticide sampling to 80 wells and including triazine herbicides in the analyses, Grady and Weaver (1989, p. 25-26) reported atrazine detections in

13 percent of the agricultural-area wells at concentrations of 0.1 to 0.2 mg/L. The frequency of atrazine detections was significantly higher in ground-water samples from agricultural areas than in samples from undeveloped or residential areas ($p = 0.0193$). In the final report on these studies, Grady (1993) reported that atrazine was detected in one-third of the 15 wells installed in tilled agricultural fields (fig. 40). The tilled agricultural land was used almost exclusively for silage corn production and atrazine was a commonly applied pre-emergent broadleaf herbicide. Although no other pesticide compound was detected in sufficient numbers of wells to relate its occurrence statistically to any land use (at the 0.05 level of significance), 19 additional pesticides (or other synthetic organic compounds with some pesticide usage) were detected in one or more

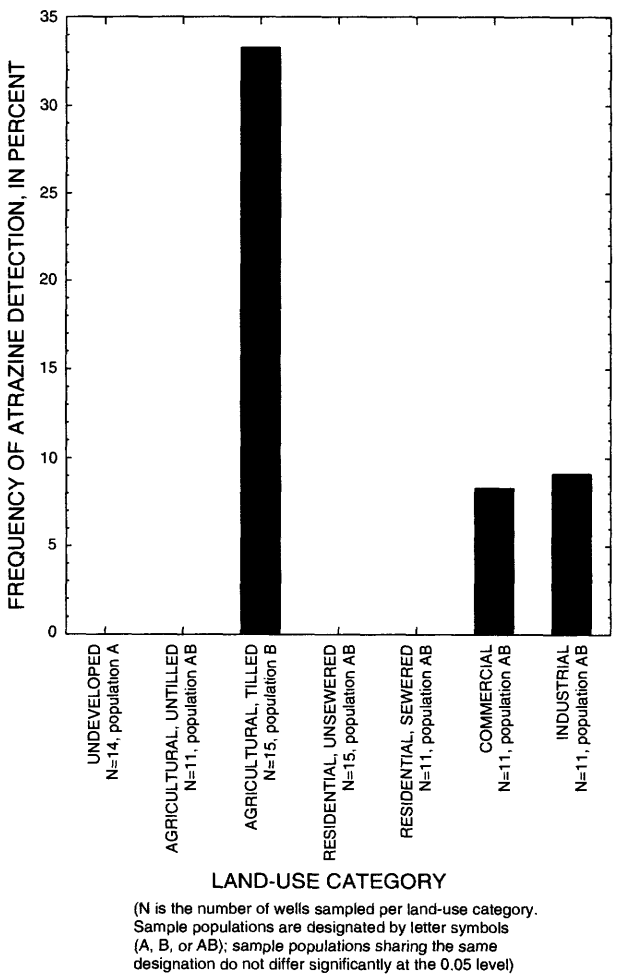


Figure 40. Frequency of detection of atrazine in wells completed in stratified-drift aquifers underlying seven land-use areas in Connecticut, 1985-89 (from Grady, 1993).

wells from seven land-use categories (Grady, 1993, p.18-20). Forty percent of the pesticide detections were in ground-water samples from wells in residential areas. Compounds detected in more than 5 percent of the wells sampled include the herbicide 2,4-D, the organochlorine insecticide dieldrin, and two volatile organic compounds used as insecticide fumigants: 1,2-dichloropropane and 1,1-dichloroethane. Other compounds detected less frequently included chlordane, diazinon, DDD, DDE, 2,4-DP, 2,4,5-T, cyanazine, phorate, prometone, propazine, silvex, simazine, acenaphthene, 1,2-dichloroethane, 1,1-dichloroethylene, diethyl phthalate, and naphthalene.

In a similar study of the occurrence of pesticides in shallow ground water in Connecticut, Mullaney and others (1991) sampled 89 wells installed at 59 agricultural and nonagricultural sites for pesticides known or inferred to have been applied. Agricultural sites included corn, potato, mixed vegetable, nursery, and orchard production. Nonagricultural sites included golfcourses, recreational areas, and residential and commercial land-use areas. Wells were installed in stratified-drift, till, and shallow bedrock aquifers, and core samples of soils and sediments from the unsaturated zone were collected. Twenty-four pesticides were detected in one or more of the wells. Atrazine, the most commonly detected pesticide in the study, was detected in 25 wells at 76 percent of the agricultural sites, at concentrations ranging from 0.1 to 9.7 mg/L (Mullaney and others, 1991, p. 12-13). The herbicide dimethyl tetrachloroterephthalate (DCPA) was the most commonly detected pesticide at nonagricultural sites, reported at concentrations ranging from 0.01 to 124 mg/L in 15 wells at 73 percent of those wells. Pesticides appeared to be present in ground water beneath agricultural settings regardless of underlying aquifer type (stratified drift or till-mantled bedrock). By contrast, pesticides in nonagricultural areas were detected in fewer till and bedrock wells than in wells in stratified-drift aquifers. At 21 of the 32 agricultural sites, multilevel sampling was conducted to determine the vertical distribution of pesticides in the aquifers. In wells in agricultural areas underlain by stratified drift, Mullaney and others (1991, p. 14) found that the preponderance of pesticide detections at shallow depths (less than 15 ft below the water table) was statistically significant (fig. 41).

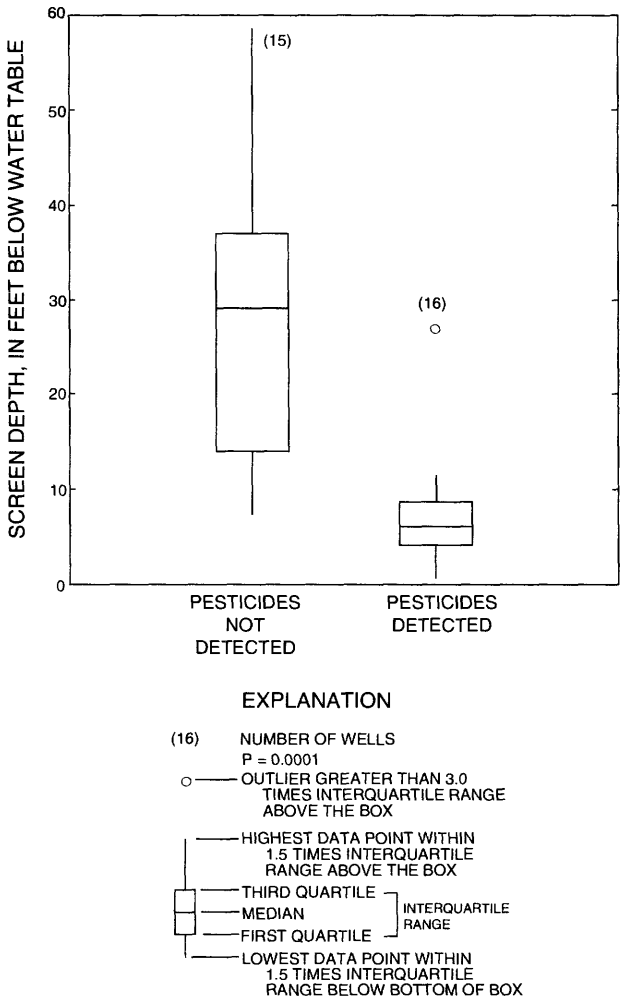


Figure 41. Pesticides in ground water by screened depth of wells in stratified-drift aquifers in agricultural areas of Connecticut, 1987-89 (from Mullaney and others, 1991).

Massachusetts

In response to incidents of ground-water contamination resulting from the use of aldicarb and EDB in potato- and tobacco-producing areas of New York and Connecticut, respectively, Massachusetts initiated monitoring for aldicarb in 1983, and for EDB in 1984 (Ellis, 1986). The Massachusetts Interagency Pesticide Task Force (MIPTF) expanded its pesticide sampling in 1985 to cover six additional compounds: carbofuran, oxamyl, dinoseb, alachlor, 1,2-dichloropropane and 1,3-dichloropropene. The MIPTF had sampled 341 sites in 27 western Massachusetts communities that are in the study unit through 1986; 1,2-dichloropropane, aldicarb, and EDB were the most frequently detected pesticides (Massachusetts Interagency Pesticide Task Force, 1986).

In 1986, the Massachusetts Department of Environmental Quality and Engineering (MDEQE) contracted for a study of the occurrence of pesticides in relation to land use for a 20-town area of the Connecticut River Valley encompassing areas of shade tobacco cultivation similar to those in Connecticut, as well as potato and vegetable production (Stone and Webster Engineering Corporation, 1988). Using the data collected by the MIPTF during 1984-86, the MDEQE-sponsored study reported one or more of seven pesticide compounds (all the above except 1,3-dichloropropane) in water samples from 146 of 358 sites. Most of the pesticide detections were concentrated in the lowland areas of the Connecticut River Valley where agricultural activities were heaviest (Stone and Webster Engineering Corporation, 1988)—the towns of Deerfield, Hatfield, Southwick, Sunderland, and Whately, Mass. Detections of specific pesticide compounds were strongly correlated to crop type and pesticide use on fields adjacent to contaminated sites. Eighty-eight percent of the 358 sites sampled in the Massachusetts study were wells, including 248 domestic and 66 public-supply wells; 44 surface-water sources (reservoirs) also were sampled. Contamination generally was found in "shallow wells...[although] some deep wells showed contamination" (Stone and Webster Engineering Corporation, 1988). Site-specific information on aquifer type, well depth, or depth to the water table was generally unreported for the wells sampled.

Data on EDB in ground-water samples were available for 23 Massachusetts towns that generally lie within or adjacent to the Connecticut River Valley (fig. 39; table 17). Although EDB was detected in only 5 of the 23 towns, all but 4 towns had fewer than 10 wells sampled. More than one-half (51 percent) of the wells sampled for EDB were in the town of Whately, Mass. Comparing the results for Massachusetts with those reported for Connecticut shows that, although many more wells were sampled in Connecticut than Massachusetts, the frequency of EDB detections was similar (about 23 percent), as was the range of EDB concentrations reported (0.02 to 7.40 mg/L).

Information on the occurrence of 1,2-dichloropropane and aldicarb in ground water for the 27 towns in or near the Connecticut River Valley in Massachusetts also was available (Massachusetts Interagency Pesticide Task Force, 1986; Stone and Webster Engineering Corporation, 1988). A nematocide used on tobacco and strawberry fields, 1,2-dichloropropane was

the most frequently detected pesticide, reported in 36 percent of the wells sampled in 12 of the 27 towns (Massachusetts Interagency Task Force, 1986). Most of the detections (61 percent), however, were in Whately, Mass., where samples were far more numerous (129 wells) than in other towns. Concentrations of 1,2-dichloropropane in ground water ranged from 0.03 to 51 mg/L, with nearly 40 percent of the detections exceeding the State's 1.0 mg/L drinking-water standard (Massachusetts Interagency Pesticide Task Force, 1986). Aldicarb, a carbamate pesticide widely used to control soil insect pests in potato fields, was detected in 30 percent of the wells from 9 of the 27 towns. Concentrations of aldicarb in ground-water samples from Massachusetts ranged from 1.0 to 34 mg/L, with about one-third of the detections exceeding the 10 mg/L State drinking-water standard.

Detections of other carbamate pesticides in samples from wells in Massachusetts were much less common—11 percent for carbofuran and less than 1 percent for oxamyl—and similarly, most occurred in Whately. Dinoseb, a phenol herbicide used to kill potato vines to facilitate harvest, and alachlor, a broadleaf herbicide commonly used on corn fields, were detected in 7 and 1.5 percent, respectively, of the wells sampled; all detections were in Whately. There were no detections of 1,3-dichloropropene in any of the wells (Massachusetts Interagency Pesticide Task Force, 1986).

Vermont

The ongoing Vermont Pesticide Monitoring Program began in 1986, with the goal of providing Vermont's citizens, agricultural community, and State agencies with information on the extent and patterns of pesticide use and on the environmental fate of pesticides in the State. A major component of this program was sampling wells and other sources of drinking water in or near areas of pesticide application (Vermont Department of Agriculture, 1988). Vermont's monitoring focused on pesticides used on corn, fruits and vegetables, and Christmas trees; three separate projects investigated each of these crop categories.

The corn herbicide survey project focused on existing private and public drinking-water wells adjacent to cropland (within 500 ft) used for silage and sweet corn production (Vermont Department of Agriculture, 1988). Samples collected in 1986-87 were analyzed for five herbicides applied to corn fields—alachlor, atrazine, cyanazine, metolachlor, and simazine; in 1988, analyses for carbamates (aldicarb sulfoxide and aldicarb sulfone) and pendimethalin were added to the program. Between April 1986 and February 1993, 967 samples were collected from 491 wells and other sampling locations statewide (J.G. Comstock, Vermont Department of Agriculture, written commun., March 1993). The sampling sites included 305 drilled wells, 75 dug wells, 27 driven points, 68 springs, 6 streamflow-gaging stations, and 10 wells of unknown construction. Of these, only 137 were in the Connecticut River Basin and, hence, in the study unit. Overall, 6 percent of the sites sampled had detections of one or more of the five corn herbicides; atrazine was the most frequently detected, followed by metolachlor. Herbicide concentrations ranged from 0.1 to 24.3 mg/L. However, 43 percent of the detections were in samples from shallow ground-water sources—springs, dug wells, and drive-point wells—which collectively comprised only 35 percent of the sites. Of the 137 sites tested in the Connecticut River Basin, 10 percent produced water samples with detectable concentrations of pesticides, but information on the type of wells was unavailable.

Fewer data were collected by 1993 for Vermont's fruit and vegetable and Christmas tree pesticide surveys than for the corn herbicide project. Through February 1993, the fruit and vegetable survey had collected 52 samples from 19 wells statewide (J.G. Comstock, Vermont Department of Agriculture, written commun., March 1993). The fruit and vegetable survey samples were analyzed for DCPA (dacthal) plus metabolites, napropamide, pendimethalin, and terbacil. Thirty-two percent of the samples tested positive for DCPA or its metabolites; one sample also contained terbacil. Three of the nine fruit and vegetable survey wells that occur in the Connecticut River Basin were among the wells that tested positive for DCPA or its metabolites. Through February 1993, 34 samples were collected from 23 Christmas tree survey wells statewide—only nine of the Christmas tree survey wells sampled were in the study unit. Samples from Christmas tree survey wells were analyzed for herbicides and carbamate insecticides, including those pesticides previously identified as among the corn herbicide and fruit and vegetable herbicide surveys. In the Christmas tree survey, only one of nine wells sampled in the study unit tested positive for any herbicide. This well, the only one of 23 wells in the state with a detection in this category, tested positive for atrazine and cyanazine, but not for any of the fruit and vegetable herbicides.

In addition to sampling existing wells, the State of Vermont installed 23 monitoring wells during 1988 and 1989 on three farms, one of which was in the study unit. According to the Vermont Department of Agriculture, these were primarily shallow wells with depths ranging from 11 to 50 ft and were sampled quarterly. Water samples were analyzed for the same herbicides as the wells in the corn survey project. Corn herbicides were detected in 52.2 percent of the wells in this study. On the one farm that is in the NAWQA study unit, pesticides were detected in five of nine wells. Atrazine, cyanazine, and metolachlor, the only herbicides detected, were in 35, 9, and 35 percent, respectively, of all wells sampled in this three-farm study (J.G. Comstock, Vermont Department of Agriculture, oral commun., March 1993).

New Hampshire

The State of New Hampshire coordinated a multi-agency study in 1986 to collect and analyze ground-water samples from existing wells near agricultural lands and golfcourses (New Hampshire Division of Public Health Services, 1986). Twenty-five shallow wells (25 ft deep or less) located on or directly downgradient from farms cultivating potatoes, corn, mixed vegetables, berries and ornamentals or from golfcourses with well-documented pesticide use were selected in 15 towns. Information on well location was unavailable, but county designations suggest that no more than five of the wells may be in the study unit. Wells were sampled for as many as 18 commonly used pesticides during July, August, and September 1986. The analytes, selected based on usage and mobility in ground water, included alachlor, aldicarb, atrazine, carbofuran, chlordane, chlorpyrifos, dacthal, diazinon, dicamba, dieldrin, dinoseb, disyston, endosulfan, lindane, methoxychlor, oxamyl, simazine, and 2,4-D. Pesticides were not detected in samples from any of the 25 wells at laboratory reporting levels, which ranged from 1.0 to 2.5 mg/L.

Analysis of Information

In the NWIS data base, a total of 34 pesticide compounds were detected in one or more samples from one or more wells in the study unit (table 30, at back of report); no ground-water pesticide data from STORET were used in this analysis. All of the NWIS wells in the study unit where pesticides have been detected are in Connecticut (fig. 25); few NWIS wells outside of Connecticut have had any pesticide sampling. The few wells in New Hampshire were sampled only for volatile organic compounds and did not provide a satisfactory data base for comparison. However, no pesticides were detected in any of the limited number of NWIS springs sampled in the study unit (see pl. 1).

Of the pesticide compounds sampled for in a large number of wells (80 or more), only one compound, atrazine (present in 22 percent of wells sampled), was detected in water from more than 10 percent of the wells. Six other pesticides, simazine, metolachlor, 1,2-dichloropropane, 1,1-dichloroethane, dieldrin, and 2,4-D, were detected in 5 percent or more of the wells. Three compounds, DCPA, diuron, and terbacil, were reported at unusually high detection frequencies (43 to 100 percent of wells sampled), but were only included as target analytes in samples from a small number of wells (see table 24). The remaining 24 compounds identified in ground-water samples from NWIS wells in the study unit (table 30) were detected in only one to five of the wells sampled.

Evaluation of the relations between pesticide occurrence or concentration and hydrogeologic factors (aquifer type) or human activities (land use or crop type) was limited using the NWIS data. Not all wells were sampled for all pesticides and different detection limits were reported for some pesticide compounds. A variety of well-construction characteristics and sampling protocols, which could have influenced the detection of some compounds present at low concentration, were used. Comparisons were not valid because too few wells were sampled in some aquifers and some land-use or crop types. Still, some useful information can be provided by presenting such comparisons, tempered by realization of the shortcomings in the data.

Pesticides were detected in NWIS wells in every sampled aquifer in the study unit (fig. 42; see also fig. 16). Nearly one-third of the wells in stratified drift, the most intensively sampled aquifer, contained one or more pesticide compound. This frequency, however, was biased because most of the wells were installed in areas of known or inferred pesticide use. The same bias was reflected to a greater degree in till and bedrock aquifer detections because the small sample number included few or no wells in undeveloped, relatively natural areas. Additional data are needed for these aquifers, particularly for carbonate bedrock aquifers where no wells were sampled for pesticides, to accurately assess the relation between pesticide occurrence and aquifer type.

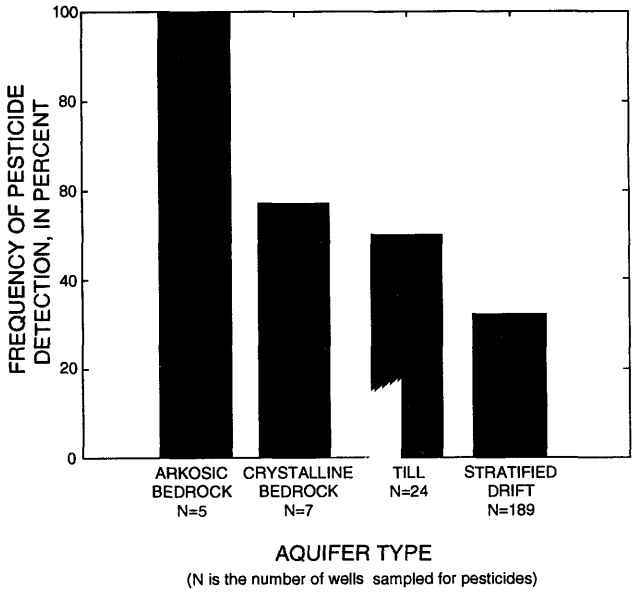


Figure 42. Frequency of detection of pesticides by aquifer type for wells in Connecticut, 1978-89. (Data from U.S. Geological Survey's National Water Information System data base.)

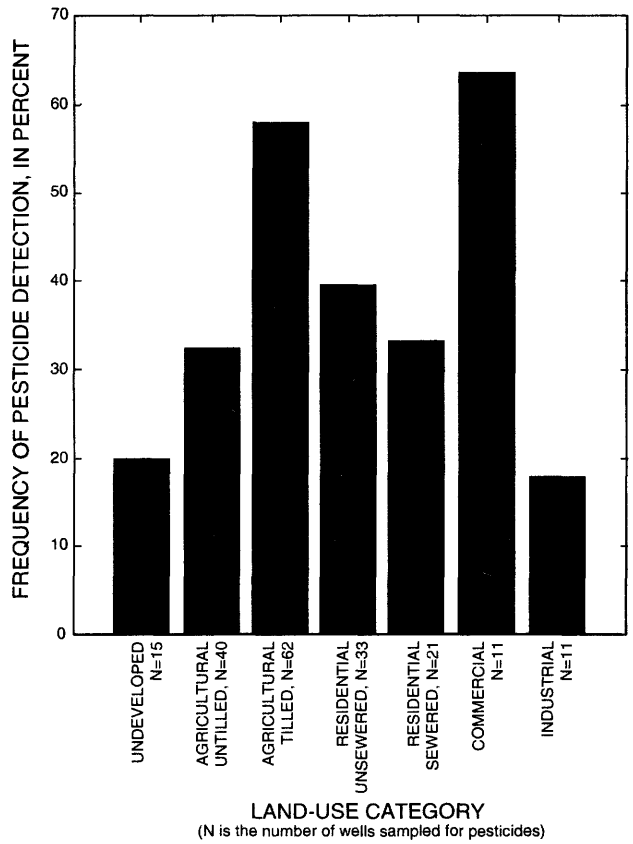


Figure 43. Frequency of detection of pesticides by land use for wells in Connecticut, 1978-89. (Data from U.S. Geological Survey's National Water Information System data base.)

Pesticides also were detected in ground-water samples from NWIS wells in seven land-use categories examined in the study unit (fig. 43). The high frequency of pesticide detections in urban areas may have reflected a great variety and heavy application rate of pesticides by commercial applicators, businesses, and homeowners. The high frequency of pesticide detections in ground water in tilled agricultural areas was not unexpected, considering the intensity of agricultural use of pesticides and the favorable hydrogeologic conditions (shallow water tables, thin permeable soils, and porous, unconsolidated or fractured bedrock aquifers near the land surface).

Further examination of the frequency of pesticide detections in the 102 wells in untilled and tilled agricultural land-use categories (fig. 43) indicated that frequencies of pesticide detections was high for vegetables, orchards, and nurseries (fig. 44). Although intensive pesticide usage may be associated with these agricultural practices, the number of wells sampled was far too small to provide a valid comparison among crop types. Pesticide detections in more than one-half of the wells sampling ground water in corn fields reflected the widespread use of herbicides, particularly atrazine, simazine, and metolachlor, on the principal crop in the study unit. Atrazine was present in more

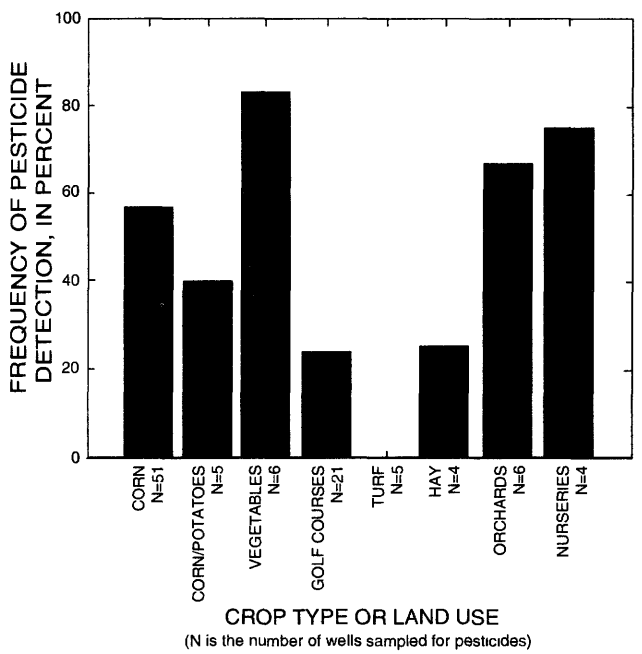


Figure 44. Frequency of detection of pesticides by agricultural crop type and related land use for wells in Connecticut, 1978-89. (Data from U.S. Geological Survey's National Water Information System data base.)

than 50 percent of the wells in corn fields (fig. 45) and also in ground water in areas of mixed corn and potatoes, or in areas where corn may have been grown in prior years.

Atrazine was the most frequently detected pesticide compound in ground-water samples from NWIS wells in the study unit. Reported in 32 of the 143 wells sampled for this compound, atrazine was present at concentrations equal to or greater than 0.1 µg/L in 24 percent of the NWIS ground-water samples collected from all aquifers. However, only one well yielded water with an atrazine concentration (9.7 µg/L) exceeding the USEPA's 3.0 µg/L MCL for drinking-water supplies. Although all the wells sampled for atrazine were monitoring wells, the presence of concentrations exceeding the MCL in productive aquifers is a cause for concern.

The NWIS data collected to date indicate that atrazine detections were limited to the shallowest part of the ground-water resource. All the wells with atrazine detections were less than 40 ft deep, regardless of aquifer. In the stratified-drift aquifers where most of the atrazine was detected, the depth of the screened (sampled) interval below the water table was significantly different in wells in which atrazine was detected than in those wells without atrazine detections (fig. 46). Atrazine detections were limited to the top 15 ft of the saturated zone in stratified-drift aquifers.

In addition to atrazine, three other pesticides were detected in ground water at concentrations exceeding USEPA MCL's for drinking water—alachlor in one well at a concentration of 11 µg/L, 1,2-dibromoethylene (EDB) in two wells at concentrations of 0.2 and 0.5 µg/L, and simazine in one well at concentrations of 9.1 to 10 µg/L. The high EDB concentrations also exceeded the Connecticut drinking-water guidelines, as do the 41 µg/L of 1,2-dichloropropane in water from one well and the detection of 1,2-dichloroethane at concentrations of 2.4

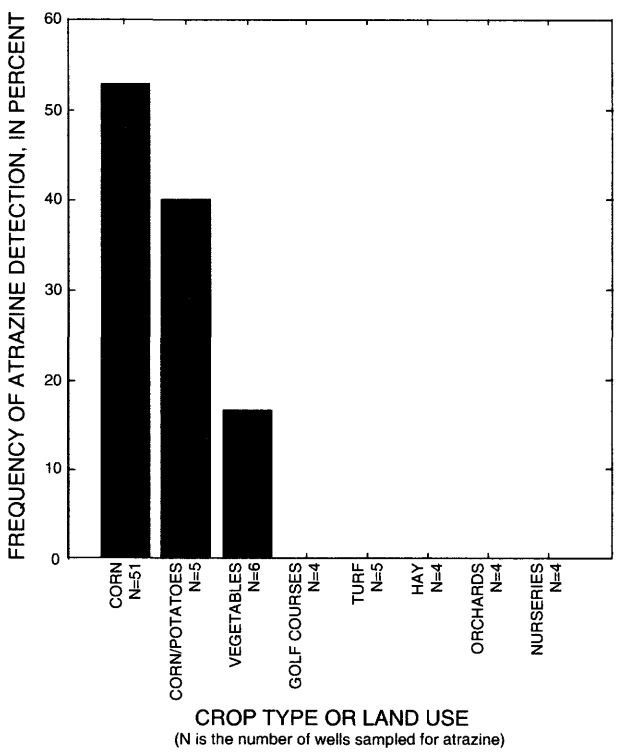


Figure 45. Frequency of detection of atrazine by agricultural crop type and related land use for wells in Connecticut, 1978-89.(Data from U.S. Geological Survey's National Water Information System data base.)

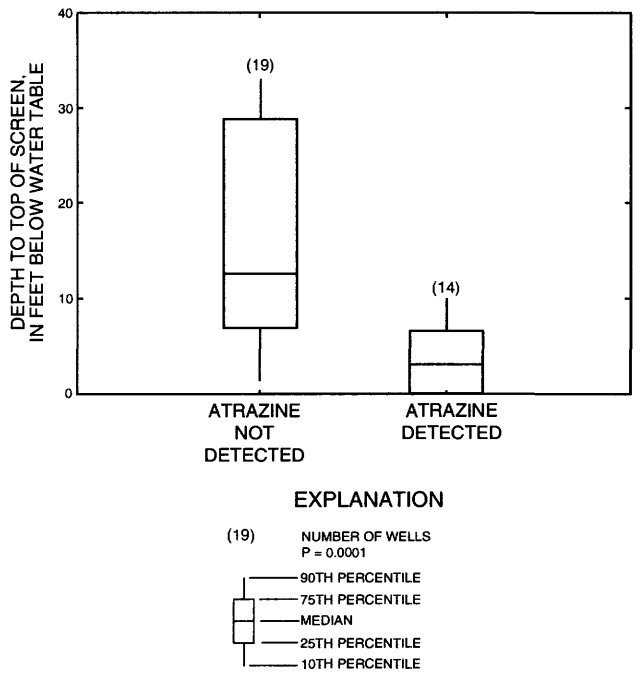


Figure 46. Detections of atrazine in stratified-drift aquifer by depth to top of well screen below the water table in Connecticut, 1978-89. (Data from U.S. Geological Survey's National Water Information System data base.)

to 4.8 µg/L in four wells. However, all of the above pesticide concentrations that exceeded Federal or State drinking-water MCL's, as well as all other pesticide detections in the study unit from NWIS wells, were in samples from observation wells. Many of these observation wells were designed to maximize the likelihood of detecting pesticides in ground water. They were shallow small-diameter wells with a short screened-interval installed within or just downgradient from areas of known or inferred pesticide application. As such, these wells provided a biased assessment of the extent to which pesticides have degraded the available ground-water resource. The sampling sites did not accurately indicate the extent to which drinking-water supplies may be impaired in the study unit. The observation-well samples did, however, show that pesticides were present in the shallow ground-water and provided information on the association of pesticides with land uses or human activities. With this knowledge, more appropriate monitoring of ground-water supplies may be possible.

As of 1992, the CTDOHS monitoring of public-supply wells confirmed the assessment (Frink and Hankin, 1986) that few incidents of pesticide contamination of water supplies occurred in Connecticut other than the widespread EDB problem. Only two of the CTDOHS-monitored public-supply wells inventoried for this report (see fig. 22) had any pesticide detections (trace levels of dieldrin). However, only four organochlorine insecticides, two chlorophenoxy-acid herbicides, and EDB were routinely analyzed in samples from public-supply wells.

Although 77 different pesticide compounds or metabolites were analyzed in at least some NWIS ground-water samples, nearly two-thirds of the pesticides reportedly used in the study unit (see table 3) were never included in laboratory pesticide analyses of water samples. Additional pesticide compounds without reporting requirements were not listed in table 3, but they were applied widely in the study unit. Detections of nearly one-half (44 percent) of the pesticide compounds for which ground-water samples were analyzed indicated that a substantial number of additional pesticides also may be present in ground water in the study unit. Better information on the use of pesticides by agricultural, commercial, and private homeowner applicators would allow a more comprehensive assessment of pesticide occurrence and distribution in surface and ground water of the study unit.

Review of the previous investigations and analysis of the NWIS data indicated that the occurrence of pesticides in ground water in the study unit was diversified and extensive. Overall, 40 different pesticide compounds or metabolites were reported in ground-water samples collected from three States—Connecticut, Massachusetts, and Vermont—which constituted a major part of the study unit. In parts of the study unit where pesticides were not yet detected in ground water, New Hampshire, in particular, sampling for pesticides was minimal or absent. Even in the three States where sampling was substantial, much of it focused on specific areas or types of agriculture, leaving large areas of untargeted land use and some aquifers unsampled.

Pesticides were detected in the unconsolidated, surficial, glacial aquifers, till and stratified drift, as well as in fractured bedrock aquifers in the study unit. Much of the sampling to date focused on the stratified-drift aquifers, although the bedrock aquifers, particularly the carbonate-bedrock aquifer, were undersampled. The occurrence of pesticides appeared to be limited largely to the shallower parts of the sampled aquifers, but fewer samples were analyzed for pesticides from deeper parts of the ground-water-flow system.

The distribution of pesticides in ground water in the study unit is associated with land use. Although some pesticides were detected in ground-water samples from all major land-use categories, the relation with agricultural land use is best defined. The most widespread occurrence of pesticides in ground water, as well as the most severe contamination of drinking-water supplies, is related to agricultural pesticide use, particularly from the former use of EDB on tobacco-growing regions of central Connecticut and Massachusetts. Urban pesticide use also results in the detection of pesticides in ground water in the study unit, but sampling in urban areas is not nearly as intensive as in agricultural areas. A few pesticides have even been detected in ground water beneath undeveloped areas, however, these areas were sampled the least and data are insufficient to assess the occurrence or the sources of pesticides relative to undeveloped land use.

TRANSPORT OF NUTRIENTS AND SUSPENDED SEDIMENT BY SURFACE WATER AND ATMOSPHERIC DEPOSITION

The movement of nutrients, especially nitrogen, through the study unit and into the waters of Long Island Sound is a major regional concern. State, Federal, and local agencies have gone to great lengths to determine the sources of nutrients that have caused hypoxic conditions during summer months in the western part of the Sound. This chapter addresses questions related to the sources and magnitudes of these nutrient contributions and provides some estimates of the loadings of nitrogen and phosphorus to rivers that discharge into Long Island Sound.

Relation of Nutrient and Suspended-Sediment Concentrations to Streamflow

Concentrations of many water-quality constituents vary with changes in stream discharge. Relations between total phosphorus and discharge

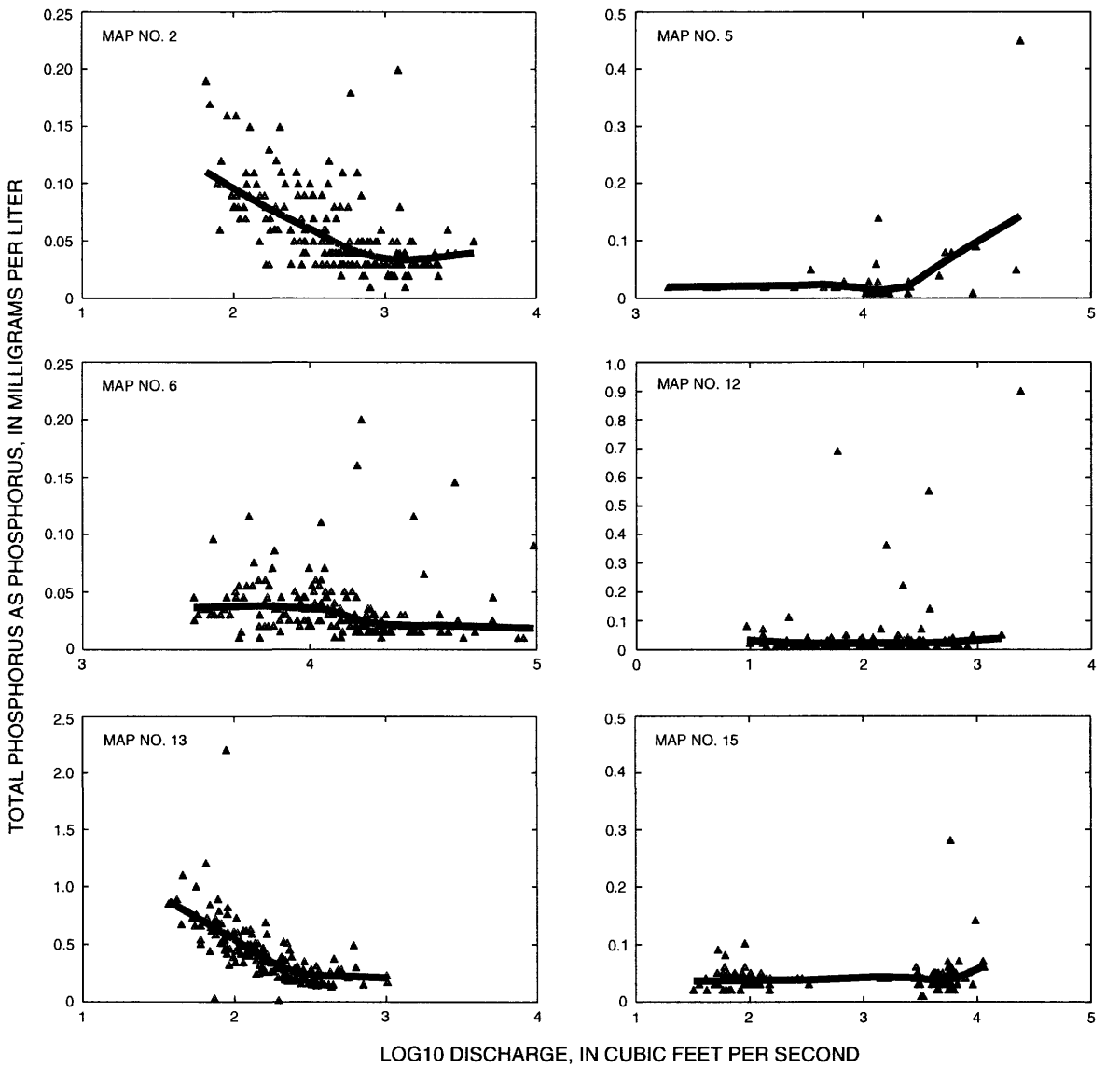
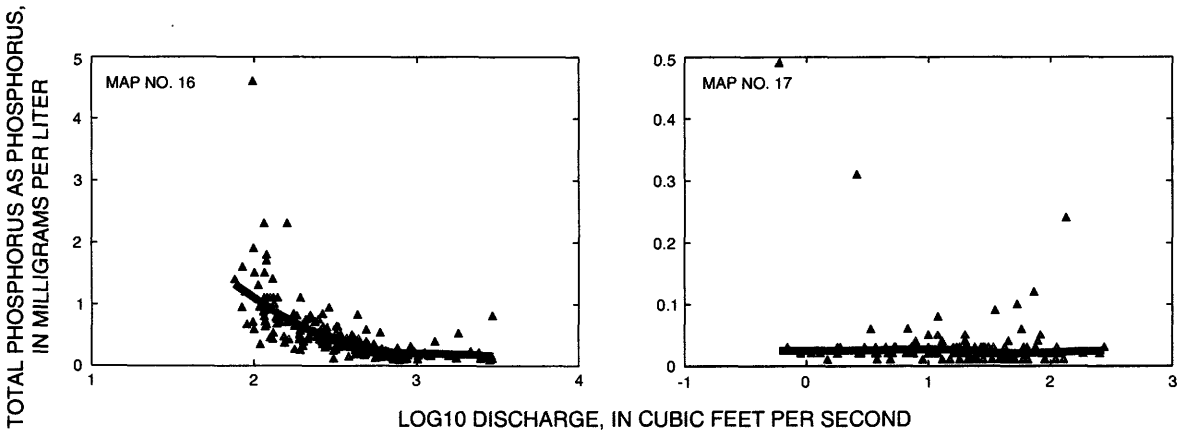


Figure 47. Relation between total phosphorus concentrations and discharge at selected stations in the Connecticut, Housatonic, and Thames River Basins study unit. (Data from U.S. Geological Survey's National Water Information System data base.)



EXPLANATION

(see plate 1 and table 23)

MAP NO.	STATION NAME
2	Shetucket River at South Windham, Conn.
5	Connecticut River at North Walpole, N.H.
6	Connecticut River at Thompsonville, Conn.
12	Salmon River near East Hampton, Conn.
13	Quinnipiac River at Wallingford, Conn.
15	Housatonic River at Stevenson, Conn.
16	Naugatuck River at Beacon Falls, Conn.
17	Saugatuck River near Redding, Conn.

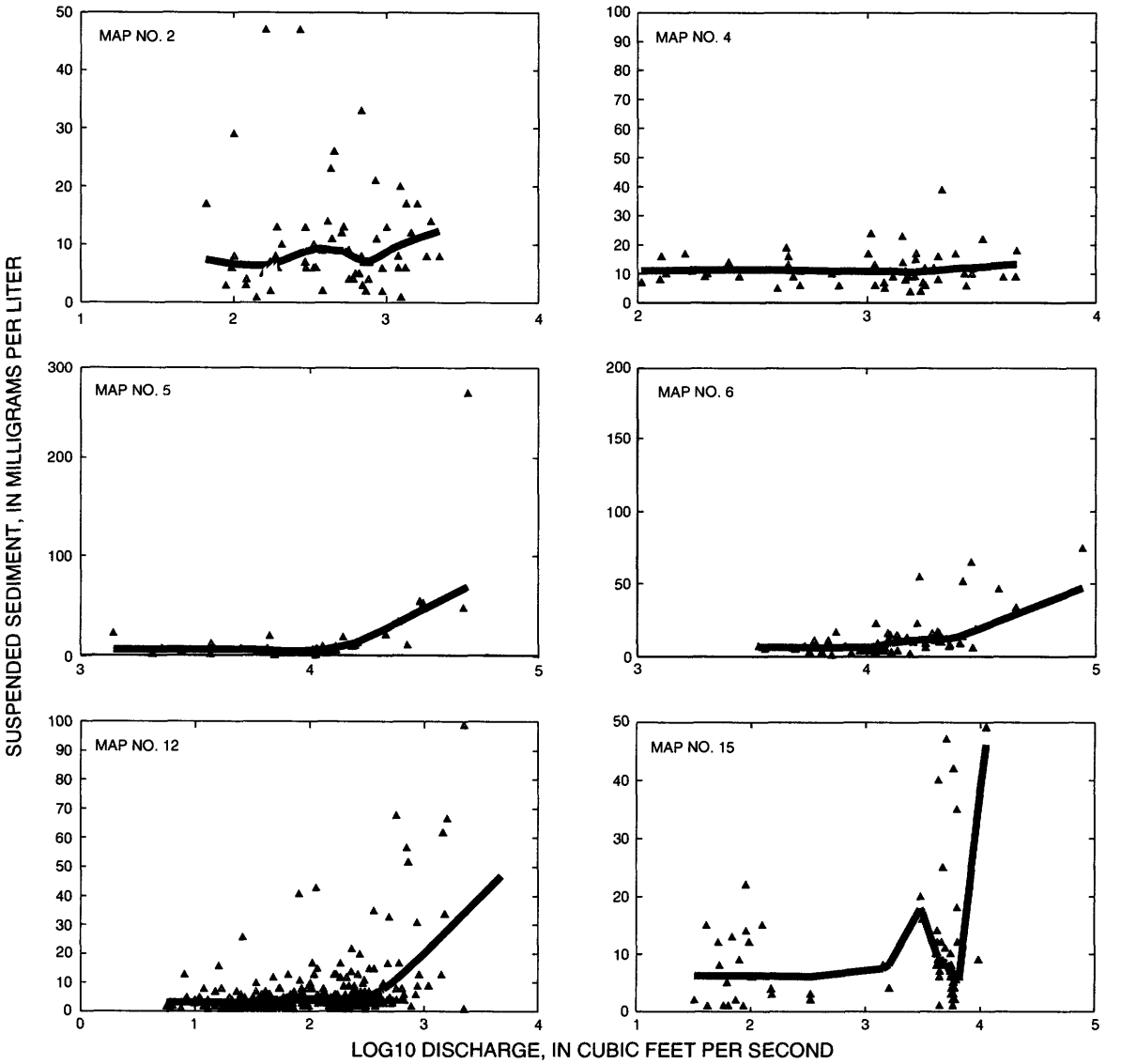
LOWESS SMOOTH

Figure 47.—*Continued.*

(fig. 47) closely resemble those for other phosphorus and nitrogen constituents. High total phosphorus concentrations decrease with increasing streamflow at the Shetucket River at South Windham, Conn., the Quinnipiac River at Wallingford, Conn., and the Naugatuck River at Beacon Falls, Conn.—three stations significantly affected by urban land use; this relation indicates that inputs are relatively constant and increasing discharge tends to dilute them. Concentrations increase at low flows, when point discharges are major influences. Conversely, low concentrations of phosphorus, regardless of streamflow, characterize the water quality of the two stations with sparsely populated drainage basins—the Salmon River near East Hampton, Conn., and the Saugatuck River near Redding, Conn.

Data from three stations, the Connecticut River at North Walpole, N.H., and at Thompsonville, Conn., and the Housatonic River at Stevenson, Conn., indicate the impacts of flow regulation and discharge volume. In spite of the wide range of discharges at these stations, the concentration ranges do not vary greatly. The North Walpole and Stevenson stations are just downstream from hydropower dams. The effects of discharge regulation at Stevenson are most striking, with two distinct groupings of concentration measurements at about 100 and 5,000 ft^3/s, reflecting nongenerating and peak power-generating discharges. Reservoir releases farther upstream affect discharges at Thompsonville and the magnitude of the discharge dilutes any high nutrient concentrations contributed from smaller tributary streams or from point sources.

Suspended-sediment concentrations follow somewhat different patterns with respect to discharge, but stations and data are fewer (fig. 48). Suspended-sediment deposition is not perceived as a major



EXPLANATION

(see plate 1 and table 23)

MAP NO.	STATION NAME
2	Shetucket River at South Windham, Conn.
4	Quinebaug River at Jewett City, Conn.
5	Connecticut River at North Walpole, N.H.
6	Connecticut River at Thompsonville, Conn.
12	Salmon River near East Hampton, Conn.
15	Housatonic River at Stevenson, Conn.

LOWESS SMOOTH

Figure 48. Relation between suspended-sediment concentrations and discharge at selected stations in the Connecticut, Housatonic, and Thames River Basins study unit. (Data from U.S. Geological Survey's National Water Information System data base.)

problem in the study unit. The physiography and agricultural practices are such that sedimentation is only infrequently a local problem. Suspended-sediment concentrations generally increase with increasing streamflow, especially for the high discharges at stations with long-term data. High discharges resuspend sediments that have settled between hydrologic events and carry particles off land surfaces to streams. Most suspended-sediment concentrations for all selected stations range from 10 to 30 mg/L. Deviations from this range, or a greater variability at any given station generally are associated with high discharges. Suspended-sediment concentrations are highest at high discharges, such as those for the Connecticut River at North Walpole, N.H., and Thompsonville, Conn.

Constituent Loads

Annual Loads

Loads of total nitrogen and total phosphorus were calculated for low, high, and median discharge years for six water-quality stations in the study unit (table 18), using the USGS's Estimator computer program (Cohn and others, 1992). Load estimates were based on a simple log-linear concentration model that used functions of two explanatory variables, flow and time.

All estimated loads were significantly and positively related to the log of discharge (coefficient a, table 19) Statistically significant positive quadratic relations with the log of discharge squared (coefficient *b*, table 19) are noted at the Naugatuck River at Beacon

Table 18. Estimated annual loads of total nitrogen and total phosphorus at selected water-quality sampling stations in Connecticut during typical low-, high-, and median-flow water years, based on annual mean discharge

[Map No.: See plate 1 for location of water-quality sampling stations and table 23 for summary of data. Stations are presented in order of increasing drainage area. kg, kilogram; ft^3/s, cubic foot per second; mi^2, square mile]

Map No.	Station name	Drainage area (mi^2)	Calibration period		Low-flow year	Mean discharge (ft^3/s)	Low-flow load	
			Nitrogen, total ($kg \times 10^3$)	Phosphorus, total ($kg \times 10^3$)			Nitrogen, total ($kg \times 10^3$)	Phosphorus, total ($kg \times 10^3$)
17	Saugatuck River near Redding	21.0	10/73-9/92	10/71-9/92	1985	20.2	12.6	0.456
12	Salmon River at East Hampton	100	10/73-9/91	10/73-9/91	1981	116	80.3	3.13
13	Quinnipiac River at Wallingford	115	7/72-9/92	7/72-9/92	1981	137	410	61.3
16	Naugatuck River at Beacon Falls	260	7/74-9/92	7/74-9/92	1981	331	1,150	140
15	Housatonic River at Stevenson	1,544	10/73-9/92	10/71-9/92	1981	1,550	1,470	75.3
6	Connecticut River at Thompsonville	9,660	10/72-9/91	1/70-9/92	1985	10,900	9,380	664

Station name	High-flow year	Mean discharge (ft^3/s)	High-flow load		Median-flow year	Mean discharge (ft^3/s)	Median-flow load	
			Nitrogen, total ($kg \times 10^3$)	Phosphorus, total ($kg \times 10^3$)			Nitrogen, total ($kg \times 10^3$)	Phosphorus, total ($kg \times 10^3$)
Saugatuck River near Redding	1978	58.6	27.2	1.52	1990	46.6	28.6	0.941
Salmon River at East Hampton	1979	271	187	12.5	1975	200	94	4.76
Quinnipiac River at Wallingford	1983	310	640	78.3	1991	240	577	54.5
Naugatuck River at Beacon Falls	1983	773	1,800	192	1980	594	1,500	165
Housatonic River at Stevenson	1976	3,740	2,780	199	1982	2,720	2,610	131
Connecticut River at Thompsonville	1990	22,300	17,200	881	1987	17,700	13,200	862

Table 19. Regression models used to estimate transport of nitrogen and phosphorus at selected water-quality sampling stations in the Connecticut, Housatonic, and Thames River Basins study unit

[All available data collected from 1972 to 1992 were used to calibrate the models shown in this table. Stations are presented in order of increasing drainage area. Map No.: See plate 1 for location of water-quality sampling station.

$\ln(C \bullet Q) = I + a(\ln(Q) + b(\ln Q^2) + c(\text{TIME}) + d(\text{TIME}^2) + e(\text{sine}(2 \bullet \text{pi} \bullet \text{TIME})) + f(\text{cosine}(2 \bullet \text{pi} \bullet \text{TIME}))$,

where
ln is the natural logarithm;
C is the constituent concentration, in milligrams per liter;
Q is the discharge, in cubic feet per second;
I is the regression intercept;
a, b, c, d, e, and f are the regression coefficients;
TIME is the date in decimal years; pi is 3.141592; and
R^2, the coefficient of determination, is the proportion of total variance in the dependent variable (load) explained by the variation in the independent variables]

Map No.	Station name	Regression intercept	Regression coefficients						Coefficient of determination (R^2)
			a	*b*	*c*	*d*	*e*	*f*	
			Nitrogen, total (as N)						
17	Saugatuck River near Redding	[1]2.9025	[1]1.0430	-0.0049	[1]0.0341	[1]-0.0043	-0.0163	-0.0386	89.6
12	Salmon River near East Hampton	[1]4.9888	[1]1.0548	.0166	[1].0341	[1]-.0029	[1].1255	.0392	91.5
13	Quinnipiac River at Wallingford	[1]7.2202	[1].5914	.0192	-.0023	.0003	.0419	.0410	67.6
16	Naugatuck River at Beacon Falls	[1]8.1338	[1].4757	[1].1179	-.0008	[1]-.0031	-.0158	.0456	57.5
15	Housatonic River at Stevenson	[1]7.8399	[1].9936	.0021	[1].0092	[1]-.0043	.0096	[1].1116	92.3
6	Connecticut River at Thompsonville	[1]10.3935	[1].8000	[1]-.0674	[1].0072	.0005	[1].0873	.0453	83.7
			Phosphorus, total (as P)						
17	Saugatuck River near Redding	[1]-0.4406	[1]1.0504	[1]0.0452	-0.0086	[1]-0.0033	[1]-0.2209	[1]-0.3577	76.6
12	Salmon River near East Hampton	[1]1.2574	[1]1.2342	[1].0966	-.0065	[1]-.0045	-.1725	[1]-.2895	69.4
13	Quinnipiac River at Wallingford	[1]5.1351	[1].4089	.0652	[1]-.0354	.0001	-.0025	-.0577	42.3
16	Naugatuck River at Beacon Falls	[1]5.7221	[1].3294	[1].2487	-.0086	.0007	-.0844	.0703	40.8
15	Housatonic River at Stevenson	[1]4.6656	[1]1.0573	[1].0378	[1]-.0254	[1]-.0029	-.0347	[1].1625	87.0
6	Connecticut River at Thompsonville	[1]7.7576	[1] .7301	.0723	[1]-.0433	-.0007	-.0376	.0372	54.5

Falls, Conn., for total nitrogen and phosphorus and at the Salmon River near East Hampton, Conn., the Housatonic River at Stevenson, Conn., and the Saugatuck River near Redding, Conn., for phosphorus only.

For nitrogen loads, the linear time coefficient (coefficient *c*, table 19) was statistically significant and positive for the Saugatuck River near Redding, Conn., the Salmon River near East Hampton, Conn., the Housatonic River at Stevenson, Conn., and the Connecticut River at Thompsonville, Conn., indicating increasing total nitrogen loads with time at these locations. The linear time coefficient for total phosphorus was statistically significant and negative for the Quinnipiac River at Wallingford, Conn., the Housatonic River at Stevenson, Conn., and the Connecticut River at Thompsonville, Conn., indicating decreasing total phosphorus loads with time.

Seasonal effects were indicated by statistically significant ($p<0.05$) positive coefficients (coefficients *e* and *f*, table 19) for the sine or cosine terms of the load models for total nitrogen in the Connecticut River at Thompsonville, Conn., Housatonic River at Stevenson, Conn., and Salmon near East Hampton, Conn., and for total phosphorus in the Housatonic River at Stevenson, Conn.

Estimated loads of total nitrogen and phosphorus (table 18) reflect characteristics of their station's environmental settings and also are related to the sizes of their drainage basins (tables 9 and 27). Estimated annual loads are smallest for the two stations with the smallest drainage areas. However, the Saugatuck River near Redding, Conn., and the Salmon River at East Hampton, Conn., also have high percentages of forested land use with no major point sources. The Saugatuck station has substantial, low-population-density, unsewered residential areas. For the low, high, and median discharge years selected, the ratios of estimated total nitrogen and total phosphorus loads between the two basins are roughly proportional to their drainage-basin areas. Although the annual mean discharges in the years selected differ considerably between these stations, the ratios of annual mean discharges between stations remain relatively consistent for each of the three discharge conditions selected.

The Quinnipiac River at Wallingford, Conn., drains approximately the same number of square miles as the Salmon River station (115 compared to 100 mi^2), but has a highly urbanized drainage basin, containing numerous point-source discharges. Although their annual mean flows during low, median, and high discharge years are comparable, their estimated loads differ greatly. The estimated loads of nitrogen and phosphorus carried by the Quinnipiac River greatly exceed those carried by the Salmon River. By contrast, the Naugatuck River at Beacon Falls, Conn., drains an urbanized basin about 2.3 times the area of the Quinnipiac River Basin at Wallingford and transports about two to three times the total nitrogen and phosphorus loads; the ratio of their annual mean flows is similar to the ratio between their basin areas and nutrient loads.

Total nitrogen and total phosphorus loads at the Housatonic River at Stevenson, Conn., and the Connecticut River at Thompsonville, Conn., which drain 1,544 and 9,660 mi^2, respectively, are roughly proportional to their drainage areas. Their drainage basins integrate a wide variety of environmental settings and cannot be uniquely classified.

Loads for the six sites investigated fit a pattern associated with drainage area and environmental setting (fig. 49). Within lightly developed, forested, or urban basins, loads are clearly functions of drainage area. The two integrator basins also exhibit loads proportional to their drainage areas. For a given drainage area, highly urbanized basins transport substantially larger loads of nitrogen and phosphorus than forested basins. Available information is insufficient to evaluate load characteristics of drainage basins dominated by agriculture in the study unit.

As noted previously, an historic negative bias in total phosphorus data may indicate that environmental concentrations of total phosphorus are greater than reported concentrations under some conditions. The effect of this bias is believed to be negligible for large, integrator basins and relatively undeveloped smaller basins during median and low-flow conditions. The bias may be a factor during extreme high flows in some basins, and in small, highly urbanized basins under all streamflow conditions. Reported phosphorus loads under these circumstances may be underestimated.

Point-Source Loads

Data on point-source discharges to individual drainage basins of the study unit were obtained from the USEPA, in the PCS of the NPDES, and from the Commonwealth of Massachusetts (Massachusetts Department of Environmental Quality Engineering, 1984, 1985a, 1985b, 1986a, 1986b, 1986c, 1988;

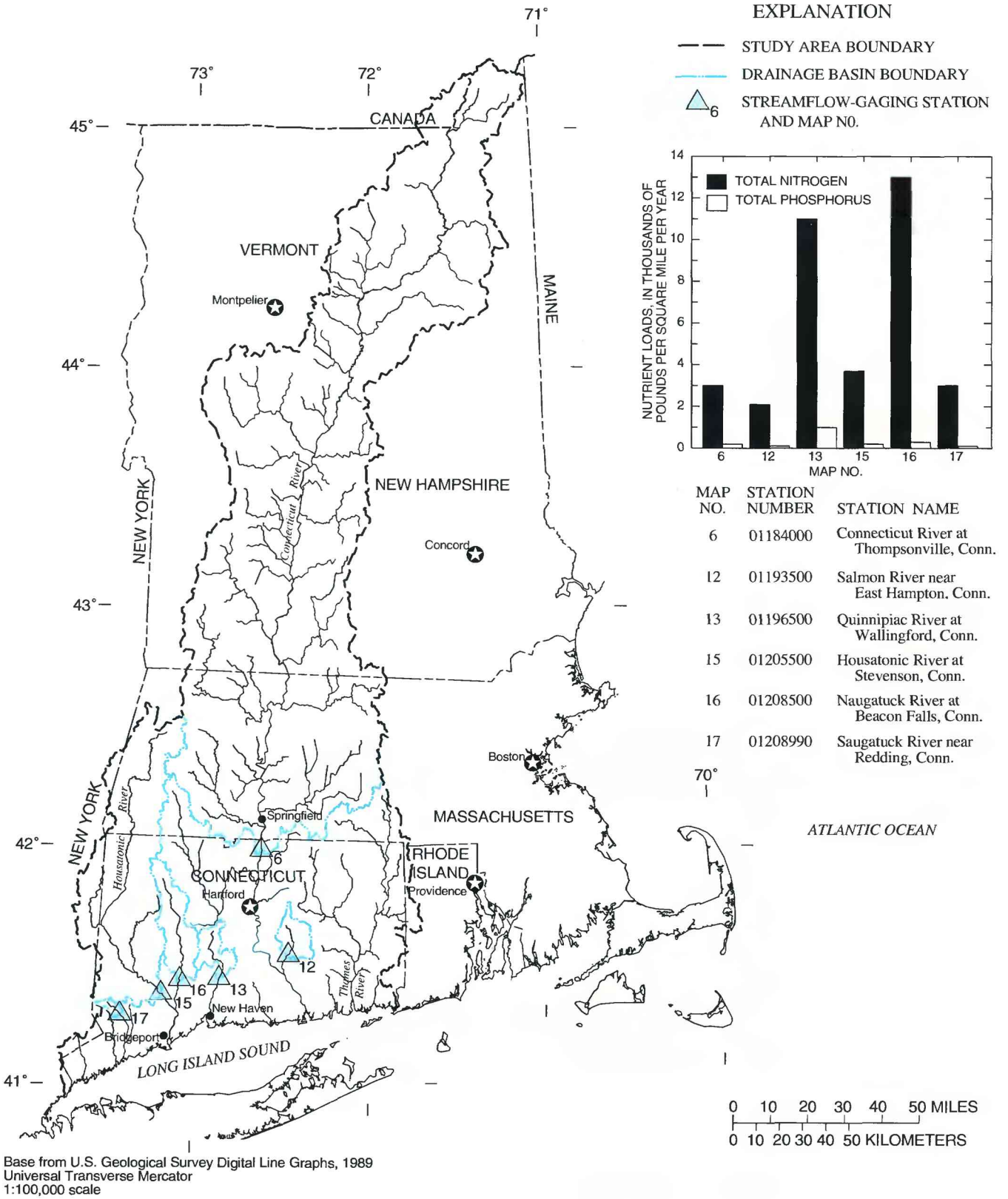


Figure 49. Estimated loads of total nitrogen and phosphorus during median streamflow years at selected surface-water-quality stations in Connecticut.

Massachusetts Department of Environmental Protection, 1989). The study unit contains 177 point-source dischargers with volumes exceeding 1 Mgal/d (fig. 50). Most of these point sources are in Massachusetts and Connecticut and most discharge directly into the Connecticut, Quinebaug, Naugatuck, Pequabuck, and Quinnipiac Rivers, or to Long Island Sound. In New Hampshire and Vermont, the only large dischargers are sewage-treatment plants and paper-processing facilities. The number of industrial dischargers substantially increases in Connecticut. Inconsistent data-collection requirements in the study unit among the various States, especially differences in constituents monitored, precluded making reasonable estimates of nutrient loadings.

From 1983 to 1988, the Massachusetts Department of Environmental Protection (MDEP) (1989) conducted several synoptic studies of nutrients in sewage effluent, covering those parts of all the basins in the study unit that lie in Massachusetts. Samples in the Connecticut River drainage basin were collected at 31 sewage-treatment plant outfalls. Nutrient analyses included Kjeldahl (organic plus ammonia) nitrogen, ammonia-nitrogen, nitrate-nitrogen, and total phosphorus. Plant discharge was measured during each sampling and loads were calculated for each water-quality constituent at the 31 plants. A statistical summary of nutrient concentrations and loads from all 31 plants is shown in table 20. Loadings from sewage-treatment plants into the Connecticut River drainage basin in Massachusetts were 7,134 kg/d of total nitrogen (2.6 million kg/yr) and 950 kg/d of total phosphorus (0.35 million kg/yr).

Nitrogen Loads from Atmospheric Deposition

Atmospheric loads of nitrogen were calculated for the study unit based on data from six NADP sites (table 21). Estimates were made for wet, dry, and droplet deposition of nitrate-nitrogen, for deposition of total ammonia, and for deposition of total nitrogen (table 22).

Sisterson (1990) reports that total wet nitrogen deposition accounts for 42 percent of the total nitrogen deposition for the Eastern United States, including urban wet deposition. Urban-effect deposition (wet plus dry) comprises 28 percent of the total nitrogen deposition in the Eastern United States. Total wet deposition accounts for as much as 60 percent of the total nitrogen deposition (in the Willimantic River drainage basin), or as little as 42 percent of the total (in the Norwalk River drainage basin). All basins in the study unit receive a higher percentage of total nitrogen deposition from wet deposition than do other Eastern United States sites on average, possibly because New England receives more annual precipitation. The basins with the lowest proportion of wet to total deposition are in the more urbanized areas of western Connecticut. Those basins with a high proportion of wet to total deposition generally are in the northern and eastern parts of the State.

The ratio of urban-effect deposition to total deposition varies widely across the study unit. In the north (Connecticut River at North Walpole, N.H.), the urban effect is quite small (4 percent of the total nitrogen deposition), reflecting that area's distance from large urban centers. In the southern urbanized areas, especially the

Table 20. Statistical summary of nutrient concentrations and loads from 31 sewage-treatment plants in Massachusetts, 1983-88

[Data from Massachusetts Department of Environmental Protection, 1989. mg/L, milligram per liter; kg/d, kilogram per day)

Constituent	Percentile				
	Minimum	25th	50th (median)	75th	Maximum
Concentrations (mg/L)					
Nitrogen, total (as N)	3.82	10.91	19.75	29.7	45.7
Nitrogen, nitrate plus ammonia (as N)	.16	5.71	10.0	13.85	23.25
Nitrogen, kjeldahl (as N)	2.25	5.4	10.0	16.0	32.0
Phosphorus, total (as P)	.31	1.4	2.66	4.4	14.65
Loads (kg/d)					
Nitrogen, total (as N)	1.0	7.0	50	260	2,621
Phosphorus, total (as P)	.25	1.7	5.4	38	330

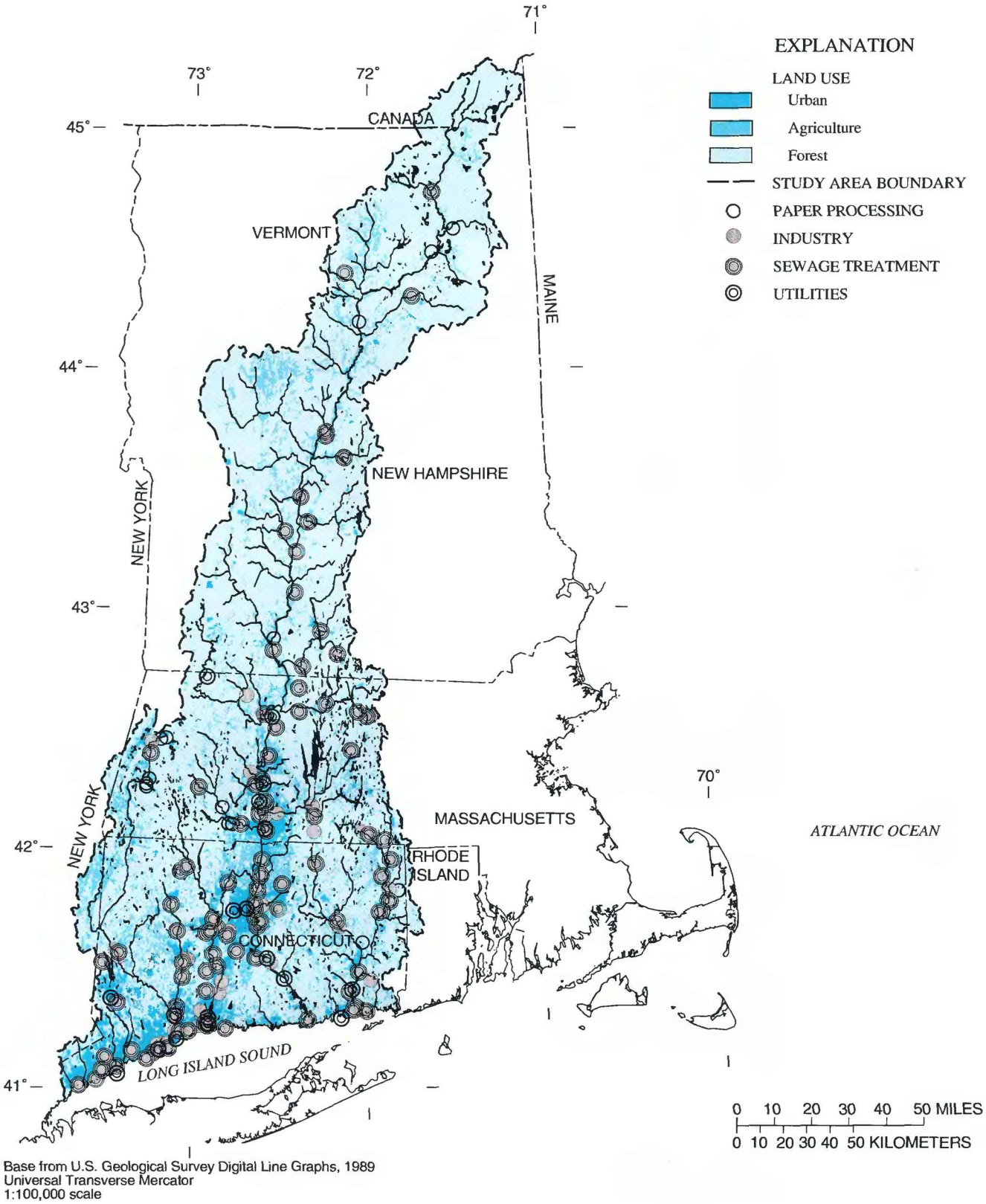


Figure 50. Permitted surface-water dischargers with discharge volumes greater than 1 million gallons per day under the National Pollutant Discharge Elimination System. (Data from the U.S. Environmental Protection Agency, 1990.)

Table 21. Locations and altitudes of National Atmospheric Deposition Program stations used to calculate nutrient deposition

Station name	Latitude ° ′ ″	Longitude ° ′ ″	Altitude (feet above sea level)
Quabbin Reservoir, Mass.	42 23 33	72 20 40	1,004
Hubbard Brook, N.H.	43 56 35	71 42 12	820
Knobit, N.Y.	42 22 42	73 30 10	1,332
West Point, N.Y.	41 21 03	74 02 55	659
Bennington, Vt.	42 52 34	73 09 48	1,001
Underhill, Vt.	44 31 42	72 52 08	1,309

Quinnipiac River and Norwalk River Basins, the ratios are 57 and 66 percent, respectively. Even in relatively nonurbanized areas of southwestern Connecticut (the Saugatuck River Basin), the proportion of urban-effect deposition is fairly high (42 percent), as a result of the basin's proximity to heavily urbanized areas.

An estimation of total nitrogen deposition per unit area per year can be made by dividing total deposition by the basin area. Thus, deposition on a unit-area basis in the Norwalk River Basin in Connecticut is about twice that in the Housatonic River Basin near New Milford, Conn., or the Still River at Riverton, Conn. For most of the basins, the amount of nitrogen deposition is about 2,200 to 2,400 kg/yr/mi^2. In the Quinnipiac and Norwalk River Basins, which are highly urbanized and close to other urban areas, the deposition is about 3,300 and 4,000 kg/yr/mi^2, respectively—much higher than in the other basins.

The effects of the corrections made to the data (see section on "Atmospheric Deposition Estimation") must not be minimized. The variations in wet, dry, and total deposition estimates among basins result almost entirely from variations in the area and the percentage of urban area used in the calculations. Variations in the median areal deposition among NADP stations are small by comparison.

Assessment of Nutrient Loads

Numerous factors complicate determination of the contributions of various components of the total nutrient loads in the study unit. First of all, the areal distribution of surface-water-quality data is strongly skewed toward the southern section of the study unit, particularly Connecticut. This is a valuable distribution of data for estimating loads to areas bordering Long Island Sound. However, this data distribution makes it difficult to obtain a more complete understanding of the hydrologic processes affecting water quality throughout the study unit. The data reported here account for surface water, atmospheric inputs (estimates from NADP data) to the study unit in general, and some point source discharges from treatment plants in Massachusetts. Contributions of ground water, additional point sources, and nonpoint sources of nutrients are not documented and deserve additional study. Rates of movement of nutrients and other materials from terrestrial areas to the aquatic media—that is, to surface and ground waters, are unknown.

The highly regulated nature of streams in New England also increases the difficulty of assessing mass movement through the system. Nutrients and suspended sediment may settle in impoundments and only occasionally are released. Water-quality-sampling stations would be needed just downstream from a greater number of dams to estimate the amount of nutrients released. Additional information needed would include in-pool data to help determine the source of released nutrients (sediment release, resuspension, or water column).

Despite these limitations, it is useful to attempt a synthesis, or compilation, of data in order to direct further work in the study unit. The strongest data sets come from NWIS stations for Connecticut and provide the basis for estimating loads leaving the nontidally affected parts of rivers in the southern part of the study unit. For median discharge water years, the estimated loads for six river basins (table 18) which account for about 75 percent of the surface water discharged from the study unit, amount to 18×10^6 kg/yr of nitrogen and 1.2×10^6 kg/yr of phosphorus. These loads represent, approximately, the nutrient output of the study unit.

The total nutrient inputs to the study unit are less well substantiated. Data on estimated atmospheric loadings of nitrate and ammonia for about 80 percent the study unit show a total input of about 32×10^6 kg/yr of nitrogen; comparable phosphorus data do not exist. Atmospheric deposition is unlikely to be a significant source of phosphorus. Estimates of agricultural fertilizer use have not been quantitatively linked to surface-water quality and the processes that control movement of fertilizer compounds to surface water and ground water are not fully understood. Currently, data have not been compiled on the discharge of nutrients from ground water to surface water of the study unit.

Table 22. Estimates of atmospheric deposition of nitrogen constituents in selected basins of major water-quality stations in the Connecticut, Housatonic, and Thames River Basins study unit

[Deposition calculated based on wet nitrate as nitrogen and ammonia as nitrogen deposition data from the National Atmospheric Trends Network, the basin area, and the land use in each basin. Correction factors from Sisterson (1990) were used to calculate dry deposition and urban deposition (wet and dry). The published value for dry/wet regional deposition for New Hampshire from Sisterson is 12/7, which is more than their specified range (0.43 to 1.0) for Eastern States. The average of Connecticut, Massachusetts, and Vermont was used. Droplet deposition for the two Connecticut River Basins is based on 8 and 5 percent of the land being above 2,000 feet in elevation for the North Walpole and Thompsonville Basins, respectively. Percentage of urban and non-urban land use for each basin is shown in table 27. kg/yr, kilogram per year]

Map No.	Station name	Draiange area (mi^2)	Nitrate as nitrogen deposition (10^3 kg/yr)								Ammonia-nitrogen deposition (10^3 kg/yr)	Total nitrogen deposition (10^3 kg/yr)
			Wet deposition			Dry deposition			Droplet deposition	Total nitrate-nitrogen deposition		
			Regional	Urban	Total	Regional	Urban	Total				
1	Willimantic River at Merrow, Conn.	94.0	76	9.2	85	64	21	85	--	170	38	210
2	Shetucket River at South Windham, Conn.	408	330	52	380	260	120	380	--	760	160	920
3	Quinebaug River at Quinebaug, Conn.	155	130	22	150	100	52	150	--	300	64	360
4	Quinebaug River at Jewett City, Conn.	713	600	76	680	480	180	660	--	1,300	290	1,600
5	Connecticut River at North Walpole, N.H.	5,490	5,200	160	5,400	4,000	360	4,400	2,200	12,000	1,900	14,000
6	Connecticut River at Thompsonville, Conn.	9,660	8,400	680	9,100	6,800	1,600	8,400	2,300	20,000	4,000	24,000
7	Still River at Riverton, Conn.	86.2	68	8	76	56	19	75	--	150	25	180
10	Pequabuck River at Farmington, Conn.	57.2	44	9.6	54	36	22	58	--	110	22	130
12	Salmon River near East Hampton, Conn.	100	80	14	94	64	32	96	--	190	39	230
13	Quinnipiac River at Wallingford, Conn.	115	68	68	140	52	150	200	--	340	44	380
14	Housatonic River near New Milford, Conn.	1,022	720	100	820	600	230	830	--	1,700	370	2,100
15	Housatonic River at Stevenson, Conn.	1,544	1,300	240	1,500	1,000	560	1,600	--	3,100	560	3,700
16	Naugatuck Riverat Beacon Falls, Conn.	260	190	92	280	150	210	360	--	640	100	740
17	Saugatuck River near Redding, Conn.	21.0	17	8.4	25	14	19	33	--	58	8	66
18	Norwalk River at Winnipauk, Conn.	33.0	18	28	46	15	64	79	--	130	13	140

ARMP nitrate data provide information on the quality of surface water of Massachusetts, but generating any load estimates from these data would be difficult. Point-source loads for the Connecticut River Basin in Massachusetts total 2.6×10^6 kg/yr of nitrogen and 0.35×10^6 kg/yr of phosphorus. Comparable point-source data are not currently available elsewhere in the study unit. North of Massachusetts, water-quality data of any kind are limited.

Thus, about 35×10^6 kg/yr of nitrogen and 0.35×10^6 kg/yr of phosphorus reaching land and water in the study unit can be accounted for. At least 18×10^6 kg/yr of nitrogen and 1.2×10^6 kg/yr of phosphorus are estimated to leave the nontidal parts of the study unit annually during a year of median streamflow conditions. There are substantial gaps in information on nutrient inputs to the study unit. Point-source-loading data for Connecticut, New Hampshire, and Vermont are needed to augment these data. Additional information regarding the processes by which atmospheric and fertilizer nutrients move to the streams in the study unit would provide a better understanding of the mechanisms affecting water quality and the effects of nonpoint sources on total nutrient loads in the study unit.

CONCLUSIONS

Most of the analytical data for concentrations of nutrients, suspended sediment, and pesticides in ground water and surface water pertain to the southern part of the study unit, most of which is in Connecticut. Although most of the reviewed data-collection activities to date focused in one State, the implications of these data analyses can be extrapolated to other comparable environmental settings throughout the study unit. Additionally, several water-quality stations located in Connecticut integrate water-quality effects of upstream drainage areas in Massachusetts, New Hampshire, Vermont, New York, Rhode Island, and Quebec.

Data on nutrients in surface water are abundant but unevenly distributed. In Connecticut, surface-water samples were collected from most types of environmental settings in the study area: large and small drainage basins, urbanized and forested basins, and most geologic settings. The following areas and settings are not well represented in the data for the study unit: the northern part of the study unit, particularly New Hampshire and Vermont; agricultural areas in all parts of the study unit; small basins in areas underlain by carbonate bedrock; and small forested basins in the Connecticut Valley Lowland.

Suspended-sediment data are not abundant for streams in the study unit. Long-term data are available only for the larger basins. Information generally is sparse for small drainage basins, agricultural areas, and the northern part of the study unit.

Distribution of nutrient and suspended-sediment data in relation to streamflow conditions is generally good at stations with a substantial amount of data. However, additional data on concentrations at the highest stream discharges would benefit load estimation for all these water-quality constituents. Although concentration data are present for some flows in the top 10th percentile of flow, these data typically do not represent the maximum discharges, which are responsible for carrying most of the annual load for suspended sediment and possibly other constituents. Furthermore, the highest loads are associated with the onset of a flood discharge and most of the above-mentioned data probably were obtained as the discharge was abating.

Urban settings in the study unit are clearly marked by substantially higher concentrations of nutrients in surface water than sparsely populated, forested areas. Median concentrations of nitrogen and phosphorus generally increase from north to south across the study unit, following the pattern of increasing population density and increasing proportion of urbanized land. Median concentrations of nitrogen and phosphorus were highest at stations that monitor streams with numerous major point discharges. Concentrations are consistently high in the Pequabuck, Quinnipiac, Still (near Danbury, Conn.), and Naugatuck Rivers, which drain urbanized areas in central and southwestern Connecticut.

Nitrogen in the surface water of the study unit is a major water-quality concern primarily because of its effects on freshwater aquatic life and its effects on water quality and aquatic life in Long Island Sound. In the fresh surface water of the study unit, nitrogen also serves as an indicator of the extent of contamination from various point and nonpoint sources.

Concentrations of total nitrite plus nitrate were substantially less than 1.0 mg/L (as N), and in many cases typically less than 0.5 mg/L in streams draining forested basins, or basins with small percentages of urbanized area. Concentrations of nitrate or nitrite plus

nitrate exceed presumed background levels on many streams in the study unit, but are well below the MCL for nitrate in drinking water, even on some of the most polluted streams.

Median concentrations of total phosphorus at 20 stations in the study unit exceed 0.1 mg/L—the concentration above which phosphorus is likely to cause excessive aquatic plant growth in fresh surface water. These stations include several large drainage basins with mixed land uses, as well as smaller, heavily urbanized basins.

Data on suspended sediment are too sparse, and too unevenly distributed among the environmental settings of the study unit, to draw conclusions regarding the relation of suspended-sediment concentrations to environmental settings. However, suspended-sediment concentrations generally do not constitute a major water-quality problem in the study unit.

Nutrient data were analyzed for trend at 18 streamflow-gaging stations. These stations represent a large part of the study unit although 17 of the stations are in Connecticut. Trends in the concentration of total nitrite plus nitrate increased at five stations, incorporating several environmental settings. Trends in the concentrations of total ammonia decreased at 11 stations. Trends in the concentrations of total and dissolved phosphorus decreased at 13 and 12 stations, respectively. The decreasing trends in ammonia and phosphorus concentrations may be related to improvements in sewage-treatment practices, or, in the case of phosphorus, to changes in agricultural fertilizer use or changes in laundry detergent composition.

Loads of total nitrogen and total phosphorus were estimated for six streamflow-gaging stations in Connecticut, including two stations that drain parts of other States in the study unit. Although the data are limited, it appears that for comparable basin size, nutrient loads from urbanized basins are substantially greater than from forested basins. For a given basin size, urbanized basins may deliver 3 to 5 times as much nitrogen and 6 to 20 times as much phosphorus as primarily forested basins on an annual basis. Loads were approximately proportional to drainage area in similar environmental settings.

Long-term water-quality information from areas dominated by agriculture were not available to compare with the quality of water affected by urban settings, or to provide information on trends or loads in basins dominated by agricultural nonpoint sources. Thus, some future data-collection efforts need to focus on water-quality effects caused by agricultural activities in the study unit, if these effects are considered to be substantial. The effects of agriculture may need to be assessed to obtain adequate baseline data for determining the direction and extent of future water-quality-sampling activities.

Data are currently inadequate to calculate total nutrient inputs to the basins of the study unit. More extensive verified data are needed to quantify major point source loads in some States, and minor point source loads in all States. Current estimates of the load from atmospheric deposition incorporate numerous qualifying assumptions and considerable uncertainty. Additional understanding is needed of the processes by which nitrogen and phosphorus move from nonpoint sources, particularly agricultural nonpoint sources, to the surface water of the several environmental settings in the study unit. The extent to which ground-water discharge contributes nitrogen and phosphorus to surface water is unknown.

Although nutrient loads were estimated for several major streams in the study unit, analyses were limited to freshwater, nontidal environments. Where streams enter tidal, estuarine areas and mix with saltwater, the processes affecting the fate of nutrients are not fully understood. This lack of knowledge limits evaluation of the amount and timing of actual nutrient loads delivered by streams to Long Island Sound.

Nutrient data were compiled from nearly 2,000 wells throughout the study unit. Although the areal distribution of nutrient sampling in ground water is more extensive than in surface water, sampling is much less frequent at individual wells than at water-quality sampling stations. Wells in stratified-drift aquifers, the most productive sources of ground water, account for 41 percent of the ground-water nutrient sampling sites. High nitrogen concentrations in shallow ground water were statistically related to tilled agricultural, residential, and commercial land uses. Nitrogen concentrations in agricultural areas were high at the same locations as pesticide detections. The frequency of nitrate concentrations exceeding USEPA drinking-water standards was low, and confined to only the shallowest wells.

Data indicated that pesticide occurrence and distribution in surface and ground water of the study unit are much more limited than nutrient data. The bulk of the pesticide data for surface water are for Connecticut, where a significant proportion of these data come from monitoring of public drinking-water

supplies. Comprehensive analytical coverage is lacking, as only a relatively small number of pesticide compounds were analyzed—primarily organochlorine insecticides no longer in use.

Specific pesticide compounds were targeted in a number of studies done on ground water in Connecticut, resulting in considerably more data than are available for pesticides in surface water. Most of the data were from shallow observation wells installed in agricultural settings. Soluble herbicides were the most commonly detected compounds, but their concentrations seldom exceeded drinking-water guidelines. The most extensive ground-water contamination in the study unit was associated with former widespread use of the soil fumigant EDB in tobacco-growing areas, with several hundred wells exceeding drinking-water guidelines.

Future pesticide data-collection efforts are needed to provide more comprehensive analytical coverage, particularly soluble pesticide compounds in surface and ground water. Additional sampling is needed to identify pesticides associated with agricultural land use in the northern part of the study unit, and with urban land uses in the southern part. Information on the seasonal variability of pesticide occurrence in surface- and ground-water runoff also is needed.

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TABLES 23-30

Table 23. Summary of water-quality stations, sampling frequency, and total number of observations for selected nutrient species in the Connecticut, Housatonic, and Thames Rivers study unit, water years 1955-92

[**Map No.**: Locations of water-quality stations are shown on plate 1. Stations listed have at least 10 observations for at least one of the nitrogen, phosphorus, or suspended-sediment water-quality constituents listed in table 6 for water years 1972-92 . **Sampling frequency:** B, bimonthly; D, daily; H, hourly; I, irregular; M, monthly; Q, quarterly; S, summer months only (generally from April through November); W, weekly. **Total number of observations:** Nitrogen (total number of observations equals sum of observations for one or more of the following NWIS and STORET parameter codes: 00600, 00605, 00608, 00610, 00613, 00615, 00618, 00620, 00623, 00625, 00630, 00631, which are identified in table 6). Phosphorus—total number of observations equals sum of observations for one or more of the following NWIS and STORET parameter codes: 00665, 00666, 00671, which are identified in table 6). Suspended sediment—total number of observations for NWIS and STORET parameter code 80154, which is identified in table 6]

Map No.	Agency identification No.	Station name	Latitude ° ′ ″	Longitude ° ′ ″	Period of record	Sampling frequency	Total number of observations		
							Nitrogen (as N)	Phosphorus (as P)	Suspended sediment
		National Water Information System (NWIS)							
1	01119375	Willimantic River at Merrow, Conn.[1]	41 50 07	72 18 38	7/74-10/92	M	1,212	343	0
2	01122610	Shetucket River at South Windham, Conn.[1]	41 40 56	72 09 59	7/74-11/92	M	1759	514	65
3	01124000	Quinebaug River at Quinebaug, Conn.[1]	42 01 20	71 57 22	10/80-10/92	M	871	269	0
4	01127000	Quinebaug River at Jewett City, Conn.[1]	41 35 52	71 59 05	9/71-11/92	M	1,876	558	74
5	01154500	Connecticut River at North Walpole, N.H.[1]	43 07 34	72 26 14	10/80-7/92	Q	294	146	47
6	01184000	Connecticut River at Thompsonville, Conn.[1]	41 59 14	72 36 21	1/66-9/92	M	1,977	586	126
7	01186800	Still River at Riverton, Conn.[1]	41 57 34	73 01 12	8/71-9/91	M	1,202	320	0
8	01188000	Burlington Brook near Burlington, Conn.[1]	41 47 10	72 57 55	3/71-11/92	Q	801	208	0
9	01188085	Farmington River at State Highway 4, at Unionville, Conn.[1]	41 45 52	72 53 47	7/74-9/91	M	1,124	320	0
10	01189030	Pequabuck River at Farmington, Conn.[1]	41 43 00	72 50 25	8/71-11/92	M	1,331	346	1
11	01189995	Farmington River at Tariffville, Conn.[1]	41 54 30	72 45 40	1/71-9/92	M	1,422	377	0
12	01193500	Salmon River near East Hampton, Conn.[1]	41 32 53	72 26 59	5/71-11/92	D,M	1,305	384	321
13	01196500	Quinnipiac River at Wallingford, Conn.[1]	41 26 58	72 50 29	9/71-10/92	M	1,799	494	0
14	01200600	Housatonic River near New Milford, Conn.[1]	41 35 35	73 27 00	7/74-9/91	M	1,143	319	0
15	01205500	Housatonic River at Stevenson, Conn.[1]	41 23 02	73 10 05	2/56-11/92	M	1,964	582	133
16	01208500	Naugatuck River at Beacon Falls, Conn.[1]	41 26 32	73 03 47	7/74-9/92	M	1,632	453	0
17	01208990	Saugatuck River near Redding, Conn.[1]	41 17 40	73 23 44	5/71-10/92	M	1,264	382	0
18	01209710	Norwalk River at Winnipauk, Conn.[1]	41 08 07	73 25 36	10/80-10/92	M	1,133	378	0
19	01122001	Natchaug River at U.S. Highway 6, at Willimantic, Conn.[1]	41 43 10	72 11 46	7/74-9/80	M	393	75	0
20	01122500	Shetucket River near Willimantic, Conn.	41 42 01	72 10 57	5/71-6/74	M	151	39	0
21	01125150	French River at Mechanicsville, Conn.[1]	41 56 51	71 53 23	7/74-9/91	M	1,213	319	0
22	01125200	Quinebaug River at Putnam, Conn.[1]	41 55 07	71 54 38	7/74-9/80	M	393	75	0
23	01125415	Muddy Brook at Childs Hill Road near Woodstock, Conn.[1]	41 57 56	71 57 45	5/80-4/83	D,I	80	33	142
24	01125435	Tributary to Mill Brook at Woodstock, Conn.	41 56 28	71 58 22	12/81-4/83	I	125	57	19

Table 23. Summary of water-quality stations, sampling frequency, and total number of observations for selected nutrient species in the Connecticut, Housatonic, and Thames Rivers study unit, 1955-92 water years—*Continued*

Map No.	Agency identification No.	Station name	Latitude ° ′ ″	Longitude ° ′ ″	Period of record	Sampling frequency	Total number of observations		
							Nitrogen (as N)	Phosphorus (as P)	Suspended sediment
25	01125436	Tributary to Mill Brook at South Woodstock, Conn.	41 56 16	71 58 17	12/81-4/83	I	142	63	21
26	01125475	Tributary to Peake Brook at South Woodstock, Conn.[1]	41 55 48	71 57 17	10/81-8/83	I	355	159	52
27	01125476	Tributary to Peake Brook at Harrisville, Conn.[1]	41 55 50	71 57 05	10/81-8/83	I	356	156	52
28	01125720	Quinebaug River at Pomfret Landing, Conn.[1]	41 50 57	71 55 17	7/74-9/80	M	393	75	0
29	01127500	Yantic River at Yantic, Conn.[1]	41 33 31	72 07 19	5/71-9/80	M	543	115	0
30	01137500	Ammonoosuc River at Bethlehem Junction, N.H.	44 16 08	71 37 52	5/67-6/74	M	0	0	41
31	01138500	Connecticut River at Wells River, Vt.	44 09 13	72 02 34	4/79-7/86	Q	242	108	37
32	01141900	Connecticut River at Wilder, Vt.	43 40 04	72 18 13	8/69-6/73	M	113	49	0
33	01144000	White River at West Hartford, Vt.	43 42 51	72 25 07	4/67-6/74	I	0	0	65
34	01152750	Black River above Ludlow,Vt.	43 24 22	72 42 29	4/79-11/80	M	113	16	0
35	01152760	Black River above Cavendish, Vt.	43 23 11	72 40 08	11/76-11/80	M	198	28	0
36	01152770	Black River below Cavendish, Vt.	43 23 04	72 35 57	11/76-11/80	M	198	28	0
37	01152798	Black River near Hawk Mountain, below Cavendish, Vt.	43 23 37	72 32 14	11/76-11/80	M	198	28	0
38	01152800	Black River at Weathersfield,Vt.	43 23 55	72 31 14	11/76-11/80	M	191	27	0
39	01152850	Black River at Tolles Hill Dam, near Weathersfield, Vt.	43 22 10	72 30 06	11/76-11/80	M	198	28	0
40	01153000	Black River at North Springfield, Vt.	43 20 00	72 30 55	11/76-11/80	M	198	28	0
41	01153025	Black River at Gilman Dam, Springfield, Vt.	43 18 09	72 29 12	11/76-11/80	M	198	28	0
42	01153075	Black River below Springfield, Vt.	43 16 42	72 28 03	11/76-11/80	M	198	28	0
43	01153500	Williams River at Brockways Mills, Vt.	43 12 31	72 31 05	4/67-4/74	I	0	0	47
44	01155050	Connecticut River at Walpole, N.H.[1]	43 05 04	72 26 04	5/55-9/80	M	324	102	56
45	01161280	Connecticut River near Northfield, Mass.[1]	42 41 00	72 28 19	10/74-8/80	M	376	54	0
46	01172000	Connecticut River at Holyoke, Mass.[1]	42 12 50	72 36 10	10/74-8/80	M	378	54	0
47	01172680	Natty Pond Brook at Templeton Road, near Hubbardston, Mass.	42 29 44	72 01 24	11/84-10/86	I	150	32	11
48	01172800	Natty Pond Brook near Hubbardston, Mass.	42 27 06	72 01 48	9/83-10/86	I	161	33	11
49	01174050	East Branch Fever Brook near Petersham, Mass.[1]	42 28 49	72 13 27	9/83-8/85	W	318	60	6
50	01174565	West Branch Swift River near Shutesbury, Mass.[1]	42 27 18	72 22 56	9/83-8/85	W	310	58	4
51	01174569	Dickey Brook Tributary #2 (site 7) near Cooleyville, Mass.	42 25 10	72 20 34	6/88-8/89	M	34	17	9
52	01174570	Dickey Brook near Cooleyville, Mass.[1]	42 25 42	72 21 06	2/85-8/89	I	160	84	42
53	01174575	Dickey Brook Tributary near Cooleyville, Mass.[1]	42 26 38	72 22 13	2/85-8/89	I	155	83	42
54	01181000	West Branch Westfield River at Huntington, Mass.	42 14 14	72 53 46	5/67-11/73	I	0	0	33

Table 23. Summary of water-quality stations, sampling frequency, and total number of observations for selected nutrient species in the Connecticut, Housatonic, and Thames Rivers study unit, 1955-92 water years—*Continued*

Map No.	Agency identification No.	Station name	Latitude ° ′ ″	Longitude ° ′ ″	Period of record	Sampling frequency	Total number of observations		
							Nitrogen (as N)	Phosphorus (as P)	Suspended sediment
55	01183750	Connecticut River at Agawam, Mass.[1]	42 02 57	72 36 35	5/73-8/80	M	383	56	0
56	01184100	Stony Brook near West Suffield, Conn.[1]	41 57 38	72 42 39	10/80-9/91	B,D	508	150	347
57	01189120	Farmington River at Avon, Conn.[1]	41 48 24	72 49 23	8/71-7/91	Q	540	136	0
58	01190015	Farmington River at Windsor, Conn.	41 51 25	72 38 21	7/74-9/76	M	135	27	0
59	01190045	Podunk River at South Windsor, Conn.	41 50 07	72 33 43	1/75-7/81	M	252	53	0
60	01191510	Park River at Hartford, Conn.[1]	41 45 50	72 41 13	7/74-8/79	M	305	62	0
61	01192370	Porter Brook near Manchester, Conn.[1]	41 45 55	72 30 12	7/75-9/81	M	408	87	0
62	01192883	Coginchaug River at Middlefield, Conn.[1]	41 31 12	72 43 23	2/81-9/82	D	0	0	387
63	01193450	Schroeder Brook at South Marlborough, Conn.	41 35 33	72 25 50	7/86-1/90	I	97	49	0
64	01193465	Dickinson Creek near Marlborough, Conn.[1]	41 39 48	72 29 06	8/86-1/90	I	98	50	0
65	01196222	Quinnipiac River near Meriden, Conn.[1]	41 31 45	72 51 50	7/74-10/92	M	1,328	342	0
66	01197500	Housatonic River near Great Barrington, Mass.[1]	42 13 55	73 21 19	9/71-4/87	D	0	84	188
67	01198135	Housatonic River near Sodom, Conn.[1]	42 03 26	73 20 57	10/83-9/91	M	565	172	2
68	01198550	Housatonic River near Canaan, Conn.[1]	42 00 17	73 21 27	7/74-9/83	M	704	145	0
69	01199000	Housatonic River at Falls Village, Conn.[1]	41 57 26	73 22 11	11/71-4/80	B,D	58	10	181
70	01199290	Housatonic River at Kent, Conn.[1]	41 43 36	73 28 55	6/85-10/87	D	0	0	313
71	01200500	Housatonic River at Gaylordsville, Conn.[1]	41 39 11	73 29 25	4/79-3/80	D	0	0	224
72	01201010	Lake Waramaug Brook (inflow site 7) at Warren, Conn.	41 44 23	73 20 30	3/77-3/78	I	35	40	17
73	01201020	Lake Waramaug Brook (inflow site 26) near Warren, Conn.	41 43 10	73 19 56	6/77-3/78	I	15	33	13
74	01201030	Lake Waramaug Brook (inflow site 2) near New Preston, Conn.	41 42 02	73 20 50	3/77-3/78	I	55	54	21
75	01201485	Still River at Brookfield Junction, Conn.[1]	41 27 23	73 23 47	7/74-9/92	M	1,319	342	0
76	01203000	Shepaug River near Roxbury, Conn.[1]	41 32 59	73 19 49	7/74-11/92	M	1,214	345	0
77	01204000	Pomperaug River at Southbury, Conn.	41 28 50	73 13 30	10/66-5/79	M	144	42	1
78	01201050	East Aspetuck River (inflow site 23) at New Preston, Conn.	41 40 57	73 21 15	3/77-3/78	I	45	15	10
79	01208049	Naugatuck River near Waterville, Conn.[1]	41 36 55	73 03 30	10/67-9/92	M	928	266	0
80	01209570	Norwalk River at Georgetown, Conn.	41 14 45	73 26 05	4/76-4/78	M	108	22	22
81	01209572	Norwalk River at Cannondale, Conn.	41 13 52	73 25 35	4/76-4/78	M	110	22	22
82	01209700	Norwalk River at South Wilton, Conn.	41 09 49	73 25 11	4/76-4/78	M	105	22	22
83	01201040	Tributary to Lake Waramaug (inflow site 9) near New Preston, Conn.	41 42 29	73 22 45	3/77-3/78	I	40	50	22

Table 23. Summary of water-quality stations, sampling frequency, and total number of observations for selected nutrient species in the Connecticut, Housatonic, and Thames Rivers study unit, 1955-92 water years—*Continued*

Map No.	Agency identification No.	Station name	Latitude ° ′ ″	Longitude ° ′ ″	Period of record	Sampling frequency	Total number of observations		
							Nitrogen (as N)	Phosphorus (as P)	Suspended sediment
84	01125100	French River at North Grosvenordale, Conn.	41 58 41	71 54 03	10/91-10/92	M	79	24	0
85	01135500	Passumpsic River at Passumpsic, Vt.	44 21 56	72 02 23	5/67-11/73	I	0	0	35
86	01140560	Lake Morey Tributary #8 near Fairlee, Vt.	43 55 41	70 28 21	3/81-12/82	B	30	23	0
87	01140565	Lake Morey Tributary #7 near Fairlee, Vt.	43 55 49	72 08 18	3/81-12/82	B	31	23	0
88	01140570	Lake Morey Tributary #6 near Fairlee, Vt.	43 56 10	72 08 14	1/81-12/82	B	34	26	0
89	01140575	Lake Morey Tributary #5 near Fairlee, Vt.	43 56 10	72 08 29	1/81-12/82	B	34	26	0
90	01140580	Big Brook (Lake Morey Tributary #4) near Fairlee, Vt.	43 56 05	72 08 33	1/81-12/82	B	34	26	0
91	01140590	Glenn Falls Brook (Lake Morey Tributary #3) near Fairlee, Vt.	43 55 13	72 09 37	3/81-12/82	B	30	23	0
92	01140598	Lake Morey Tributary #1 near Fairlee, Vt.	43 54 41	72 09 36	1/81-12/82	B	37	26	0
93	01140600	Lake Morey outlet at Fairlee, Vt.	43 54 40	72 09 30	1/81-12/82	B	35	26	0
94	01153050	Black River below Comtu Falls Dam, Springfield, Vt.	43 17 49	72 28 55	11/76-11/78	M	85	12	0
95	01161200	Connecticut River at Northfield, Mass.	42 42 44	72 27 14	8/69-6/73	M,S	84	34	0
96	01172700	Natty Pond Brook at Hubbardston, Mass.	42 28 41	72 01 26	2/85-10/86	I	97	20	6
97	01172750	Natty Pond Brook at dirt road at Hubbardston, Mass.	42 27 52	72 00 49	2/85-10/86	I	105	22	6
98	01176000	Quaboag River at West Brimfield, Mass.	42 10 56	72 15 51	5/67-10/73	I	0	0	26
99	01188090	Farmington River at Unionville, Conn.	41 45 21	72 53 14	8/71-9/92	M	57	23	0
100	01192500	Hockanum River near East Hartford, Conn.	41 46 59	72 35 16	10/91-9/92	M	71	22	0
101	01195359	Farm River, at Reeds Gap Rd East, near Northford, Conn.	41 24 34	72 45 22	7/75-5/76	I	30	10	10
102	01195362	Pistapaug Pond Outlet near East Wallingford, Conn.	41 25 25	72 45 19	7/75-5/76	I	30	10	10
103	01195368	Tributary to Farm River near Northford, Conn.	41 24 34	72 46 22	7/75-5/76	I	30	10	10
104	01195370	Farm River near Northford, Conn.	41 24 18	72 46 41	7/75-5/76	I	30	10	10
105	01195380	Farm River at Northford, Conn.	41 22 56	72 47 41	7/75-5/76	I	30	10	10
106	01195390	Farm River at Augur Road, near Northford, Conn.	41 21 44	72 48 06	7/75-5/76	I	30	10	10
107	01195393	Farm River near Totoket, Conn.	41 20 34	72 48 46	7/75-5/76	I	30	10	10
108	01195395	Tributary to Farm River at Barberry Road, at Totoket, Conn.	41 20 28	72 49 20	6/75-5/76	I	30	10	10
109	01195397	Tributary to Farm River at Totoket, Conn.	41 20 09	72 49 50	7/75-5/76	I	30	10	10
110	01195402	Burrs Brook at Totoket, Conn.	41 19 26	72 48 53	6/75-5/76	I	30	10	10
111	01195405	Farm River at Foxon, Conn.	41 19 24	72 50 02	6/75-5/76	I	30	10	10
112	01195408	Maloney Brook at Foxon, Conn.	41 19 20	72 51 13	6/75-5/76	I	30	10	10
113	01195413	Farm River near East Haven, Conn.	41 17 40	72 51 43	6/75-5/76	I	30	10	10
114	01198125	Housatonic River near Ashley Falls, Mass.	42 04 29	73 20 02	10/91-10/92	M	77	24	0

Table 23. Summary of water-quality stations, sampling frequency, and total number of observations for selected nutrient species in the Connecticut, Housatonic, and Thames Rivers study unit, 1955-92 water years—*Continued*

Map No.	Agency identification No.	Station name	Latitude ° ′ ″	Longitude ° ′ ″	Period of record	Sampling frequency	Total number of observations		
							Nitrogen (as N)	Phosphorus (as P)	Suspended sediment
115	01198800	Hollenbeck River at Huntsville, Conn.	41 56 55	73 19 18	11/71-5/74	B	57	10	0
116	01198851	Wangum Lake Brook near Huntsville, Conn.	41 56 36	73 17 08	11/71-5/74	B	58	10	0
117	01198880	Wangum Lake Brook near South Canaan, Conn.	41 57 42	73 18 52	11/71-5/74	B	62	11	0
118	01201045	Tributary to Lake Waramaug (inflow site 3) near New Preston, Conn.	41 42 07	73 21 04	3/77-3/78	I	50	17	10
119	01199900	Tenmile River at South Dover near Wingdale, NY	41 39 46	73 33 35	10/91-10/92	M	66	24	0
		STOrage and RETrieval system (STORET)							
120	413124073042701	Welton Brook at Waterbury, Conn.[1]	41 31 24	73 04 27	8/74-10/88	I	224	66	0
121	413059073041001	Hop Brook Dam at Waterbury, Conn.[1]	41 30 59	73 04 10	6/70-9/87	I	111	15	0
122	413118073044301	Confluence of Hop and Shattuck Brooks at Waterbury, Conn.[1]	41 31 18	73 04 43	6/70-10/88	I,M	159	67	0
123	414137073055901	Northfield Brook at Thomaston, Conn.[1]	41 41 37	73 05 59	6/70-8/88	I	142	23	0
124	414405073062001	Naugatuck River at Campville, Conn.[1]	41 44 05	73 06 20	7/70-8/85	I,W	146	28	0
125	414207073032801	Leadmine Brook at Thomaston, Conn.[1]	41 42 07	73 03 28	7/70-9/87	I,W	141	29	0
126	414044073052801	Northfield Brook at Thomaston, Conn.[1]	41 40 44	73 05 28	6/70-8/88	W	170	23	0
127	414111073035601	Naugatuck River at Thomaston, Conn.[1]	41 41 11	73 03 56	7/70-8/85	I	187	29	0
128	414837072131501	Fenton River at Mansfield, Conn.	41 48 37	72 13 15	5/80-9/86	M	18	9	0
129	414729072103401	Mount Hope River at Mansfield, Conn.	41 47 29	72 10 34	5/80-9/86	M	18	10	0
130	414519072110201	Natchaug River at Mansfield Hollow, Conn.[1]	41 45 19	72 11 02	6/71-9/86	M,W	125	10	0
131	420541071545101	South Fork Little River at Oxford, Mass.[1]	42 05 41	71 54 51	5/71-9/88	W	189	42	0
132	420535072044501	Quinebaug River at Sturbridge, Mass.[1]	42 05 35	72 04 45	5/71-8/88	I,W	117	29	0
133	420447072025504	Quinebaug River below Westville Dam, Southbridge, Mass.[1]	42 04 47	72 02 55	8/72-8/88	M,W	148	29	0
134	420819071544701	Little River at Oxford, Mass.[1]	42 08 19	71 54 47	5/71-9/88	S	177	43	0
135	420657071541701	Little River below Buffumville Dam, Oxford, Mass.[1]	42 06 57	71 54 17	5/71-9/88	S	197	42	0
136	420704071525201	French River below Hodges Village Dam, Oxford, Mass.[1]	42 07 04	71 52 52	5/71-7/87	M	152	13	0
137	420104071573601	Tributary to Quinebaug River at Quinebaug, Conn..	42 01 04	71 57 36	5/82-10/88	I	88	44	0
138	420632072070401	Quinebaug River below East Brimfield Dam, Sturbridge, Mass.[1]	42 06 32	72 07 04	5/71-8/88	I,M	173	44	0
139	420849071523501	French River at Oxford, Mass.	42 08 49	71 52 35	5/80-7/87	I	26	14	0
140	420625072085501	Quinebaug River at East Brimfield, Mass.	42 06 25	72 08 55	5/71-8/88	I	133	45	0
141	420437072090001	Quinebaug River at Holland, Mass.[1]	42 04 37	72 09 00	5/80-8/88	M	90	45	0
142	420552072103201	Mill Brook at Brimfield, Mass.	42 05 52	72 10 32	5/80-8/88	I	84	43	0

Table 23. Summary of water-quality stations, sampling frequency, and total number of observations for selected nutrient species in the Connecticut, Housatonic, and Thames Rivers study unit, 1955-92 water years—*Continued*

Map No.	Agency identification No.	Station name	Latitude ° ′ ″	Longitude ° ′ ″	Period of record	Sampling frequency	Total number of observations		
							Nitrogen (as N)	Phosphorus (as P)	Suspended sediment
143	415858071551801	Quinebaug River at Thompson, Conn.[1]	41 58 58	71 55 18	5/71-10/88	I	188	33	0
144	415806071552001	Quinebaug River at Thompson, Conn.	41 58 06	71 55 20	1/81-10/88	M	92	45	0
145	415646071540501	Quinebaug River below West Thompson Dam at Thompson, Conn.[1]	41 56 46	71 54 05	5/71-10/88	M,W	285	59	0
146	420119071571601	Quinebaug River at Fabyan, Conn.[1]	42 01 19	71 57 16	5/78-10/88	I,S	177	59	0
147	420035071563301	Quinebaug River at Fabyan, Conn.[1]	42 00 35	71 56 33	5/71-10/88	I	183	38	0
148	415749071541306	Quinebaug River at Thompson, Conn.[1]	41 57 49	71 54 13	3/75-10/88	I,W	209	57	0
149	415920073010401	West Branch Farmington River below Goodwin Dam, Hartland, Conn.	41 59 20	73 01 04	4/81-9/86	M	16	8	0
150	421924072521501	Little River at Huntington, Mass.	42 19 24	72 52 15	7/70-5/86	S	67	10	0
151	422026072504901	Westfield River at Huntington, Mass.	42 20 26	72 50 49	5/80-7/88	S	24	12	0
152	421716072515401	Westfield River below Knightville Dam at Huntington, Mass.[1]	42 17 16	72 51 54	7/70-5/86	M	126	11	0
153	420353072170701	Vinica Brook at South Monson, Mass.[1]	42 03 53	72 17 07	5/70-7/87	M	93	14	0
154	420425072173101	Conant Brook at Monson, Mass.	42 04 25	72 17 31	6/71-7/87	I	34	16	0
155	425747072594201	Deerfield River at Somerset, Vt.	42 57 47	72 59 42	2/84-1/85	M	13	10	0
156	425214072530601	North Branch Deerfield River at Wilmington, Vt.	42 52 14	72 53 06	8/85-8/85	H	59	13	0
157	425214072525201	North Branch Deerfield River at Wilmington, Vt.	42 52 14	72 52 52	8/85-8/85	H	62	14	0
158	425220072530801	Binney Brook at Wilmington, Vt.	42 52 20	72 53 08	8/85-8/85	H	46	13	0
159	425222072531801	North Branch Deerfield River at Wilmington, Vt.	42 52 22	72 53 18	8/85-8/85	H	61	13	0
160	424126072133001	East Branch Tully River at Royalston, Mass.	42 41 26	72 13 30	5/80-9/86	M,S	28	15	0
161	423623072043301	Otter River at Baldwinsville, Mass.	42 36 23	72 04 33	4/76-7/85	S	66	26	0
162	423757072073001	Miller River below Birch Hill Dam, Royalston, Mass.	42 37 57	72 07 30	5/71-7/85	S	60	26	0
163	423856072121801	Lawrence Brook below Doanes Falls, Royalston, Mass.	42 38 56	72 12 18	1/76-9/86	S	42	14	0
164	424057072055601	Priest Brook at Royalston, Mass.	42 40 57	72 05 56	5/80-7/85	S	50	26	0
165	424103072050201	Miller River at Winchendon, Mass.	42 41 03	72 05 02	5/80-7/85	S	50	26	0
166	423832072133401	East Branch Tully River at Tully Dam in Royalston, Mass.	42 38 32	72 13 34	5/80-9/86	S	28	14	0
167	425312072171801	Ashuelot River at Swanzey, N.H.	42 53 12	72 17 18	4/76-8/89	M,S	188	77	0
168	430122072191701	Merriam Brook at Surry, N.H.[1]	43 01 22	72 19 17	4/75-9/86	S,W	100	18	0
169	430110072193001	Merriam Brook at Surry, N.H.[1]	43 01 10	72 19 30	6/70-9/86	I,W	119	18	0
170	425806072133201	Otter Brook at Keene, N.H.[1]	42 58 06	72 13 32	6/71-9/86	W	128	18	0
171	425636072143008	Otter Brook below Otter Brook Dam, at Keene, N.H.[1]	42 56 36	72 14 30	6/71-9/86	S,W	146	17	0

Table 23. Summary of water-quality stations, sampling frequency, and total number of observations for selected nutrient species in the Connecticut, Housatonic, and Thames Rivers study unit, 1955-92 water years—*Continued*

Map No.	Agency identification No.	Station name	Latitude ° ′ ″	Longitude ° ′ ″	Period of record	Sampling frequency	Total number of observations		
							Nitrogen (as N)	Phosphorus (as P)	Suspended sediment
172	425931072183004	Ashuelot River below Surry Mountain Dam, at Surry, N.H.[1]	42 59 31	72 18 30	7/70-9/86	I	158	22	0
173	430259072200001	Ashuelot River at Gilsum, N.H.	43 02 59	72 20 00	5/80-9/86	M,S	42	22	0
174	430530072522801	Kidder Brook at Jamaica, Vt.	43 05 30	72 52 28	10/85-10/87	W	53	16	0
175	430818072555401	Winhall River at Winhall, Vt.	43 08 18	72 55 54	3/82-1/85	M	22	11	0
176	430923072493701	Winhall River near Rawsonville, Vt.	43 09 23	72 49 37	5/80-9/86	M,S	24	13	0
177	431134072485501	West River at South Londonderry, Vt.	43 11 34	72 48 55	5/80-9/86	M,S	24	12	0
178	430932072475101	Winhall River at Londonderry, Vt.	43 09 32	72 47 51	8/77-9/86	M,S	38	12	0
179	430700072475601	North Branch Ball Mountain Brook at Jamaica, Vt.	43 07 00	72 47 56	10/85-10/88	W	83	30	0
180	430416072440601	Wardsboro Brook at Jamaica, Vt.	43 04 16	72 44 06	1/76-9/86	I,S	57	15	0
181	430422072440801	West River at East Jamaica, Vt.	43 04 22	72 44 08	6/71-9/86	I,W	109	14	0
182	430310072421301	West River at Townshend, Vt.	43 03 10	72 42 13	9/81-9/86	M,S	20	9	0
183	430335072522501	Ball Mountain Brook at West Jamaica, Vt.	43 03 35	72 52 25	2/84-1/85	M	13	12	0
184	430718072535901	West River at Stratton, Vt.	43 07 18	72 53 59	8/88-10/88	W	27	14	0
185	430554072514001	North Branch Ball Mountain Brook at Jamaica, Vt.	43 05 54	72 51 40	8/88-10/88	W	23	14	0
186	431956072310101	West River at North Springfield, Vt.[1]	43 19 56	72 31 01	6/71-9/86	I,W	179	29	0
187	433845072245801	Ottauquechee River at Quechee, Vt.[1]	43 38 45	72 24 58	7/74-8/88	W	142	37	0
188	432425072302301	North Branch Black River at Amsden, Vt.[1]	43 24 25	72 30 23	6/71-9/86	S,W	143	31	0
189	432225072304901	Black River at Perkinsville, Vt.[1]	43 22 25	72 30 49	6/71-9/86	I,W	142	31	0
190	433800072243701	Ottauquechee River at Quechee Gorge, Vt.	43 38 00	72 24 37	5/80-8/88	I,S	86	40	0
191	433842072250101	Ottauquechee River at Quechee Village, Vt.	43 38 42	72 25 01	9/81-10/82	W	30	14	0
192	433614072213102	Ottauquechee River at North Hartland, Vt.	43 36 14	72 21 31	9/81-9/88	I	36	15	0
193	432227072295701	North Branch Black River at Weathersfield, Vt.	43 22 27	72 29 57	5/80-8/88	S,W	58	28	0
194	440251072443401	Third Branch White River at Roxbury, Vt.	44 02 51	72 44 34	6/82-8/82	W	49	9	0
195	440338072443201	Third Branch White River at Roxbury, Vt.	44 03 38	72 44 32	6/82-8/82	W	50	10	0
196	435942072524201	White River at Granville, Vt.	43 59 42	72 52 42	2/84-1/85	M	12	11	0
197	435614072541501	Hancock Branch tributary to White River at Hancock, Vt.	43 56 14	72 54 15	2/84-1/85	M	13	12	0
198	435944072524101	Clark Brook at Granville, Vt.	43 59 44	72 52 41	2/84-1/85	M	11	10	0
199	441359071381502	Scarface Brook at Bethlehem, N.H.	44 13 59	71 38 15	12/82-11/90	M	83	0	0
200	435544072091601	Morey Tributary #3 at Fairlee, Vt.[1]	43 55 44	72 09 16	6/81-12/82	W	0	184	0
201	442838071274002	Garland Brook at Lancaster, N.H.[1]	44 28 38	71 27 40	6/80-11/90	M	172	0	0
202	441408071365001	North Branch Gale River at Bethlehem, N.H.	44 14 08	71 36 50	9/71-11/90	B	76	0	0
203	424555072303501	Connecticut River at Hinsdale, N.H.	42 45 55	72 30 35	10/80-10/86	I	0	14	0

Table 23. Summary of water-quality stations, sampling frequency, and total number of observations for selected nutrient species in the Connecticut, Housatonic, and Thames Rivers study unit, 1955-92 water years—*Continued*

Map No.	Agency identification No.	Station name	Latitude ° ′ ″	Longitude ° ′ ″	Period of record	Sampling frequency	Total number of observations		
							Nitrogen (as N)	Phosphorus (as P)	Suspended sediment
204	432824072230001	Connecticut River at Cornish, N.H.	43 28 24	72 23 00	4/76-11/81	M	161	75	0
205	442945071354501	Connecticut River at Lancaster, N.H.	44 29 45	71 35 45	6/78-7/89	I	123	61	0
206	424919072324401	Connecticut River at Hinsdale, N.H.	42 49 19	72 32 44	9/70-10/86	W	1	14	0
207	433608072211701	Ottauquechee River at North Hartland, Vt.[1]	43 36 08	72 21 17	6/71-10/82	W	166	24	0
208	441850072095301	South Peacham Brook at Peacham, Vt.	44 18 50	72 09 53	8/55-9/82	W	46	10	0
209	441840072092401	South Peacham Brook at South Peacham, Vt.	44 18 40	72 09 24	6/82-8/82	W	49	9	0
210	441824072083001	South Peacham Brook at Barnet, Vt.[1]	44 18 24	72 08 30	3/80-10/82	D	0	376	0
211	441816072082201	Steven's River at Barnet, Vt.	44 18 16	72 02 82	3/80-12/80	M	0	22	0
212	441749072074301	Harvey's Tributary #2 at Barnet, Vt.[1]	44 17 49	72 07 43	2/81-10/82	D	0	236	0
213	435430072153001	East Branch Ompompanoosuc River at West Fairlee, Vt.	43 54 30	72 15 30	6/79-10/88	M	77	32	0
214	435014072205501	West Branch Ompompanoosuc River at South Strafford, Vt.[1]	43 50 14	72 20 55	6/79-10/88	M,S	147	71	0
215	434956072151001	East Branch Ompompanoosuc River at Thetford Center, Vt.[1]	43 49 56	72 15 10	6/71-10/88	S,W	210	72	0
216	434857072162901	West Branch Ompompanoosuc River at Thetford, Vt.[1]	43 48 57	72 16 29	6/71-10/88	I,S	214	71	0
217	434731072152501	Ompompanoosuc River at Union Village, Vt.[1]	43 47 31	72 15 25	6/71-10/88	W	242	72	0
218	443500071470401	Moose River at Granby, Vt.	44 35 00	71 47 04	2/84-1/85	I	13	12	0
219	442510072005001	Passumpsic River at St. Johnsbury, Vt.	44 25 10	72 00 50	8/66-8/85	H	61	0	0
220	444930071340801	Clough Brook at Lemington, Vt.	44 49 30	71 34 08	2/84-1/85	M	13	12	0
221	441932072020801	Passumpsic River at East Barnet, Vt.	44 19 32	72 02 08	7/85-8/85	H	60	0	0
222	440553071505401	Wild Ammonoosuc River at Benton, N.H.	44 05 53	71 50 54	9/72-9/90	I	44	0	0
223	442950071214002	West Branch Upper Ammonoosuc River at Berlin, N.H.	44 29 50	71 21 40	3/79-11/90	I	68	0	0
224	442952071214401	Small Brook at Berlin, N.H.	44 29 52	71 21 44	3/79-1/93	I	21	0	0
225	442955071213201	Number 9 Brook at Berlin, N.H.	44 29 55	71 21 32	3/79-11/90	I	31	0	0
226	443024071200501	Cold Brook at Berlin, N.H.	44 30 24	71 20 05	4/73-11/90	I	78	0	0
227	441535071380002	Beaver Brook at Bethlehem, N.H.[1]	44 15 35	71 38 00	3/79-11/90	M	203	0	0
228	413150073050401	Wooster Brook at Waterbury, Conn.[1]	41 31 50	73 05 04	7/74-10/88	M,W	240	69	0

[1]Names of stations where at least 50 water-quality samples were collected. Each sample may result in several observations.

Table 24. Number of ground- and surface-water sites with pesticide water-quality data in the Connecticut, Housatonic, and Thames Rivers study unit

[Pesticides are in micrograms per liter unless otherwise noted. NWIS, U.S. Geological Survey's National Water and Information System; STORET, U.S. Environmental Protection Agency's Storage and Retrieval system. Ground-water data include samples from wells (water years 1978-89) and springs (water year 1988); surface water includes samples from streams (water years 1969-92 from NWIS and 1962-87 from STORET), estuaries (water years 1971-76 from NWIS), and lakes (water years 1975-91 from NWIS and 1973-74 from STORET).
DDD, dichlorodiphenlydichloroethane; DDE, dichloro-diphenyldichloroethylene; DDT, dichlorodiphenyltrichloroethane; DEF, S,S,S-tributylphosphorotrithioate; 2,4-D, (2,4-dichlorophenoxy) acetic acid; 2,4-DP, 2-(2,4-dichlorophenoxy) propionic acid; 2,4,5-T, (2,4,5-trichlorophenoxy) acetic acid; DCPA, dimethyltetrachloroterephthalate. µg/L, microgram per liter; µg/kg, microgram per kilogram. --, no data]

Pesticide	Parameter code	NWIS				STORET	
		Ground water		Surface water		Surface water	
		Sites	Samples	Sites	Samples	Sites	Samples
Organochlorine Insecticides							
Aldrin, dissolved	39331	25	25	7	12	0	0
Aldrin, total	39330	130	156	42	182	3	21
Aldrin, bottom material (µg/kg)	39333	0	0	82	200	4	4
Chlordane, dissolved	39352	25	25	7	12	0	0
Chlordane, total	39350	130	156	42	176	5	5
Chlordane, bottom material (µg/kg)	39351	0	0	82	200	2	2
Chlorobenzilate, total	39460	0	0	1	1	0	0
DDD, dissolved	39361	25	25	7	12	0	0
DDD, total	39360	130	156	42	184	3	21
DDD, bottom material (µg/kg)	39363	0	0	82	200	3	3
DDE, dissolved	39366	25	25	7	12	0	0
DDE, total	39365	130	156	42	184	3	21
DDE, bottom material (µg/kg)	39368	0	0	82	201	3	3
DDT, dissolved	39371	25	25	7	14	0	0
DDT, total	39370	130	156	42	184	3	21
DDT, bottom material (µg/kg)	39373	0	0	82	201	3	3
Dieldrin, dissolved	39381	25	25	7	12	0	0
Dieldrin, total	39380	130	156	42	184	3	21
Dieldrin, bottom material (µg/kg)	39383	0	0	82	201	0	0
Endosulfan, dissolved	82354	23	23	0	0	0	0
Endosulfan, total	39388	130	156	22	65	0	0
Endosulfan, bottom material (µg/kg)	39389	0	0	10	12	0	0
Endrin, dissolved	39391	25	25	7	12	0	0
Endrin, total	39390	130	156	42	183	5	23
Endrin, bottom material (µg/kg)	39393	0	0	82	201	0	0
Heptachlor, dissolved	39411	25	25	7	12	0	0
Heptachlor, total	39410	130	156	42	183	6	24
Heptachlor, bottom material (µg/kg)	39413	0	0	82	201	3	3
Heptachlor epoxide, dissolved	39421	25	25	7	12	0	0
Heptachlor epoxide, total	39420	130	156	42	150	5	23
Heptachlor epoxide, bottom material (µg/kg)	39423	0	0	82	201	3	3
Isodrin, dissolved	39431	0	0	2	2	0	0
Isodrin, total	39430	0	0	1	1	0	0
Lindane, dissolved	39341	25	25	7	14	0	0
Lindane, tota	39340	130	156	42	183	3	21
Lindane, bottom material (µg/kg)	39343	0	0	82	201	2	2

Table 24. Number of ground- and surface-water sites with pesticide water-quality data in the Connecticut, Housatonic, and Thames Rivers study unit—*Continued*

Pesticide	Parameter code	NWIS				STORET	
		Ground water		Surface water		Surface water	
		Sites	Samples	Sites	Samples	Sites	Samples
Organochlorine Insecticides—*Continued*							
Methoxychlor, dissolved	82350	23	23	0	0	0	0
Methoxychlor, total	39480	127	153	20	25	2	2
Methoxychlor, bottom material (µg/kg)	39481	0	0	13	15	1	1
Mirex, dissolved	39756	25	25	7	12	0	0
Mirex, total	39755	129	155	24	46	0	0
Mirex, bottom materials (µg/kg)	39758	0	0	10	11	0	0
Perthane, dissolved	82348	23	23	0	0	0	0
Perthane, total	39034	128	154	20	40	0	0
Perthane, bottom material (µg/kg)	81886	0	0	10	12	0	0
Toxaphene, dissolved	39401	25	25	7	12	0	0
Toxaphene, total	39400	130	156	42	150	2	2
Toxaphene, bottom material (µg/kg)	39403	0	0	83	202	2	2
Organophosphorus Insecticides							
Chlorpyrifos, total recoverable	--	110	110	0	0	0	0
DEF, total	39040	86	86	0	0	0	0
Diazinon, dissolved	39572	13	13	0	0	0	0
Diazinon, total	39570	126	149	27	106	0	0
Diazinon, bottom material (µg/kg)	39571	0	0	51	128	0	0
Disulfoton (Disyston), total	39011	88	88	0	0	0	0
Ethion, dissolved	82346	13	13	0	0	0	0
Ethion, total	39398	126	149	27	101	2	2
Ethion, bottom material (µg/kg)	39399	0	0	51	128	0	0
Fonofos, (Dyfonate), total	--	88	88	0	0	0	0
Malathion, dissolved	39532	13	13	0	0	0	0
Malathion, total	39530	126	149	27	107	0	0
Malathion, bottom material (µg/kg)	39531	0	0	51	128	0	0
Methyl parathion, dissolved	39602	13	13	0	0	0	0
Methyl parathion, total	39600	126	149	27	107	0	0
Methyl parathion, bottom material (µg/kg)	39601	0	0	51	128	2	2
Methyl trithion, dissolved	82344	13	13	0	0	0	0
Methyl trithion, total	39790	126	149	27	104	0	0
Methyl trithion, bottom material (µg/kg)	39791	0	0	51	128	0	0
Parathion, dissolved	39542	13	13	0	0	0	0
Parathion, total	39540	126	149	27	107	0	0
Parathion, bottom material	39541	0	0	51	128	0	0
Phorate, total	39023	88	88	0	0	0	0
Trithion, dissolved	82342	13	13	0	0	0	0
Trithion, total	39786	126	149	27	103	0	0
Trithion, bottom material (µg/kg)	39787	0	0	51	128	0	0

Table 24. Number of ground- and surface-water sites with pesticide water-quality data in the Connecticut, Housatonic, and Thames Rivers study unit—*Continued*

Pesticide	Parameter code	NWIS				STORET	
		Ground water		Surface water		Surface water	
		Sites	Samples	Sites	Samples	Sites	Samples
Carbamate Insecticides							
Aldicarb, total	82619	133	171	0	0	0	0
Aldicarb sulfone, total	82587	133	171	0	0	0	0
Aldicarb sulfoxide, total	82586	133	171	0	0	0	0
Carbofuran, total	82615	133	171	0	0	0	0
3-hydroxy Carbofuran, total	82584	133	171	0	0	0	0
Methomyl, total	39051	133	171	0	0	0	0
1-Naphthol, total	77441	133	171	0	0	0	0
Oxyamyl, total	82613	133	171	0	0	0	0
Propham, total	39052	133	171	0	0	0	0
Sevin, total	39750	133	171	0	0	0	0
Benzoic, Chlorophenoxy-Acid, and Pyridine Herbicides							
Dicamba, (Banvel), total	82052	57	97	0	0	0	0
2,4-D, dissolved	39732	13	13	7	12	0	0
2,4-D, total	39730	149	191	27	106	2	2
2,4-D, bottom material (µg/kg)	39731	0	0	54	102	1	1
2,4-DP, dissolved	82356	13	13	0	0	0	0
2,4-DP, total	82183	149	192	7	9	0	0
Picloram, total	39720	52	92	0	0	0	0
Silvex, dissolved	39762	13	13	7	10	0	0
Silvex, total	39760	149	192	27	107	0	0
Silvex, bottom material (µg/kg)	39761	0	0	54	102	0	0
2,4,5-T, dissolved	39742	13	13	7	10	0	0
2,4,5-T, tota	39740	149	192	27	106	2	2
2,4,5-T, bottom material (µg/kg)	39741	0	0	54	102	1	1
Acetamide, Triazine, and Trifluromethyl Herbicides							
Alachlor, dissolved	46342	0	0	3	28	0	0
Alachlor, total	77825	143	265	0	0	0	0
Ametryn, dissolved	38401	0	0	3	28	0	0
Ametryne, total	82184	143	265	0	0	0	0
Atrazine, dissolved	39633	0	0	3	28	0	0
Atrazine, total	39630	143	265	0	0	0	0
Cyanazine, dissolved	04041	0	0	3	28	0	0
Cyanazine, total	81757	143	265	0	0	0	0
Desethylatrazine, dissolved	04040	0	0	3	28	0	0
Deisopropylatrazine, dissolved	04038	0	0	3	28	0	0
Metolachlor, dissolved	39415	0	0	3	28	0	0
Metolachlor, total	82612	143	265	0	0	0	0
Metribuzin, dissolved	82630	0	0	3	28	0	0
Metribuzin, total	82611	143	265	0	0	0	0
Prometon, dissolved	04037	0	0	3	28	0	0
Prometon, total	39056	143	265	0	0	0	0

Table 24. Number of ground- and surface-water sites with pesticide water-quality data in the Connecticut, Housatonic, and Thames Rivers study unit—*Continued*

Pesticide	Parameter code	NWIS				STORET	
		Ground water		Surface water		Surface water	
		Sites	Samples	Sites	Samples	Sites	Samples
Acetamide, Triazine, and Trifluromethyl Herbicide—*Continued*							
Prometryn, dissolved	04036	0	0	3	28	0	0
Prometryn, total	39057	143	265	0	0	0	0
Propazine, dissolved	38535	0	0	3	28	0	0
Propazine, total	39024	143	265	0	0	0	0
Simazine, dissolved	04035	0	0	3	28	0	0
Simazine, total	39055	143	265	0	0	0	0
Simetryn, total	39054	143	265	0	0	0	0
Trifluralin, total	39030	143	265	0	0	0	0
Urea, Uracil, and Other Pesticides							
DCPA (Dacthal) plus metabolites, total	--	16	20	0	0	0	0
Diuron, total	--	7	7	0	0	0	0
Terbacil, total	--	5	5	0	0	0	0
Ethylene thiourea, total	--	11	11	0	0	0	0
Volatile Organic Compounds							
1,2-Dibromoethane, total	77651	8	8	0	0	0	0
1,2-Dibromoethylene, total	39082	129	146	2	2	0	0
1,1-Dichloroethane, total	34496	138	173	8	9	4	6
1,1-Dichloroethylene, total	34501	138	173	8	9	4	6
1,2-Dichloropropane, total	34541	146	181	8	9	4	6
1,3-Dichloropropene, total	34561	146	181	8	9	0	0
Semivolatile Compounds							
Acenaphthene, total	34205	11	11	0	0	0	0
Acenaphthene, bottom material (µg/kg)	34208)	0	0	14	16	0	0
Acenaphthylene, total	34200	11	11	0	0	0	0
Acenaphthylene, bottom material (µg/kg)	34203	0	0	14	16	0	0
1,2-Dichlorobenzene, total	34536	129	157	2	2	0	0
1,2-Dichlorobezene, bottom material (µg/kg)	34539	0	0	14	16	0	0
1,3-Dichlorobezene, total	34566	129	157	2	2	0	0
1,3-Dichlorobezene, bottom material (µg/kg)	34569	0	0	14	16	0	0
1,4-Dichlorobezene, total	34571	129	157	2	2	0	0
1,4-Dichlorobezene, bottom material (µg/kg)	34574	0	0	14	16	0	0
Diethylphthalate, total	34336	11	11	0	0	0	0
Diethylphthalate, bottom material (µg/kg)	34339	0	0	14	16	1	1
Dimethylphthalate, total	34341	11	11	0	0	0	0
Dimethylphthalate, bottom material (µg/kg)	34344	0	0	14	16	0	0
Hexachlorobenzene, total	39700	11	11	0	0	0	0
Hexachlorobenzene, bottom material (µg/kg)	39701	0	0	14	16	0	0
Naphthalene, total	34696	11	11	0	0	0	0
Pentachlorophenol, bottom material (µg/kg)	39061	0	0	14	16	0	0
1,2,4-Trichlorobenzene, total	34551	11	11	0	0	0	0
1,2,4-Trichlorobenzene, bottom material (µg/kg)	34554	0	0	14	16	0	0
2,4,6-Trichlorophenol, total	34621	11	11	0	0	0	0
2,4,6-Trichlorophenol, bottom material (µg/kg)	34624	0	0	14	16	0	0

Table 25. Summary statistics for selected nutrients and suspended-sediment data from the U.S. Geological Survey's National Water Information System (NWIS) for selected water-quality stations in the Connecticut, Housatonic, and Thames Rivers study unit, water years 1972-92

[Map No.: See plate 1 for location of water-quality stations and table 23 for summary of data. Quartile values are not necessarily representative of the annual conditions nor are they necessarily representative of the seasonal variations, because all months and years were not equally represented. For each constituent, stations are listed in order of increasing median concentrations. mg/L, milligram per liter; <, actual value is less than value shown]

Map No.	Station name	Period of record (water year)	Number of water-quality records	Percentage of observations in which concentrations were less than or equal to those shown		
				75	50	25
Nitrogen, total, as N (mg/L)						
38	Black River at Weathersfield, Vt.	1977-81	27	0.37	0.31	0.24
37	Black River near Hawk Mountain, below Cavendish, Vt.	1977-81	28	.43	.33	.27
39	Black River at Tolles Hill Dam near Weathersfield, Vt.	1977-81	28	.44	.34	.28
40	Black River at North Springfield, Vt.	1977-81	28	.50	.36	.31
34	Black River above Ludlow, Vt.	1979-81	16	.47	.38	.30
41	Black River at Gilman Dam, in Springfield, Vt.	1977-81	28	.56	.40	.32
36	Black River below Cavendish, Vt.	1977-81	28	.53	.43	.36
8	Burlington Brook near Burlington, Conn.	1972-92	208	.60	.43	.33
50	West Branch Swift River near Shutesbury, Mass.	1984-85	58	.66	.43	.33
35	Black River above Cavendish, Vt.	1977-81	28	.59	.45	.35
17	Saugatuck River near Redding, Conn.	1972-92	382	.69	.50	.37
9	Farmington River at State Highway 4, at Unionville, Conn.	1974-91	320	.70	.52	.41
45	Connecticut River near Northfield, Mass.	1975-80	54	.62	.53	.44
19	Natchaug River at U.S. Highway 6, at Willimantic, Conn.	1974-80	75	.63	.55	.48
42	Black River below Springfield, Vt.	1977-81	28	.72	.58	.47
12	Salmon River near East Hampton, Conn.	1972-92	207	.80	.60	.45
46	Connecticut River at Holyoke, Mass.	1975-80	54	.71	.61	.52
44	Connecticut River at Walpole, N.H.	1975-80	62	.77	.62	.44
49	East Branch Fever Brook near Petersham, Mass.	1989-81	27	.98	.63	.44
31	Connecticut River at Wells River, Vt.	1975-91	23	.76	.65	.51
76	Shepaug River near Roxbury, Conn.	1974-92	191	.80	.69	.50
55	Connecticut River at Agawam, Mass.	1973-80	54	.91	.72	.57
61	Porter Brook near Manchester, Conn.	1975-81	74	.88	.76	.64
1	Willimantic River at Merrow, Conn.	1974-92	201	1.0	.76	.60
81	Norwalk River at Cannondale, Conn.	1976-78	22	.96	.76	.64
78	East Aspetuck River (inflow site 23) at New Preston, Conn.	1974-76	28	.92	.80	.72
80	Norwalk River at Georgetown, Conn.	1976-78	22	1.0	.81	.61
82	Norwalk River at South Wilton, Conn.	1976-78	22	.99	.81	.57
29	Yantic River at Yantic, Conn.	1972-80	98	1.0	.86	.70
14	Housatonic River near New Milford, Conn.	1974-91	185	1.1	.87	.73
15	Housatonic River at Stevenson, Conn.	1972-92	244	1.1	.88	.73
2	Shetucket River at South Windham, Conn.	1974-92	234	1.1	.90	.70
18	Norwalk River at Winnipauk, Conn.	1981-92	152	1.2	.90	.79
3	Quinebaug River at Quinebaug, Conn.	1981-92	127	1.2	.90	.70
6	Connecticut River at Thompsonville, Conn.	1972-92	251	1.1	.90	.71
58	Farmington River at Windsor, Conn.	1974-76	27	1.1	.94	.83
68	Housatonic River near Canaan, Conn.	1974-83	110	1.2	.96	.78
77	Pomperaug River at Southbury, Conn.	1972-79	16	1.1	.99	.78
7	Still River at Riverton, Conn.	1974-91	189	1.6	1.0	.68
20	Shetucket River near Willimantic, Conn.	1972-74	23	1.4	1.0	.85
22	Quinebaug River at Putnam, Conn.	1974-80	75	1.4	1.0	.81
57	Farmington River at Avon, Conn.	1974-91	91	1.5	1.1	.86

Table 25. Summary statistics for selected nutrients and suspended-sediment data from the U.S. Geological Survey's National Water Information System (NWIS) for selected water-quality stations in the Connecticut,Housatonic, and Thames Rivers study unit, water years 1972-92—*Continued*

Map No.	Station name	Period of record (water year)	Number of water-quality records	Percentage of observations in which concentrations were less than or equal to those shown		
				75	50	25
Nitrogen, total, as N (mg/L)—*Continued*						
4	Quinebaug River at Jewett City, Conn.	1972-92	246	1.3	1.1	0.90
67	Housatonic River near Sodom, Conn.	1984-91	85	1.4	1.1	.90
28	Quinebaug River at Pomfret Landing, Conn.	1974-80	75	1.5	1.2	.90
11	Farmington River at Tariffville, Conn.	1972-92	224	1.5	1.2	.94
56	Stony Brook near West Suffield, Conn.	1981-91	76	1.6	1.2	.99
60	Park River at Hartford, Conn.	1974-79	61	1.5	1.3	1.1
25	Tributary to Mill Brook at South Woodstock, Conn.	1981-83	19	2.1	1.5	1.1
26	Tributary to Peake Brook at South Woodstock, Conn.	1982-83	52	2.0	1.5	1.1
21	French River at Mechanicsville, Conn.	1974-91	196	2.3	1.6	1.1
24	Tributary to Mill Brook at Woodstock, Conn.	1981-83	17	1.95	1.6	.90
79	Naugatuck River near Waterville, Conn.	1981-92	133	3.1	2.2	1.3
59	Podunk River at South Windsor, Conn.	1975-81	49	2.8	2.3	1.85
27	Tributary to Peake Brook at Harrisville, Conn.	1982-83	49	3.95	2.8	1.95
65	Quinnipiac River near Meriden, Conn.	1974-92	208	3.8	2.9	2.1
13	Quinnipiac River at Wallingford, Conn.	1972-92	261	4.0	3.3	2.5
16	Naugatuck River at Beacon Falls, Conn.	1974-92	232	5.2	3.4	2.13
75	Still River at Brookfield Junction, Conn.	1974-92	206	5.95	3.9	2.5
10	Pequabuck River at Farmington, Conn.	1974-92	207	6.3	4.2	2.7
Ammonia, dissolved, as N (mg/L)						
50	West Branch Swift River near Shutesbury, Mass.	1983-85	46	.03	<.01	<.01
64	Dickinson Creek near Marlborough, Conn.	1986-90	50	.032	.01	.006
63	Schroeder Brook at South Marlborough, Conn.	1986-90	49	.027	.02	.007
47	Natty Pond Brook at Templeton Road near Hubbardston, Mass.	1985-87	16	.04	.02	.01
12	Salmon River near East Hampton, Conn.	1972-92	67	.06	.02	.01
76	Shepaug River near Roxbury, Conn.	1974-92	51	.05	.02	.01
18	Norwalk River at Winnipauk, Conn.	1981-92	54	.04	.02	.01
8	Burlington Brook near Burlington, Conn.	1972-92	67	.06	.02	.01
52	Dickey Brook near Cooleyville, Mass.	1985-89	82	.06	.02	<.01
53	Dickey Brook tributary near Cooleyville, Mass.	1985-89	80	.04	.02	<.01
44	Connecticut River at Walpole, N.H.	1975-80	12	.06	.03	.012
59	Podunk River at South Windsor, Conn.	1975-81	37	.05	.03	.02
17	Saugatuck River near Redding, Conn.	1972-92	67	.07	.03	.01
19	Natchaug River at U.S. Highway 6, at Willimantic, Conn.	1974-80	51	.07	.03	.01
61	Porter Brook near Manchester, Conn.	1975-81	39	.05	.03	.01
82	Norwalk River at South Wilton, Conn.	1976-78	19	.05	.03	.02
5	Connecticut River at North Walpole, N.H.	1980-91	52	.04	.03	.01
81	Norwalk River at Cannondale, Conn.	1976-78	22	.055	.04	.03
49	East Branch Fever Brook near Petersham, Mass.	1983-85	45	.08	.04	<.01
9	Farmington River at Unionville, Conn.	1974-91	51	.07	.04	.02
14	Housatonic River near New Milford, Conn.	1974-92	51	.06	.04	.02
1	Willimantic River at Merrow, Conn.	1974-92	51	.12	.04	.01
31	Connecticut River at Wells River, Vt.	1980-86	37	.06	.04	.01
29	Yantic River at Yantic, Conn.	1972-80	75	.11	.05	.01
48	Natty Pond Brook near Hubbardston, Mass.	1985-86	16	.12	.05	.02
80	Norwalk River at Georgetown, Conn.	1976-78	21	.12	.06	.035

Table 25. Summary statistics for selected nutrients and suspended-sediment data from the U.S. Geological Survey's National Water Information System (NWIS) for selected water-quality stations in the Connecticut,Housatonic, and Thames Rivers study unit, water years 1972-92—*Continued*

Map No.	Station name	Period of record (water year)	Number of water-quality records	Percentage of observations in which concentrations were less than or equal to those shown		
				75	50	25
Ammonia, dissolved, as N (mg/L)—*Continued*						
68	Housatonic River near Canaan, Conn.	1974-83	51	0.14	0.08	0.04
15	Housatonic River at Stevenson, Conn.	1972-92	196	.13	.08	.05
78	East Aspetuck River (inflow site 23) at New Preston, Conn.	1977-78	27	.15	.08	.05
4	Quinebaug River at Jewett City, Conn.	1972-92	177	.17	.08	.04
77	Pomperaug River at Southbury, Conn.	1972-79	19	.15	.09	.06
51	Dickey Brook tributary #2 (site 7) near Cooleyville, Mass.	1988-89	17	.26	.10	.03
2	Shetucket River at South Windham, Conn.	1974-92	161	.14	.10	.07
6	Connecticut River at Thompsonville, Conn.	1972-92	188	.21	.12	.07
60	Park River at Hartford, Conn.	1974-79	50	.192	.12	.058
58	Farmington River at Windsor, Conn.	1974-76	27	.26	.13	.09
28	Quinebaug River at Pomfret Landing, Conn.	1974-80	51	.22	.14	.09
22	Quinebaug River at Putnam, Conn.	1974-80	51	.21	.15	.09
11	Farmington River at Tariffville, Conn.	1972-92	70	.28	.16	.11
7	Still River at Riverton, Conn.	1974-91	51	.32	.17	.10
21	French River at Mechanicsville, Conn.	1974-91	51	.39	.18	.10
20	Shetucket River near Willimantic, Conn.	1972-74	25	.36	.19	.115
57	Farmington River at Avon, Conn.	1974-91	39	.45	.31	.19
13	Quinnipiac River at Wallingford, Conn.	1972-92	129	.97	.34	.11
65	Quinnipiac River near Meriden, Conn.	1974-92	51	1.1	.54	.28
16	Naugatuck River at Beacon Falls, Conn.	1974-92	105	2.0	1.0	.565
75	Still River at Brookfield Junction, Conn.	1974-92	51	1.6	1.1	.73
10	Pequabuck River at Farmington, Conn.	1974-92	51	4.7	1.9	1.0
Nitrate, dissolved, as N (mg/L)						
49	East Branch Fever Brook near Petersham, Mass.	1983-85	48	.05	.02	<.05
50	West Branch Swift River near Shutesbury, Mass.	1983-85	49	.04	.02	.01
52	Dickey Brook near Cooleyville, Mass.	1985-89	72	.08	.03	.01
53	Dickey Brook tributary near Cooleyville, Mass.	1985-89	70	.06	.03	.02
51	Dickey Brook tributary #2 (site 7) near Cooleyville, Mass.	1988-89	17	.14	.10	.04
63	Schroeder Brook at South Marlborough, Conn.	1986-90	46	.31	.16	.10
17	Saugatuck River near Redding, Conn.	1972-92	16	.24	.19	.09
32	Connecticut River at Wilder, Vt.	1972-73	21	.40	.27	.20
8	Burlington Brook near Burlington, Conn.	1972-92	18	.30	.28	.20
12	Salmon River near East Hampton, Conn.	1972-92	16	.47	.30	.20
6	Connecticut River at Thompsonville, Conn.	1972-92	35	.44	.38	.30
2	Shetucket River at South Windham, Conn.	1974-92	18	.59	.39	.30
15	Housatonic River at Stevenson, Conn.	1972-92	44	.59	.45	.33
64	Dickinson Creek near Marlborough, Conn.	1986-90	47	1.0	.45	.30
20	Shetucket River near Willimantic, Conn.	1972-74	16	.59	.46	.40
4	Quinebaug River at Jewett City, Conn.	1972-92	33	.60	.49	.40
11	Farmington River at Tariffville, Conn.	1972-92	18	.75	.55	.50
29	Yantic River at Yantic, Conn.	1972-80	16	.78	.55	.44
77	Pomperaug River at Southbury, Conn.	1972-79	16	.98	.77	.51
13	Quinnipiac River at Wallingford, Conn.	1972-92	22	2.36	2.0	1.45

Table 25. Summary statistics for selected nutrients and suspended-sediment data from the U.S. Geological Survey's National Water Information System (NWIS) for selected water-quality stations in the Connecticut,Housatonic, and Thames Rivers study unit, water years 1972-92—*Continued*

Map No.	Station name	Period of record (water year)	Number of water-quality records	Percentage of observations in which concentrations were less than or equal to those shown		
				75	50	25
Nitrite plus nitrate, total, as N (mg/L)						
49	East Branch Fever Brook near Petersham, Mass.	1983-85	56	0.03	0.01	<0.01
50	West Branch Swift River near Shutesbury, Mass.	1983-85	56	.03	.01	<.01
47	Natty Pond Brook at Templeton Road near Hubbardston, Mass.	1985-88	16	.10	<.1	<.1
48	Natty Pond Brook near Hubbardston, Mass.	1985-87	16	.20	<.1	<.1
38	Black River at Weathersfield, Vt.	1977-81	27	.20	.11	.06
17	Saugatuck River near Redding, Conn.	1972-92	218	.20	.12	.07
37	Black River near Hawk Mountain, below Cavendish, Vt.	1977-81	28	.19	.13	.06
39	Black River at Tolles Hill Dam near Weathersfield, Vt.	1977-81	28	.21	.14	.08
40	Black River at North Springfield, Vt.	1977-80	28	.17	.14	.08
41	Black River at Gilman Dam, Springfield, Vt.	1977-81	28	.22	.16	.12
34	Black River above Ludlow, Vt.	1979-81	16	.22	.17	.11
8	Burlington Brook near Burlington, Conn.	1972-92	131	.22	.18	.11
9	Farmington River at State Highway 4 at Unionville, Conn.	1974-91	197	.25	.20	.16
76	Shepaug River near Roxbury, Conn.	1974-92	209	.32	.20	.10
42	Black River below Springfield, Vt.	1977-81	28	.32	.21	.18
36	Black River below Cavendish, Vt.	1977-81	28	.28	.21	.16
12	Salmon River near East Hampton, Conn.	1972-92	218	.33	.21	.16
45	Connecticut River near Northfield, Mass.	1975-80	54	.33	.21	.17
35	Black River above Cavendish, Vt.	1977-81	28	.31	.22	.15
19	Natchaug River at U.S. Highway 6, at Willimantic, Conn.	1974-80	75	.29	.22	.16
44	Connecticut River at Walpole, N.H.	1975-80	65	.36	.25	.19
46	Connecticut River at Holyoke, Mass.	1975-80	54	.33	.27	.20
31	Connecticut River at Wells River, Vt.	1979-81	23	.34	.29	.22
5	Connecticut River at North Walpole, N.H.	1981-92	17	.40	.29	.22
7	Still River at Riverton, Conn.	1974-91	198	.60	.30	.20
3	Quinebaug River at Quinebaug, Conn.	1981-92	133	.48	.30	.20
1	Willimantic River at Merrow, Conn.	1974-92	208	.46	.30	.22
55	Connecticut River at Agawam, Mass.	1973-80	54	.36	.31	.22
80	Norwalk River at Georgetown, Conn.	1976-78	22	.53	.33	.23
81	Norwalk River at Cannondale, Conn.	1976-78	22	.55	.34	.23
6	Connecticut River at Thompsonville, Conn.	1972-92	245	.42	.35	.30
78	East Aspetuck River (inflow site 23) at New Preston, Conn.	1977-78	28	.46	.35	.28
2	Shetucket River at South Windham, Conn.	1974-92	238	.50	.40	.30
22	Quinebaug River at Putnam, Conn.	1974-80	75	.51	.40	.28
14	Housatonic River near New Milford, Conn.	1974-91	197	.50	.40	.27
15	Housatonic River at Stevenson, Conn.	1972-92	244	.50	.40	.30
82	Norwalk River at South Wilton, Conn.	1976-78	22	.56	.41	.27
18	Norwalk River at Winnipauk, Conn.	1981-92	161	.60	.42	.30
4	Quinebaug River at Jewett City, Conn.	1972-92	246	.53	.44	.30
68	Housatonic River near Canaan, Conn.	1974-83	110	.60	.45	.32
57	Farmington River at Avon, Conn.	1974-91	91	.60	.45	.35
28	Quinebaug River at Pomfret Landing, Conn.	1974-80	75	.61	.46	.35
29	Yantic River at Yantic, Conn.	1972-80	84	.63	.48	.37
61	Porter Brook near Manchester, Conn.	1975-81	75	.59	.50	.42
67	Housatonic River near Sodom, Conn.	1984-91	87	.70	.50	.40

Table 25. Summary statistics for selected nutrients and suspended-sediment data from the U.S. Geological Survey's National Water Information System (NWIS) for selected water-quality stations in the Connecticut,Housatonic, and Thames Rivers study unit, water years 1972-92—*Continued*

Map No.	Station name	Period of record (water year)	Number of water-quality records	Percentage of observations in which concentrations were less than or equal to those shown		
				75	50	25
Nitrite plus nitrate, total, as N (mg/L)—*Continued*						
25	Tributary to Mill Brook at South Woodstock, Conn.	1982-83	21	1.1	0.55	0.30
58	Farmington River at Windsor, Conn.	1974-76	27	.67	.56	.49
26	Tributary to Peake Brook at South Woodstock, Conn.	1982-83	53	.64	.58	.40
21	French River at Mechanicsville, Conn.	1974-91	197	1.1	.60	.40
11	Farmington River at Tariffville, Conn.	1972-92	209	.74	.60	.46
56	Stony Brook near West Suffield, Conn.	1981-91	76	.80	.60	.40
24	Tributary to Mill Brook at Woodstock, Conn.	1982-83	19	1.2	.63	.30
60	Park River at Hartford, Conn.	1974-79	61	.88	.66	.49
27	Tributary to Peake Brook at Harrisville, Conn.	1982-83	52	.90	.80	.50
10	Pequabuck River at Farmington, Conn.	1974-92	209	1.5	1.0	.82
75	Still River at Brookfield Junction, Conn.	1974-92	209	1.7	1.2	.90
79	Naugatuck River near Waterville, Conn.	1981-92	133	2.05	1.2	.71
16	Naugatuck River at Beacon Falls, Conn.	1974-92	236	2.1	1.3	.90
65	Quinnipiac River near Meriden, Conn.	1974-92	208	2.2	1.7	1.2
13	Quinnipiac River at Wallingford, Conn.	1972-92	246	2.7	1.8	1.4
59	Podunk River at South Windsor, Conn.	1975-81	49	2.3	1.9	1.45
Nitrite plus nitrate, dissolved, as N (mg/L)						
47	Natty PondBrook at Templeton Road near Hubbardston, Mass.	1985-88	16	.13	<.1	<.1
48	Natty Pond Brook near Hubbardston, Mass.	1983-87	17	.14	.11	<.1
5	Connecticut River at North Walpole, N.H.	1981-91	52	.33	.23	.20
31	Connecticut River at Wells River, Vt.	1979-86	38	.33	.28	.20
44	Connecticut River at Walpole, N.H.	1979-80	13	.34	.29	.20
6	Connecticut River at Thompsonville, Conn.	1972-92	121	.44	.37	.31
2	Shetucket River at South Windham, Conn	1974-92	111	.50	.39	.30
15	Housatonic River at Stevenson, Conn.	1972-92	138	.50	.4	.30
18	Norwalk River at Winnipauk, Conn.	1981-92	54	.54	.41	.30
4	Quinebaug River at Jewett City, Conn.	1972-92	109	.54	.45	.35
16	Naugatuck River at Beacon Falls, Conn.	1974-92	54	2.8	1.6	1.08
13	Quinnipiac River at Wallingford, Conn.	1972-92	61	3.35	2.5	1.7
Phosphorus, total, as P (mg/L)						
47	Natty Pond Brook at Templeton Road near Hubbardston, Mass.	1985-87	16	.02	.01	<.01
34	Black River above Ludlow, Vt.	1979-81	16	.03	.01	<.01
31	Connecticut River at Wells River, Vt.	1979-86	44	.03	.01	.01
12	Salmon River near East Hampton, Conn.	1972-92	242	.03	.01	.01
8	Burlington Brook near Burlington, Conn.	1972-92	155	.03	.01	.01
36	Black River below Cavendish, Vt.	1977-81	28	.03	.01	.01
37	Black River below Cavendish, Vt.	1977-81	28	.02	.01	.01
38	Black River at Weathersfield, Vt.	1977-81	27	.02	.01	.01
39	Black River at Tolles Hill Dam near Weathersfield, Vt.	1977-81	28	.02	.01	<.01
5	Connecticut River at North Walpole, N.H.	1981-91	52	.02	.02	<.01
32	Connecticut River at Wilder, Vt.	1972-73	21	.02	.02	.02
44	Connecticut River at Walpole, N.H.	1981-91	65	.03	.02	.01
61	Porter Brook near Manchester, Conn.	1975-81	75	.03	.02	.01
35	Black River above Cavendish, Vt.	1977-81	28	.04	.02	.01
40	Black River at North Springfield, Vt.	1977-81	28	.03	.02	.01

Table 25. Summary statistics for selected nutrients and suspended-sediment data from the U.S. Geological Survey's National Water Information System (NWIS) for selected water-quality stations in the Connecticut,Housatonic, and Thames Rivers study unit, water years 1972-92—*Continued*

Map No.	Station name	Period of record (water year)	Number of water-quality records	Percentage of observations in which concentrations were less than or equal to those shown		
				75	50	25
Phosphorus, total, as P (mg/L)—*Continued*						
19	Natchaug River at U.S. Highway 6, at Willimantic, Conn.	1974-80	75	0.03	0.02	0.01
17	Saugatuck River near Redding, Conn.	1972-92	242	.03	.02	.01
72	Lake Waramaug Brook (inflow site 7) at Warren, Conn.	1977-78	27	.07	.02	.01
45	Connecticut River near Northfield, Mass.	1975-80	54	.03	.03	.02
9	Farmington River at State Highway 4 at Unionville, Conn.	1974-91	198	.04	.03	.02
76	Shepaug River near Roxbury, Conn.	1974-92	209	.04	.03	.02
29	Yantic River at Yantic, Conn.	1972-80	108	.04	.03	.02
41	Black River at Gilman Dam, in Springfield, Vt.	1977-81	28	.04	.03	.02
83	Tributary to Lake Waramaug (inflow site 9) near New Preston, Conn.	1977-78	32	.05	.03	.02
1	Willimantic River at Merrow, Conn.	1974-92	208	.05	.03	.02
59	Podunk River at South Windsor, Conn.	1975-81	49	.07	.04	.03
15	Housatonic River at Stevenson, Conn.	1972-92	269	.05	.04	.03
46	Connecticut River at Holyoke, Mass.	1975-80	54	,05	.04	.03
18	Norwalk River at Winnipauk, Conn.	1981-92	161	.07	.05	.04
42	Black River below Springfield, Vt.	1977-81	28	.07	.05	.03
82	Norwalk River at South Wilton, Conn.	1976-78	22	.07	.05	.03
14	Housatonic River near New Milford, Conn.	1974-91	197	.06	.05	.03
77	Pomperaug River at Southbury, Conn.	1972-79	36	.13	.05	.03
2	Shetucket River at South Windham, Conn.	1974-92	238	.09	.05	.04
6	Connecticut River at Thompsonville, Conn.	1972-92	264	.09	.06	.04
48	Natty Pond Brook near Hubbardston, Mass.	1985-87	16	.11	.06	.03
56	Stony Brook near West Suffield, Conn.	1981-91	75	.11	.06	.05
67	Housatonic River near Sodom, Conn.	1984-91	86	.08	.06	.03
73	Lake Waramaug Brook (inflow site 26) near Warren, Conn.	1977-78	23	.19	.06	.02
78	East Aspetuck River (inflow site 23) at New Preston, Conn.	1977-78	28	.07	.06	.05
55	Connecticut River at Agawam, Mass.	1973-80	55	.10	.07	.04
24	Tributary to Mill Brook at Woodstock, Conn.	1981-83	19	.13	.07	.06
68	Housatonic River near Canaan, Conn.	1974-83	110	.12	.08	.07
81	Norwalk River at Cannondale, Conn.	1976-78	22	.13	.08	.06
7	Still River at Riverton, Conn.	1974-91	198	.18	.09	.05
4	Quinebaug River at Jewett City, Conn.	1972-92	270	.14	.09	.06
60	Park River at Hartford, Conn.	1974-79	62	.16	.10	.07
80	Norwalk River at Georgetown, Conn.	1976-78	22	.14	.11	.06
74	Lake Waramaug Brook (inflow site 2) near New Preston, Conn.	1977-78	34	.27	.11	.03
20	Shetucket River near Willimantic, Conn.	1972-74	33	.17	.11	.07
11	Farmington River at Tariffville, Conn.	1972-92	235	.20	.14	.10
22	Quinebaug River at Putnam, Conn.	1974-80	75	.25	.14	.09
79	Naugatuck River near Waterville, Conn.	1981-92	133	.22	.13	.09
3	Quinebaug River at Quinebaug, Ct.	1981-92	133	.24	.13	.06
66	Housatonic River near Great Barrington, Mass.	1981-82	28	.20	.13	.09
26	Tributary to Peake Brook at South Woodstock, Conn.	1982-83	53	.18	.13	.05
28	Quinebaug River at Pomfret Landing, Conn.	1974-80	75	.23	.15	.09
57	Farmington River at Avon, Conn.	1974-91	91	.23	.16	.11

Table 25. Summary statistics for selected nutrients and suspended-sediment data from the U.S. Geological Survey's National Water Information System (NWIS) for selected water-quality stations in the Connecticut,Housatonic, and Thames Rivers study unit, water years 1972-92—*Continued*

Map No.	Station name	Period of record (water year)	Number of water-quality records	Percentage of observations in which concentrations were less than or equal to those shown		
				75	50	25
Phosphorus, total, as P (mg/L)—*Continued*						
58	Farmington River at Windsor, Conn.	1974-76	27	0.18	0.14	0.09
21	French River at Mechanicsville, Conn.	1974-91	197	.28	.17	.10
25	Tributary to Mill Brook at South Woodstock, Conn.	1982-83	21	.70	.28	.07
16	Naugatuck River at Beacon Falls, Conn.	1974-92	237	.67	.39	.21
75	Still River at Brookfield Junction, Conn.	1974-92	208	.63	.40	.26
65	Quinnipiac River near Meriden, Conn.	1974-92	208	.69	.42	.26
13	Quinnipiac River at Wallingford, Conn.	1972-92	270	.61	.43	.28
27	Tributary to Peake Brook at Harrisville, Conn.	1982-83	52	.93	.48	.27
10	Pequabuck River at Farmington, Conn.	1974-92	209	1.5	.80	.46
Phosphorus, dissolved, as P (mg/L)						
83	Tributary to Lake Waramaug (inflow site 9) near New Preston, Conn.	1977-78	18	.009	.006	.004
31	Connecticut River at Wells River, Vt.	1979-86	43	.01	<.01	<.01
8	Burlington Brook near Burlington, Conn.	1972-92	45	.01	<.01	<.01
5	Connecticut River at North Walpole, N.H.	1981-91	52	.01	<.01	<.01
44	Connecticut River at Walpole, N.H.	1975-80	36	.01	.01	<.01
17	Saugatuck River near Redding, Conn.	1972-92	132	.02	.01	<.01
12	Salmon River near East Hampton, Conn.	1972-92	134	.02	.01	<.01
76	Shepaug River near Roxbury, Conn.	1974-92	134	.02	.01	<.01
24	Tributary to Mill Brook at Woodstock, Conn.	1982-83	19	.03	.02	.02
9	Farmington River at State Highway 4 at Unionville, Conn.	1974-91	122	.03	.02	.01
14	Housatonic River near New Milford, Conn.	1974-91	122	.03	.02	.01
15	Housatonic River at Stevenson, Conn.	1972-92	197	.03	.02	.01
1	Willimantic River at Merrow, Conn.	1974-92	133	.03	.02	<.01
18	Norwalk River at Winnipauk, Conn.	1981-92	161	.04	.03	.02
26	Tributary to Peake Brook at South Woodstock, Conn.	1982-83	53	.05	.03	.02
67	Houousatonic River near Sodom, Conn.	1984-91	86	.04	.03	.01
6	Connecticut River at Thompsonville, Conn.	1972-92	198	.05	.03	.02
2	Shetucket River at South Windham, Conn	1974-92	181	.05	.03	.02
4	Quinebaug River at Jewett City, Conn.	1972-92	195	.07	.04	.03
68	Housatonic River near Canaan, Conn.	1974-83	35	.08	.05	.03
56	Stony Brook near West Suffield, Conn.	1981-91	75	.08	.05	.03
25	Tributary to Mill Brook at South Woodstock, Conn.	1982-83	21	.12	.06	.03
7	Still River at Riverton, Conn.	1974-91	121	.15	.07	.03
3	Quinebaug River at Quinebaug, Conn.	1981-92	133	.18	.08	.04
74	Lake Waramaug Brook (inflow site 2) near New Preston, Conn.	1977-78	20	.127	.08	.054
79	Naugatuck River near Waterville, Conn.	1981-92	133	.175	.09	.06
66	Housatonic River near Great Barrington, Mass.	1981-82	28	.137	.095	.06
11	Farmington River at Tariffville, Conn.	1972-92	134	.16	.10	.06
21	French River at Mechanicsville, Conn.	1974-91	122	.202	.11	.05
57	Farmington River at Avon, Conn.	1974-91	44	.218	.13	.072
27	Tributary to Peake Brook at Harrisville, Conn.	1982-83	52	.358	.22	.093
75	Still River at Brookfield Junction, Conn.	1974-92	134	.452	.265	.12
13	Quinnipiac River at Wallingford, Conn.	1972-92	161	.43	.28	.175

Table 25. Summary statistics for selected nutrients and suspended-sediment data from the U.S. Geological Survey's National Water Information System (NWIS) for selected water-quality stations in the Connecticut,Housatonic, and Thames Rivers study unit, water years 1972-92—*Continued*

Map No.	Station name	Period of record (water year)	Number of water-quality records	Percentage of observations in which concentrations were less than or equal to those shown		
				75	50	25
Phosphorus, dissolved, as P (mg/L)—*Continued*						
16	Naugatuck River at Beacon Falls, Conn.	1974-92	162	0.56	0.3	0.137
65	Quinnipiac River near Meriden, Conn.	1974-92	132	.588	.33	.183
10	Pequabuck River at Farmington, Conn.	1974-92	134	1.1	.66	.28
Phosphorus, orthophosphate, as P (mg/L)						
63	Schroeder Brook at South Marlborough, Conn.	1986-90	49	.003	.001	<.001
5	Connecticut River at North Walpole, N.H.	1982-91	42	<.01	<.01	<.01
31	Connecticut River at Wells River, Vt.	1982-86	21	<.01	<.01	<.01
47	Natty Pond Brook at Templeton Road near Hubbardston, Mass.	1985-87	16	<.01	<.01	<.01
49	East Branch Fever Brook near Petersham, Mass.	1983-85	59	<.01	<.01	<.01
50	West Branch Swift River near Shutesbury, Mass.	1983-85	58	<.01	<.01	<.01
51	Dickey Brook tributary #2 (site 7) near Cooleyville, Mass.	1988-89	17	<.01	<.01	<.01
52	Dickey Brook near Cooleyville, Mass.	1985-89	81	<.01	<.01	<.01
53	Dickey Brook tributary near Cooleyville, Mass.	1985-89	80	<.01	<.01	<.01
15	Housatonic River at Stevenson, Conn.	1972-92	112	.02	.01	<.01
2	Shetucket River at South Windham, Conn.	1974-92	92	.03	.02	.01
6	Connecticut River at Thompsonville, Conn.	1972-92	95	.04	.02	.01
18	Norwalk River at Winnipauk, Conn.	1981-92	54	.04	.02	<.01
24	Tributary to Mill Brook at Woodstock, Conn.	1982-83	19	.03	.02	.02
48	Natty Pond Brook near Hubbardston, Mass.	1983-87	17	.05	.02	<.01
64	Dickinson Creek near Marlborough, Conn.	1986-90	50	.036	.027	.013
4	Quinebaug River at Jewett City, Conn.	1972-92	90	.04	.03	.01
26	Tributary to Peake Brook at South Woodstock, Conn.	1982-83	53	.05	.03	.01
25	Tributary to Mill Brook at South Woodstock, Conn.	1982-83	21	.11	.06	.02
66	Housatonic River near Great Barrington, Mass.	1981-82	28	.137	.085	.04
27	Tributary to Peake Brook at Harrisville, Conn.	1982-83	52	.425	.24	.093
13	Quinnipiac River at Wallingford, Conn.	1972-92	60	.38	.26	.132
16	Naugatuck River at Beacon Falls, Conn.	1974-92	54	.615	.28	.108
Suspended-sediment concentration (mg/L)						
53	Dickey Brook tributary near Cooleyville, Mass.	1985-89	42	5.3	2.5	1
23	Muddy Brook at Childs Hill Road near Woodstock, Conn.	1980-83	142	5	3	2
30	Ammonoosuc River at Bethlehem Junction, N.H.	1972-74	15	7	3	1
54	West Branch Westfield River at Huntington, Mass.	1972-74	16	7.5	3.5	2
52	Dickey Brook near Cooleyville, Mass.	1985-89	42	8	3.5	2
12	Salmon River near East Hampton, Conn.	1972-92	321	7	4	2
43	Williams River at Brockways Mills, Vt.	1972-74	16	7	5	1.3
56	Stony Brook near West Suffield, Conn.	1981-91	347	8	5	2
62	Coginchaug River at Middlefield, Conn.	1981-82	387	9	5	3
80	Norwalk River at Georgetown, Conn.	1976-78	22	12.5	5	4
33	White River at West Hartford, Vt.	1972-74	28	11.5	5.5	3
44	Connecticut River at Walpole, N.H.	1975-80	56	12	6	2.3
81	Norwalk River at Cannondale, Conn.	1976-78	22	18.8	6	3
31	Connecticut River at Wells River, Vt.	1979-85	37	14	7	3.5
5	Connecticut River at North Walpole, N.H.	1981-91	47	13	7	4
15	Housatonic River at Stevenson, Conn.	1972-92	133	12	7	3

Table 25. Summary statistics for selected nutrients and suspended-sediment data from the U.S. Geological Survey's National Water Information System (NWIS) for selected water-quality stations in the Connecticut,Housatonic, and Thames Rivers study unit, water years 1972-92—*Continued*

Map No.	Station name	Period of record (water year)	Number of water-quality records	Percentage of observations in which concentrations were less than or equal to those shown		
				75	50	25
Suspended-sediment concentration (mg/L)—*Continued*						
82	Norwalk River at South Wilton, Conn.	1976-78	22	14.3	7.5	2.8
2	Shetucket River at South Windham, Conn	1974-92	65	13	8	5
66	Housatonic River near Great Barrington, Mass.	1979-87	187	12	9	5
6	Connecticut River at Thompsonville, Conn.	1972-92	126	17.3	10	6
4	Quinebaug River at Jewett City, Conn.	1972-92	74	14.3	10	7
69	Housatonic River at Falls Village, Conn.	1972-80	181	23.5	14	8
71	Housatonic River at Gaylordsville, Conn.	1979-80	224	20	14	10
24	Tributary to Mill Brook at Woodstock, Conn.	1982-83	19	37	16	11
70	Housatonic River at Kent, Conn.	1985-88	313	38.5	16	5
83	Tributary to Lake Waramaug (inflow site 9) near New Preston, Conn.	1977-78	22	34.3	26	10.3
72	Lake Waramaug Brook (inflow site 7) at Warren, Conn.	1977-78	17	148	29	15
26	Tributary to Peake Brook at South Woodstock, Conn.	1982-83	52	110.3	43	13
74	Lake Waramaug Brook (inflow site 2) near New Preston, Conn.	1977-78	21	200.5	45	18
27	Tributary to Peake Brook at Harrisville, Conn.	1982-83	52	446	101.5	26.3
25	Tributary to Mill Brook at South Woodstock, Conn.	1982-83	21	384.5	161	45

Table 26. Summary of median concentrations for selected nitrogen data from the U.S. Environmental Protection Agency's computer STOrage and RETrieval system (STORET) for selected water-quality stations in the Connecticut, Housatonic, and Thames Rivers study unit, water years 1972-90

[Map No.: See plate 1 for location of selected water-quality stations and table 23 for summary of data. Median concentrations are not necessarily representative of the annual conditions nor are they necessarily representative of the seasonal variations, because all months and years were not equally represented. Stations are listed in descending order of map no.. <, actual value is less than value shown; --, no data]

Map No.	Station name	Period of record (water year)	Nitrogen, total (mg/L as N)		Ammonia, total (mg/L as N)		Nitrate, total (mg/L as N)	
			Median	Samples	Median	Samples	Median	Samples
228	Wooster Brook at Waterbury, Conn.	1974-88	0.765	5	0.09	64	0.80	83
227	Beaver Brook at Bethlehem, N.H.	1979-90	.365	48	<.05	28	--	--
226	Cold Brook at Berlin, N.H.	1973-90	.28	31	<.05	4	--	--
225	Number 9 Brook at Berlin, N.H.	1979-90	.32	6	<.05	2	--	--
224	Small Brook at Berlin, N.H.	1979-83	.29	4	<.05	1	--	--
223	West Branch Upper Ammonoosuc River at Berlin, N.H.	1979-90	.19	16	--	--	--	--
222	Wild Ammonoosuc River at Benton, N.H.	1983-90	.26	11	--	--	--	--
221	Passumpsic River at East Barnet, Vt.	1985	.31	12	.05	13	--	--
220	Clough Brook at Lemington, Vt.	1984-85	--	--	--	--	--	--
219	Passumpsic River at St. Johnsbury, Vt.	1985	.31	13	.038	12	--	--
218	Moose River at Granby, Vt.	1984-85	--	--	--	--	--	--
217	Ompompanoosuc River at Union Village, Vt.	1972-88	--	--	<.05	77	.38	81
216	West Branch Ompompanoosuc River at Thetford, Vt.	1972-88	--	--	<.05	72	.39	60
215	East Branch Ompompanoosuc River at Thetford Center, Vt.	1972-88	--	--	<.05	72	.20	58
214	West Branch Ompompanoosuc River at South Strafford, Vt.	1979-88	--	--	<.05	66	.21	9
213	East Branch Ompompanoosuc River at West Fairlee, Vt.	1979-88	--	--	<.05	30	.18	10

Table 26. Summary of median concentrations for selected nitrogen data from the U.S. Environmental Protection Agency's computer STOrage and RETrieval system (STORET) for selected water-quality stations in the Connecticut, Housatonic, and Thames Rivers study unit, water years 1972-90—*Continued*

Map No.	Station name	Period of record (water year)	Nitrogen, total (mg/L as N)		Ammonia, total (mg/L as N)		Nitrate, total (mg/L as N)	
			Median	Samples	Median	Samples	Median	Samples
212	Harvey's Tributary #2 at Barnet, Vt.	1981-82	--	--	--	--	--	--
211	Steven's River at Barnet, Vt.	1980	--	--	--	--	--	--
210	South Peacham Brook at Barnet, Vt.	1980-82	--	--	--	--	--	--
209	South Peacham Brook at South Peacham, Vt.	1982	0.79	10	0.004	10	--	--
208	South Peacham Brook at Peacham, Vt.	1982	.645	10	.004	10	--	--
207	Ottauquechee River at North Hartland, Vt.	1972-82	.90	1	.06	31	0.48	96
206	Connecticut River at Hinsdale, N.H.	1980-86	--	--	--	--	--	--
205	Connecticut River at Lancaster, N.H.	1978-89	.48	28	<.10	32	.41	2
204	Connecticut River at Cornish, N.H.	1976-81	.52	37	<.10	41	--	--
203	Connecticut River at Hinsdale, N.H.	1980-86	--	--	--	--	--	--
202	North Branch Gale River at Bethlehem, N.H.	1976-90	.535	14	<.05	16	--	--
201	Garland Brook at Lancaster, N.H.	1980-90	.47	31	<.05	30	--	--
200	Morey Tributary #3 at Fairlee, Vt.	1981-82	--	--	--	--	--	--
199	Scarface Brook at Bethlehem, N.H.	1982-90	.24	17	--	--	--	--
198	Clark Brook at Granville, Vt.	1984-85	--	--	--	--	--	--
197	Hancock Branch tributary to White River at Hancock, Vt.	1984-85	--	--	--	--	--	--
196	White River at Granville, Vt.	1984-85	--	--	--	--	--	--
195	Third Branch White River at Roxbury, Vt.	1982	.53	10	.03	10	--	--
194	Third Branch White River at Roxbury, Vt.	1982	.395	10	.008	10	--	--
193	North Branch Black River at Weathersfield, Vt.	1980-88	--	--	<.05	29	--	--
192	Ottauquechee River at North Hartland, Vt.	1981-88	--	--	.02	15	.21	3
191	Ottauquechee River at Quechee Village, Vt.	1981-82	--	--	<.05	15	--	--
190	Ottauquechee River at Quechee Gorge, Vt.	1980-88	--	--	.03	40	.21	3
189	Black River at Perkinsville, Vt.	1972-86	.70	2	<.05	32	.22	70
188	North Branch Black River at Amsden, Vt.	1972-86	.58	1	<.05	31	.30	73
187	Ottauquechee River at Quechee, Vt.	1974-88	.78	1	<.05	37	.29	58
186	West River at North Springfield, Vt.	1972-86	.91	1	.02	35	.40	99
185	North Branch Ball Mountain Brook at Jamaica, Vt.	1988	.57	2	<.02	7	--	--
184	West River at Stratton, Vt.	1988	.79	4	<.02	7	--	--
183	Ball Mountain Brook at West Jamaica, Vt.	1984-85	--	--	--	--	--	--
182	West River at Townshend, Vt.	1981-86	--	--	.06	10	--	--
181	West River at East Jamaica, Vt.	1972-86	--	--	<.05	14	.18	71
180	Wardsboro Brook at Jamaica, Vt.	1976-86	--	--	<.05	14	.12	24
179	North Branch Ball Mountain Brook at Jamaica, Vt.	1985-88	.795	14	<.02	23	--	--
178	Winhall River at Londonderry, Vt.	1977-86	--	--	<.05	12	.13	11
177	West River at South Londonderry, Vt.	1980-86	--	--	<.05	12	--	--
176	Winhall River at near Rawsonville, Vt.	1980-86	--	--	<.05	12	--	--
175	Winhall River at Winhall, Vt.	1982-85	--	--	--	--	--	--
174	Kidder Brook at Jamaica, Vt.	1985-87	.25	5	<.02	16	--	--
173	Ashuelot River at Gilsum, N.H.	1980-86	--	--	<.05	21	--	--
172	Ashuelot River below Surry Mountain Dam, at Surry, N.H.	1972-86	--	--	<.05	23	.18	99
171	Otter Brook below Otter Brook Dam at Keene, N.H.	1972-86	--	--	<.05	18	.14	96
170	Otter Brook at Keene, N.H.	1972-86	--	--	<.05	18	.10	79
169	Merriam Brook at Surry, N.H.	1973-86	--	--	<.05	18	.27	73
168	Merriam Brook at Surry, N.H.	1975-86	--	--	<.05	18	.19	53
167	Ashuelot River at Swanzey, N.H.	1976-89	.76	35	.25	41	.31	2
166	East Branch Tully River at Tully Dam in Royalston, Mass.	1980-86	--	--	<.05	14	--	--
165	Miller River at Winchendon, Mass.	1980-85	--	--	.24	25	--	--

Table 26. Summary of median concentrations for selected nitrogen data from the U.S. Environmental Protection Agency's computer STOrage and RETrieval system (STORET) for selected water-quality stations in the Connecticut, Housatonic, and Thames Rivers study unit, water years 1972-90—*Continued*

Map No.	Station name	Period of record (water year)	Nitrogen, total (mg/L as N)		Ammonia, total (mg/L as N)		Nitrate, total (mg/L as N)	
			Median	Samples	Median	Samples	Median	Samples
164	Priest Brook at Royalston, Mass.	1980-85	--	--	0.02	25	--	--
163	Lawrence Brook below Doanes Falls, Royalston, Mass.	1976-86	--	--	<.05	13	0.14	16
162	Miller River below Birch Hill Dam, Royalston, Mass.	1972-85	--	--	.18	25	.89	6
161	Otter River at Baldwinsville, Mass.	1976-85	--	--	.55	25	.754	16
160	East Branch Tully River at Royalston, Mass.	1980-86	--	--	<.05	14	--	--
159	North Branch Deerfield River at Wilmington, Vt.	1985	0.51	13	.30	13	--	--
158	Binney Brook at Wilmington, Vt.	1985	.26	3	.03	14	--	--
157	North Branch Deerfield River at Wilmington, Vt.	1985	.295	10	.04	14	--	--
156	North Branch Deerfield River at Wilmington, Vt.	1985	.52	12	.30	13	--	--
155	Deerfield River at Somerset, Vt.	1984-85	--	--	--	--	--	--
154	Conant Brook at Monson, Mass.	1973-87	--	--	<.05	15	--	--
153	Vinica Brook at South Monson, Mass.	1972-87	--	--	.04	18	.17	55
152	Westfield River below Knightville Dam at Huntington, Mass.	1973-86	--	--	.04	16	.30	79
151	Westfield River at Huntington, Mass.	1980-88	--	--	<.05	11	.10	1
150	Little River at Huntington, Mass.	1973-86	--	--	<.05	11	.39	38
149	West Branch Farmington River below Goodwin Dam in Hartland, Conn.	1981-86	--	--	.04	8	--	--
148	Quinebaug River at Thompson, Conn.	1975-88	--	--	.03	65	.42	65
147	Quinebaug River at Fabyan, Conn.	1972-88	--	--	.08	43	.63	75
146	Quinebaug River at Fabyan, Conn.	1978-88	--	--	.13	66	.39	23
145	Quinebaug River below West Thompson Dam at Thompson, Conn.	1972-88	--	--	.05	73	.49	110
144	Quinebaug River at Thompson, Conn.	1981-88	--	--	.11	39	.62	7
143	Quinebaug River at Thompson, Conn.	1972-88	--	--	.05	42	.59	90
142	Mill Brook at Brimfield, Mass.	1980-88	--	--	<.05	39	.15	3
141	Quinebaug River at Holland, Mass.	1980-88	--	--	<.05	42	.07	3
140	Quinebaug River at East Brimfield, Mass.	1972-88	--	--	<.05	42	.33	42
139	French River at Oxford, Mass.	1980-87	--	--	.05	13	--	--
138	Quinebaug River below East Brimfield Dam, Sturbridge, Mass.	1972-88	--	--	<.05	44	.43	68
137	Tributary to Quinebaug River at Quinebaug, Conn.	1982-88	--	--	.12	37	.28	7
136	French River below Hodges Village Dam, Oxford, Mass.	1972-87	--	--	.11	24	.76	93
135	Little River below Buffumville Dam, Oxford, Mass.	1972-88	--	--	<.05	51	.44	85
134	Little River at Oxford, Mass.	1972-88	--	--	.03	48	.60	73
133	Quinebaug River below Westville Dam, Southbridge, Mass.	1972-88	--	--	<.05	34	.60	76
132	Quinebaug River at Sturbridge, Mass.	1972-88	--	--	.06	29	.60	53
131	South Fork Little River at Oxford, Mass.	1972-88	--	--	<.05	50	.26	82
130	Natchaug River at Mansfield Hollow, Conn.	1972-86	--	--	.14	18	.41	81
129	Mount Hope River at Mansfield, Conn.	1980-86	--	--	<.05	9	--	--
128	Fenton River at Mansfield, Conn.	1980-86	--	--	.04	9	--	--
127	Naugatuck River at Thomaston, Conn.	1972-85	--	--	.16	37	.86	101
126	Northfield Brook at Thomaston, Conn.	1972-88	--	--	.03	28	.34	102
125	Leadmine Brook at Thomaston, Conn.	1972-87	--	--	.01	31	.35	72
124	Naugatuck River at Campville, Conn.	1972-85	--	--	.48	30	.81	74
123	Northfield Brook at Thomaston, Conn.	1972-88	--	--	<.05	23	.37	87
122	Confluence of Hop and Shattuck Brooks at Waterbury, Conn.	1972-88	1.30	5	.03	61	.43	12
121	Hop Brook Dam at Waterbury, Conn.	1972-87	--	--	.02	15	.75	72
120	Welton Brook at Waterbury, Conn.	1974-88	1.10	5	<.05	60	.70	78

Table 27. Land-use characteristics of the drainage basins of selected stream water-quality sampling stations in the

[Map No.: See plate 1 for location of water-quality sampling stations and table 23 for summary of data. Land-use categories from Anderson and others (Mitchell and others, 1977). Detailed land-use data not available for Farmington River at State Highway 4, at Unionville, Conn. (map reference No. 9).

Map No.	Station No.	Station name	Drainage area (mi^2)	Land use			
				Resi-dential	Commercial and services	Transpor-tation and utilities	Mixed and other urban
1	01119375	Willimantic River at Merrow, Conn.	94.0	4.2	0.5	0.7	0.8
2	01122610	Shetucket River at South Windham, Conn.	408	5.9	1.0	.6	.7
3	01124000	Quinebaug River at Quinebaug, Conn.	155	5.5	.8	1.9	.8
4	01127000	Quinebaug River at Jewett City, Conn.	713	4.0	.7	1.0	1.2
5	01154500	Connecticut River at North Walpole, N.H.	5,490	.7	.1	.6	.4
6	01184000	Connecticut River at Thompsonville, Conn.	9,660	2.6	.6	.6	.6
7	01186800	Still River at Riverton, Conn.	86.2	4.1	1.2	.7	.3
8	01188000	Burlington Brook near Burlington, Conn.	4.10	0	1.0	0	0
10	01189030	Pequabuck River at Farmington, Conn.	57.2	22.2	4.7	.2	5.3
11	01189995	Farmington River at Tariffville, Conn.	577	7.7	1.2	.2	1.6
12	01193500	Salmon River near East Hampton, Conn.	100	5.5	.8	1.5	1.0
13	01196500	Quinnipiac River at Wallingford, Conn.	115	23.9	5.3	2.5	4.7
14	01200600	Housatonic River near New Milford, Conn.	1,022	5.1	.7	.4	1.0
15	01205500	Housatonic River at Stevenson, Conn.	1,544	7.1	1.0	.4	1.1
16	01208500	Naugatuck River at Beacon Falls, Conn.	260	15.2	3.1	1.1	2.7
17	01208990	Saugatuck River near Redding, Conn.	21.0	19.8	.7	0	1.6
18	01209710	Norwalk River at Winnipauk, Conn.	33.0	41.6	2.0	0	3.1

(1976). Land-use data from mid-1970's extracted using the U.S. Geological Survey's Geographic Information Retrieval and Analysis System (GIRAS)
<, actual value is less than value shown]

Percentage of drainage area in land-use classes											
Cropland and pasture	Orchards	Other agricul-tural	Deciduous forest	Evergreen forest	Mixed forest	Streams	Lakes and reservoirs	Forested and non-forested wetlands	Mines and gravel pits	Transi-tional areas	Other
7.1	0	<0.1	50.6	3.2	29.6	0	1.5	0.8	0.4	0.2	0.4
10.7	.1	.1	58.0	2.5	16.7	0	2.2	.8	.4	.1	.2
7.1	.4	<.1	55.4	5.2	13.3	0	2.3	6.7	.2	.1	.3
11.5	.2	<.1	54.1	4.6	16.4	<.1	2.5	2.9	.3	.4	.1
13.5	0	0	24.9	23.0	34.4	.2	1.0	1.1	<.1	<.1	.1
11.8	.1	<.1	31.1	22.0	26.8	.3	1.4	1.8	.1	.1	.1
5.7	0	0	65.3	14.6	.7	0	2.1	3.2	.4	.6	1.1
25.9	0	.5	67.0	.1	4.2	0	0	0	.2	0	1.1
6.2	.2	0	50.9	2.2	1.4	0	1.7	2.9	1.1	1.0	<.1
8.9	.1	.2	49.6	14.2	9.4	0	2.7	2.6	.5	.7	.4
12.5	.6	.1	73.6	1.3	0	0	1.3	1.1	.2	.1	.4
16.3	1.2	0	43.0	0	.4	0	1.8	0	.8	.1	<.1
20.4	<.1	<.1	52.3	14.7	1.2	0	1.4	2.3	.3	<.1	.1
19.8	.1	<.1	53.9	10.0	1.7	<.1	2.3	1.7	.3	.1	.5
10.5	0	0	61.6	.1	2.6	0	2.1	.3	.4	.1	.2
5.6	0	0	69.0	0	0	0	1.6	.8	0	0	.9
4.8	0	0	47.4	0	0	0	1.0	0	0	0	.1

Table 28. Summary of median nitrate concentrations for selected Acid Rain Monitoring Project (ARMP) sites, 1984-93

[Overall median for the ARMP sites is <0.2 milligram per liter (mg/L). Stations are listed in order of decreasing median concentration. Data are from University of Massachusetts Water Resources Research Center. <, actual value is less than value shown]

Site name	Period of record	Latitude ° ′ ″	Longitude ° ′ ″	Nitrate as N (mg/L)		Number of samples
				Median	Maximum	
Bottom Brook, Northfield, Mass.	1984-93	42 42 28	72 27 33	5.15	8.83	26
Wellington Brook, Oxford, Mass.	1986-92	42 08 44	71 52 32	4.66	9.63	19
Mountain Brook, Southwick, Mass.	1984-91	42 00 28	72 46 14	3.22	5.13	16
Plum Brook, Southampton, Mass.	1985-93	42 16 12	72 39 15	1.26	1.90	25
Bagg Brook, West Springfield, Mass.	1986-92	42 07 26	72 37 24	1.25	1.97	15
Mill River, Springfield, Mass.	1986-93	42 05 18	72 35 02	1.06	1.94	22
Turkey Brook, Northampton, Mass.	1984-93	42 17 48	72 44 03	.82	2.62	23
Dry Brook, Gill, Mass.	1985-93	42 38 03	72 29 29	.81	1.77	27
South River, Ashfield, Mass.	1986-93	42 32 27	72 39 29	.63	1.41	23
Sleepy Hollow Brook, Richmond, Mass.	1985-92	42 23 17	73 21 34	.62	2.48	20
Stafford Brook, Colrain, Mass.	1986-93	42 40 48	72 39 12	.62	1.72	23
Hinsdale Brook, Shelburne, Mass.	1986-93	42 37 18	72 37 32	.61	1.12	25
Cady Brook, Charlton, Mass.	1986-92	42 04 36	72 01 32	.60	1.81	23
Swift River, Ashfield, Mass.	1985-90	42 26 39	72 51 33	.60	1.16	16
Sucker Brook, New Braintree, Mass.	1986-92	42 15 24	72 09 24	.45	1.35	23
Plum Brook, Amherst, Mass.	1984-93	42 20 31	72 31 37	.44	1.22	22
Esther Brook, Conway, Mass.	1986-93	42 27 16	72 38 16	.44	1.10	23
Moose Meadow Brook, Montgomery, Mass.	1986-93	42 08 13	72 47 03	.43	1.28	27
Sackett Brook, Dalton, Mass.	1986-90	42 25 23	73 14 07	.42	.73	11
Chicopee River, Chicopee, Mass.	1985-92	42 08 54	72 37 19	.39	3.64	18
Little River, Westfield, Mass.	1986-93	42 07 00	72 44 02	.39	.88	27
Housatonic River, Sheffield, Mass.	1986-90	42 02 54	73 20 27	.38	.81	14
Stage Brook, Blandford, Mass.	1986-93	42 11 17	72 51 52	.38	1.24	24
French River, Oxford, Mass.	1984-92	42 07 18	71 52 48	.38	1.01	21
West Branch/Brook, Worthington, Mass.	1985-93	42 24 00	72 52 28	.37	1.61	26
Mill River Diversion, Northampton, Mass.	1984-93	42 17 29	72 38 26	.37	.85	20
Mill River, Northampton, Mass.	1986-93	42 17 03	72 38 57	.36	1.66	22
Shingle Brook, Shelburne, Mass.	1986-93	42 32 39	72 39 39	.35	.98	22
Naultaug Brook, Warren, Mass.	1986-93	42 13 33	72 10 27	.33	1.29	21
Munn Brook, Southwick, Mass.	1985-93	42 07 11	72 47 54	.30	.77	22
Big Brook, Wilbraham, Mass.	1986-93	42 03 51	72 24 36	.29	1.27	22
Fort River, Hadley, Mass.	1986-93	42 19 33	72 35 05	.26	1.19	23
Deerfield River, Deerfield, Mass.	1986-90	42 34 40	72 34 41	.26	1.04	12
Mill River, Conway, Mass.	1986-93	42 21 06	72 36 56	.25	1.13	16
Gibbs Brook, Blandford, Mass.	1984-93	42 12 00	72 54 56	.25	.56	21
Scantic River, Hampden, Conn.	1985-93	41 52 06	72 37 00	.25	1.19	28
Underwood Brook, Heath, Mass.	1986-93	42 42 20	72 49 26	.23	.68	22
Mount Lebanon Brook, Hancock, Mass.	1986-92	42 24 59	73 19 50	.23	.57	21
Beaman Brook, Winchendon, Mass.	1984-92	42 37 35	72 05 35	<.2	2.90	19
Hartwell Brook, Charlemont, Mass.	1985-92	42 37 01	72 50 12	<.2	2.13	24
West Brook, Great Barrington, Mass.	1986-91	42 16 08	73 16 23	<.2	1.46	12
Bailey Brook, Bernardston, Mass.	1985-93	42 40 41	72 28 45	<.2	1.45	23
Jabish Brook, Palmer, Mass.	1986-93	42 12 28	72 21 48	<.2	1.29	25
Dry Brook, Sheffield, Mass.	1985-93	42 03 42	73 24 32	<0.2	1.20	20

Table 28. Summary of median nitrate concentrations for selected Acid Rain Monitoring Project (ARMP) sites, 1984-93 —*Continued*

Site name	Period of record	Latitude ° ′ ″	Longitude ° ′ ″	Nitrate as N (mg/L)		Number of samples
				Median	Maximum	
May Brook, Holland, Mass.	1984-93	42 01 57	72 09 57	<.2	1.17	24
West Branch Ware River, Hubbardston, Mass.	1984-93	42 25 45	72 01 15	<.2	1.16	26
Flat Brook, Ware, Mass.	1986-93	42 14 27	72 16 00	<.2	1.12	26
Doolittle Brook, Leverett, Mass.	1986-93	42 25 46	72 29 34	<.2	1.10	25
Factory Brook, Middlefield , Mass.	1985-92	42 18 33	73 01 30	<.2	1.08	22
Bozrah Brook, Hawley, Mass.	1986-93	42 37 30	72 52 54	<.2	1.07	23
Scarboro Brook	1986-93	42 19 18	72 26 54	<.2	1.06	22
Roberts Meadow Brook, Northampton, Mass.	1986-93	42 21 12	72 42 06	<.2	1.06	21
Sykes Brook, Pittsfield, Mass.	1986-91	42 25 15	73 14 11	<.2	.90	15
Drake Brook, Southwick, Mass.	1984-93	42 04 53	72 50 12	<.2	.78	18
Tully River, Athol, Mass.	1986-92	42 35 48	72 14 25	<.2	.73	21
Hamant Brook, Sturbridge, Mass.	1986-93	42 05 54	72 05 27	<.2	.71	21
Breakneck Brook, Sturbridge, Mass.	1985-93	42 04 38	72 04 54	<.2	.69	24
Burrow Brook, Oakham, Mass.	1986-92	42 22 27	72 05 54	<.2	.68	25
Hop Brook, Lee, Mass.	1986-93	42 16 19	73 15 09	<.2	.67	21
Florida Brook, Huntington, Mass.	1984-92	42 18 51	72 51 07	<.2	.67	17
Barton Brook, Dalton , Mass.	1986-93	42 27 43	73 12 05	<.2	.67	21
Moody Brook, Tolland, Mass.	1986-92	42 04 11	73 03 41	<.2	.66	15
Bow Brook, New Salem, Mass.	1984-93	42 31 10	72 16 18	<.2	.63	24
Williams River, West Stockbridge, Mass.	1986-92	42 13 37	73 21 10	<.2	.62	21
Gulf Brook, Pelham, Mass.	1986-93	42 21 57	72 22 05	<.2	.62	20
Pond Brook, Granville, Mass.	1986-93	42 02 18	72 56 31	<.2	.61	25
Stones Brook, Ashfield, Mass.	1985-90	42 27 18	72 50 51	<.2	.60	17
Umpachene River, Monterey, Mass.	1985-93	42 05 39	73 16 19	<.2	.59	18
Baker Brook, Gardner, Mass.	1986-93	42 33 10	71 59 21	<.2	.59	25
Fourmile Brook, Northfield, Mass.	1985-93	42 36 56	72 28 45	<.2	.58	26
Gate Hill Brook, Wendell, Mass.	1984-93	42 34 08	72 25 45	<.2	.58	23
Charles Brook, Brimfield, Mass.	1984-93	42 06 57	72 13 04	<.2	.57	21
Benton Brook, Otis, Mass.	1984-93	42 11 06	73 05 03	<.2	.56	22
Vincent Brook, Colrain, Mass.	1986-93	42 41 57	72 46 27	<.2	.56	24
Swift River, Ware, Mass.	1985-92	42 11 21	72 21 29	<.2	.56	27
Robbins Brook, Winchendon, Mass.	1986-93	42 41 45	72 04 55	<.2	.55	26
Mahoney Brook, Gardner, Mass.	1984-93	42 33 38	71 59 03	<.2	.53	24
Kilburn Brook, Peru, Mass.	1986-93	42 25 17	73 04 55	<.2	.53	23
West Branch Westfield, Chester, Mass.	1985-93	42 13 50	72 52 24	<.2	.53	29
Wilder Brook, Gardner, Mass.	1986-93	42 35 03	72 00 46	<.2	.52	24
Cady Brook, Washington, Mass.	1986-93	42 24 59	73 06 36	<.2	.52	17
Hume Brook, Windsor, Mass.	1986-93	42 29 22	73 01 37	<.2	.51	18
Hoyt Brook, Phillipston, Mass.	1984-93	42 36 42	72 07 42	<.2	.51	25
Yokum Brook, Becket, Mass.	1986-91	42 19 58	73 04 58	<.2	.50	13
Hearthstone Brook, Shutesbury, Mass.	1984-93	42 23 08	72 29 21	<.2	.50	25
Mill Brook, Washington, Mass.	1986-93	42 23 16	73 14 26	<.2	.49	22
Anthony Brook, Dalton , Mass.	1986-93	42 28 45	73 09 08	<.2	.49	22
Parsons Brook, Northampton, Mass.	1984-93	42 18 33	72 41 17	<.2	.48	27

Table 28. Summary of median nitrate concentrations for selected Acid Rain Monitoring Project (ARMP) sites, 1984-93 —*Continued*

Site name	Period of record	Latitude ° ′ ″	Longitude ° ′ ″	Nitrate as N (mg/L)		Number of samples
				Median	Maximum	
Sawmill River, Upper, Montague, Mass.	1984-92	42 31 35	72 34 00	<0.2	0.48	25
Gulf Brook, Savoy, Mass.	1984-92	42 37 59	72 59 50	<.2	.47	19
Yokun Brook, Lenox, Mass.	1987-91	42 22 42	73 14 43	<.2	.47	13
Lawrence Brook, Royalston, Mass.	1985-93	42 38 57	72 12 24	<.2	.47	26
Walker Brook, Becket, Mass.	1985-92	42 16 40	72 58 48	<.2	.46	24
East Oxbow Brook, Charlemont, Mass.	1985-93	42 37 17	72 46 50	<.2	.45	24
Priest Brook, Winchendon, Mass.	1986-93	42 38 39	72 05 58	<.2	.44	22
Bronson Brook, Worthington, Mass.	1985-93	42 25 27	72 55 10	<.2	.44	24
West Brook, Upper, Orange, Mass.	1986-93	42 34 33	72 15 42	<.2	.44	10
Middle Brook, Westfield River, Middlefield, Mass.	1986-93	42 15 33	72 52 00	<.2	.44	24
Maynard Brook, Oakham, Mass.	1987-92	42 19 18	72 02 36	<.2	.43	15
Dunbar Brook, Monroe, Mass.	1985-89	42 42 25	72 57 05	<.2	.42	14
Pond Brook, Huntington, Mass.	1984-93	42 16 49	72 52 06	<.2	.42	26
Soda Creek, Sheffield, Mass.	1984-93	42 06 21	73 19 36	<.2	.42	20
Dean Brook, Shutesbury, Mass.	1985-93	42 24 54	72 28 44	<.2	.41	26
Moccasin Brook, Petersham, Mass.	1986-93	42 28 14	72 09 43	<.2	.41	17
Clear Brook, Windsor, Mass.	1984-93	42 31 55	72 59 38	<.2	.40	20
Tyler Brook, Wendell, Mass.	1984-93	42 30 41	72 25 48	<.2	.39	18
Amethyst Brook, Amherst, Mass.	1985-93	42 22 41	72 29 36	<.2	.39	29
Windsor Jambs Brook, Windsor, Mass.	1986-93	42 30 47	72 59 21	<.2	.38	24
Moosehorn Brook, New Salem, Mass.	1986-93	42 30 33	72 18 50	<.2	.36	24
Bearmeadow Brook, Ashburnham	1986-93	42 41 16	71 57 16	<.2	.35	22
West Brook, Lower, Orange, Mass.	1985-93	42 34 33	72 15 42	<.2	.35	14
Black Brook, Warwick, Mass.	1984-93	42 40 55	72 19 13	<.2	.35	16
East Branch Tully River, Royalston, Mass.	1985-93	42 36 48	72 14 34	<.2	.34	26
White Brook, Florida, Mass.	1984-92	42 40 22	73 01 59	<.2	.33	20
Kenny Brook, Royalston, Mass.	1985-93	42 37 59	72 08 22	<.2	.33	21
Newton Brook, Hardwick, Mass.	1984-92	42 18 56	72 14 23	<.2	.33	23
Ground Brook, Conway, Mass.	1985-93	42 26 03	72 40 18	<.2	.32	25
Chickley River, Hawley, Mass.	1985-90	42 37 48	72 54 06	<.2	.32	16
Rice Brook, Rowe, Mass.	1986-93	42 41 13	72 54 34	<.2	.32	19
Shattuck Brook, Phillipston, Mass.	1984-93	42 31 49	72 09 01	<.2	.32	17
Lord Brook, Rowe, Mass.	1986-93	42 43 26	72 56 18	<.2	.31	23
Chaffee Brook, Pelham, Mass.	1986-92	42 20 02	72 22 08	<.2	.31	16
Wilder Brook, Charlemont, Mass.	1985-93	42 37 13	72 47 24	<.2	.31	23
Mill Brook, Northfield, Mass.	1985-93	42 42 20	72 27 29	<.2	.31	23
Templeton Brook, Templeton, Mass.	1986-93	42 32 32	72 00 48	<.2	.31	24
Shaker Mill Brook, Becket, Mass.	1986-93	42 19 59	73 05 03	<.2	.31	23
Whiting River, New Canaan, Conn.	1986-92	42 00 38	73 16 26	<.2	.30	15
Middle Brook Swift River, Petersham, Mass.	1985-93	42 31 14	72 18 05	<.2	.30	27
Todd Brook, Charlemont, Mass.	1985-93	42 39 02	72 55 53	<.2	.30	22
Glendale Brook, Middlefield, Mass.	1984-93	42 21 06	72 57 42	<.2	.30	26
Fish Brook, Royalston, Mass.	1986-93	42 39 54	72 16 13	<.2	.30	18
Cobb Brook, Shutesbury, Mass.	1985-93	42 25 45	72 22 51	<.2	.29	27
Mongue Meadow Brook, Windsor, Mass.	1986-92	42 28 05	72 59 26	<.2	.29	18

Table 28. Summary of median nitrate concentrations for selected Acid Rain Monitoring Project (ARMP) sites, 1984-93 *—Continued*

Site name	Period of record	Latitude ° ′ ″	Longitude ° ′ ″	Nitrate as N (mg/L)		Number of samples
				Median	Maximum	
Cadwell Creek, Pelham, Mass.	1985-93	42 20 03	72 22 12	<0.2	0.29	25
Breakneck Brook,Huntington, Mass.	1984-93	42 14 34	72 47 57	<.2	.29	20
Dead Branch/Brook, Chesterfield, Mass.	1984-93	42 20 24	72 50 47	<.2	.29	27
Osgood Brook, Wendell, Mass.	1986-92	42 35 45	72 21 56	<.2	.29	21
Hamilton Brook, Becket, Mass.	1986-93	42 16 24	73 03 42	<.2	.28	18
Bilodeau Brook, Hinsdale, Mass.	1986-93	42 23 43	73 06 20	<.2	.27	17
Roaring Brook, Leverett, Mass.	1985-92	42 26 09	72 29 34	<.2	.27	25
Mormon Hollow Brook, Wendell, Mass.	1985-93	42 35 26	72 26 32	<.2	.27	26
Boyce Brook, Royalston, Mass.	1986-92	42 40 46	72 13 09	<.2	.27	23
Jack's Brook, Upper, Northfield, Mass.	1985-93	42 36 51	72 24 03	<.2	.26	16
Atherton Brook, Shutesbury, Mass.	1985-92	42 24 51	72 22 49	<.2	.26	24
Bennett Brook, Northfield, Mass.	1984-93	42 40 13	72 28 24	<.2	.26	24
Purgee Brook, Pelham, Mass.	1985-93	42 23 12	72 22 26	<.2	.26	25
Keyup Brook, Lower, Northfield, Mass.	1986-93	42 35 53	72 24 06	<.2	.26	13
Pelham Brook, Rowe, Mass.	1985-89	42 39 17	72 56 15	<.2	.25	14
Ellinwood Brook, Athol, Mass.	1985-93	42 32 48	72 16 04	<.2	.23	27
Giles Brook, New Salem, Mass.	1984-93	42 30 11	72 18 54	<.2	.23	24
Unquomonk Brook, Williamsburg, Mass.	1986-93	42 23 00	72 43 01	<.2	.22	23
Spud Brook, Winchendon, Mass.	1987-93	42 42 50	72 05 11	<.2	.22	20
Kidder Brook, Warwick, Mass.	1984-93	42 42 33	72 21 03	<.2	.21	23
Tully Brook, Warwick, Mass.	1985-92	42 40 54	72 17 08	<.2	.21	23
Towne Brook, Royalston, Mass.	1984-93	42 42 10	72 07 18	<.2	.20	22
Moss Brook, Upper, Warwick, Mass.	1985-93	42 35 56	72 21 33	<.2	<.2	5
Bluefield Brook, Ashburnham, Mass.	1985-93	42 41 16	71 57 16	<.2	<.2	21
Bailey Brook, Gardner, Mass.	1985-93	42 35 35	72 01 38	<.2	<.2	22
Blodgett Mill Brook, Warren, Mass.	1985-93	42 10 11	72 15 43	<.2	<.2	18
West Brook Tully River, Orange, Mass.	1984-93	42 36 49	72 14 34	<.2	<.2	26
Fox Brook, Granville, Conn.	1984-93	42 01 30	72 51 38	<.2	<.2	22
Hubbard Brook, Granville, Conn.	1986-93	42 01 50	72 56 04	<.2	<.2	26
Valley Brook, Granville, Conn.	1985-93	42 01 50	72 56 02	<.2	<.2	26
Thousand Acre Brook, Phillipston, Mass.	1984-92	42 37 01	72 09 32	<.2	<.2	19
Hop Brook, New Salem, Mass.	1985-93	42 28 00	72 19 09	<.2	<.2	25
Scott Brook, Royalston, Mass.	1985-93	42 42 10	72 07 18	<.2	<.2	26
Nelson Brook, Athol, Mass.	1984-93	42 31 54	72 12 50	<.2	<.2	22
Ripley Brook, Granville, Mass.	1986-93	42 07 26	72 57 10	<.2	<.2	15
Briggs Brook, Pelham, Mass.	1986-93	42 24 35	72 22 53	<.2	<.2	25
East Branch Fever Brook, Petersham, Mass.	1986-93	42 26 44	72 15 15	<.2	<.2	27
Lyons Brook, Wendell, Mass.	1986-93	42 34 36	72 28 00	<.2	<.2	23
Bigelow Brook, Phillipston, Mass.	1985-93	42 31 33	72 09 10	<.2	<.2	25
Popple Camp Brook, Phillipston, Mass.	1984-92	42 31 49	72 09 01	<.2	<.2	24
Baker Brook, Wendell, Mass.	1984-92	42 34 09	72 25 46	<.2	<.2	20
Adams Brook, Shutesbury, Mass.	1985-93	42 22 41	72 29 36	<.2	<.2	28
Keyup Brook, Northfield, Mass.	1986-93	42 35 53	72 24 06	<.2	<.2	12
West Branch Fever Brook, Petersham, Mass.	1986-92	42 28 02	72 15 34	<.2	<.2	22
West Branch Swift River, Shutesbury, Mass.	1986-93	42 16 59	72 20 38	<.2	<.2	24

Table 28. Summary of median nitrate concentrations for selected Acid Rain Monitoring Project (ARMP) sites, 1984-93 —*Continued*

Site name	Period of record	Latitude ° ′ ″	Longitude ° ′ ″	Nitrate as N (mg/L)		Number of samples
				Median	Maximum	
Riceville Brook, Athol, Mass.	1985-93	42 32 12	72 15 24	<0.2	<0.2	28
Whetstone Brook, Wendell, Mass.	1985-93	42 35 42	72 21 47	<.2	<.2	26
Rocky Brook, Douglas, Mass.	1985-92	42 00 12	71 48 34	<.2	<.2	20
Moss Brook, Warwick, Mass.	1990-93	42 35 56	72 21 33	<.2	<.2	11
Orcutt Brook, Upper, Warwick, Mass.	1990-93	42 35 53	72 20 29	<.2	<.2	18
Orcutt Brook, Lower, Warwick, Mass.	1985-93	42 35 53	72 20 29	<.2	<.2	10
Keyup Brook, Upper, Northfield, Mass.	1986-93	42 35 53	72 24 06	<.2	<.2	15

Table 29. Summary of median concentrations for selected phosphorus data from the U.S. Environmental Protection Agency's storage and retrieval system for selected water-quality stations in the Connecticut, Housatonic, and Thames Rivers study unit, water years 1972-90

[Map No.: See plate 1 for locations of water-quality stations and table 23 for summary of data. Median concentrations are not necessarily representative of the annual conditions and they are not necessarily representative of the seasonal variations because all months and years were not equally represented. Stations are listed in descending order of map no. mg/L, milligram per liter; --, no data; <, actual value is less than value shown]

Map No.	Station name	Period of record (water year)	Phosphorus, total as P (mg/L)		Orthophosphate phosphorus, dissolved as P (mg/L)		Phosphorus, dissolved, as P (mg/L)	
			Median	Samples	Median	Samples	Median	Samples
228	Wooster Brook at Waterbury, Conn.	1974-88	0.02	62	<0.10	7	--	--
227	Beaver Brook at Bethlehem, N.H.	1979-90	--	--	--	--	--	--
226	Cold Brook at Berlin, N.H.	1973-90	--	--	--	--	--	--
225	Number 9 Brook at Berlin, N.H.	1979-90	--	--	--	--	--	--
224	Small Brook at Berlin, N.H.	1979-83	--	--	--	--	--	--
223	West Branch Upper Ammonoosuc River at Berlin, N.H.	1979-90	--	--	--	--	--	--
222	Wild Ammonoosuc River at Benton, N.H.	1978-90	--	--	--	--	--	--
221	Passumpsic River at East Barnet, Vt.	1985	--	--	--	--	--	--
220	Clough Brook at Lemington, Vt.	1984-85	.005	12	--	--	--	--
219	Passumpsic River at St. Johnsbury, Vt.	1985	--	--	--	--	--	--
218	Moose River at Granby, Vt.	1984-85	.004	12	--	--	--	--
217	Ompompanoosuc River at Union Village, Vt.	1972-88	<.03	65	<.10	7	--	--
216	West Branch Ompompanoosuc River at Thetford, Vt.	1972-88	<.01	64	<.10	7	--	--
215	East Branch Ompompanoosuc River at Thetford Center, Vt.	1972-88	<.02	65	<.10	7	--	--
214	West Branch Ompompanoosuc River at South Strafford, Vt.	1979-88	<.02	64	<.10	7	--	--
213	East Branch Ompompanoosuc River at West Fairlee, Vt.	1979-88	<.01	24	<.10	8	--	--
212	Harvey's Tributary #2 at Barnet, Vt.	1981-82	.26	196	--	--	0.26	40
211	Steven's River at Barnet, Vt.	1980	.014	16	--	--	.005	6
210	South Peacham Brook at Barnet, Vt.	1980-82	.01	313	--	--	.005	63
209	South Peacham Brook at South Peacham, Vt.	1982	--	--	--	--	.005	9
208	South Peacham Brook at Peacham, Vt.	1982	--	--	--	--	.008	10
207	Ottauquechee River at North Hartland, Vt.	1972-82	.01	24	--	--	--	--
206	Connecticut River at Hinsdale, N.H.	1980-86	.025	14	--	--	--	--

Table 29. Summary of median concentrations for selected phosphorus species from the U.S. Environmental Protection Agency's computer STorage and RETrieval (STORET) system for selected water-quality stations in the Connecticut, Housatonic, and Thames Rivers study unit, water years 1972-90
—Continued

Map No.	Station name	Period of record (water year)	Phosphorus, total as P (mg/L)		Orthophosphate phosphorus, dissolved as P (mg/L)		Phosphorus, dissolved, as P (mg/L)	
			Median	Samples	Median	Samples	Median	Samples
205	Connecticut River at Lancaster, N.H.	1978-89	0.03	32	0.005	29	--	--
204	Connecticut River at Cornish, N.H.	1976-81	.01	40	.004	35	--	--
203	Connecticut River at Hinsdale, N.H.	1980-86	.021	14	--	--	--	--
202	North Branch Gale River at Bethlehem, N.H.	1976-90	--	--	--	--	--	--
201	Garland Brook at Lancaster, N.H.	1980-90	--	--	--	--	--	--
200	Morey Tributary #3 at Fairlee, Vt.	1981-82	.039	93	--	--	0.027	91
199	Scarface Brook at Bethlehem, N.H.	1982-90	--	--	--	--	--	--
198	Clark Brook at Granville, Vt.	1984-85	<.003	10	--	--	--	--
197	Hancock Branch tributary to White River at Hancock, Vt.	1984-85	<.003	12	--	--	--	--
196	White River at Granville, Vt.	1984-85	.003	11	--	--	--	--
195	Third Branch White River at Roxbury, Vt.	1982	--	--	--	--	.02	10
194	Third Branch White River at Roxbury, Vt.	1982	--	--	--	--	.007	9
193	North Branch Black River at Weathersfield, Vt.	1980-88	<.03	28	--	--	--	--
192	Ottauquechee River at North Hartland, Vt.	1981-88	<.01	12	<.10	3	--	--
191	Ottauquechee River at Quechee Village, Vt.	1981-82	<.03	14	--	--	--	--
190	Ottauquechee River at Quechee Gorge, Vt.	1980-88	<.02	37	<.10	3	--	--
189	Black River at Perkinsville, Vt.	1972-86	<.03	31	--	--	--	--
188	North Branch Black River at Amsden, Vt.	1972-86	<.0 3	31	--	--	--	--
187	Ottauquechee River at Quechee, Vt.	1974-88	<.03	34	<.10	3	--	--
186	West River at North Springfield, Vt.	1972-86	.01	29	--	--	--	--
185	North Branch Ball Mountain Brook at Jamaica, Vt.	1988	.008	7	--	--	.004	7
184	West River at Stratton, Vt.	1988	.008	7	--	--	.006	7
183	Ball Mountain Brook at West Jamaica, Vt.	1984-85	.004	12	--	--	--	--
182	West River at Townshend, Vt.	1981-86	<.03	9	--	--	--	--
181	West River at East Jamaica, Vt.	1972-86	<.02	14	--	--	--	--
180	Wardsboro Brook at Jamaica, Vt.	1976-86	<.02	15	--	--	--	--
179	North Branch Ball Mountain Brook at Jamaica, Vt.	1985-88	.01	9	--	--	.004	21
178	Winhall River at Londonderry, Vt.	1977-86	<.03	12	--	--	--	--
177	West River at South Londonderry, Vt.	1980-86	<.03	12	--	--	--	--
176	Winhall River near Rawsonville, Vt.	1980-86	<.02	13	--	--	--	--
175	Winhall River at Winhall, Vt.	1982-85	.007	11	--	--	--	--
174	Kidder Brook at Jamaica, Vt.	1985-87	.003	2	--	--	.002	14
173	Ashuelot River at Gilsum, N.H.	1980-86	<.03	22	--	--	--	--
172	Ashuelot River below Surry Mountain Dam at Surry, N.H.	1972-86	.01	22	--	--	--	--
171	Otter Brook below Otter Brook Dam, at Keene, N.H.	1972-86	<.03	17	--	--	--	--
170	Otter Brook at Keene, N.H.	1972-86	<.03	18	--	--	--	--
169	Merriam Brook at Surry, N.H.	1973-86	<.03	18	--	--	--	--
168	Merriam Brook at Surry, N.H.	1975-86	<.03	18	--	--	--	--

Table 29. Summary of median concentrations for selected phosphorus species from the U.S. Environmental Protection Agency's computer STorage and RETrieval (STORET) system for selected water-quality stations in the Connecticut, Housatonic, and Thames Rivers study unit, water years 1972-90
—Continued

Map No.	Station name	Period of record (water year)	Phosphorus, total as P (mg/L)		Orthophosphate phosphorus, dissolved as P (mg/L)		Phosphorus, dissolved, as P (mg/L)	
			Median	Samples	Median	Samples	Median	Samples
167	Ashuelot River at Swanzey, N.H.	1976-89	0.10	43	0.042	34	--	--
166	East Branch Tully River at Tully Dam in Royalston, Mass.	1980-86	.02	14	--	--	--	--
165	Miller River at Winchendon, Mass.	1980-85	.06	26	--	--	--	--
164	Priest Brook at Royalston, Mass.	1980-85	.02	26	--	--	--	--
163	Lawrence Brook below Doanes Falls, Royalston, Mass.	1976-86	.03	14	--	--	--	--
162	Miller River below Birch Hill Dam, Royalston, Mass.	1972-85	.16	26	--	--	--	--
161	Otter River at Baldwinsville, Mass.	1976-85	.365	26	--	--	--	--
160	East Branch Tully River at Royalston, Mass.	1980-86	.02	15	--	--	--	--
159	North Branch Deerfield River at Wilmington, Vt.	1985	.073	13	--	--	--	--
158	Binney Brook at Wilmington, Vt.	1985	.007	13	--	--	--	--
157	North Branch Deerfield River at Wilmington, Vt.	1985	.012	14	--	--	--	--
156	North Branch Deerfield River at Wilmington, Vt.	1985	.076	13	--	--	--	--
155	Deerfield River at Somerset, Vt.	1984-85	.01	10	--	--	--	--
154	Conant Brook at Monson, Mass.	1973-87	<.03	16	--	--	--	--
153	Vinica Brook at South Monson, Mass.	1972-87	<.03	14	--	--	--	--
152	Westfield River below Knightville Dam at Huntington, Mass.	1973-86	<.02	11	--	--	--	--
151	Westfield River at Huntington, Mass.	1980-88	<.02	11	<.10	1	--	--
150	Little River at Huntington, Mass.	1973-86	<.02	10	--	--	--	--
149	West Branch Farmington River below Goodwin Dam, at Hartland, Conn.	1981-86	<.03	8	--	--	--	--
148	Quinebaug River at Thompson, Conn.	1975-88	.192	50	<.30	7	--	--
147	Quinebaug River at Fabyan, Conn.	1972-88	.18	31	<.30	7	--	--
146	Quinebaug River at Fabyan, Conn.	1978-88	.22	52	.12	7	--	--
145	Quinebaug River below West Thompson Dam, at Thompson, Conn.	1972-88	.144	52	<.30	7	--	--
144	Quinebaug River at Thompson, Conn.	1981-88	.145	38	<.30	7	--	--
143	Quinebaug River at Thompson, Conn.	1972-88	.15	28	.10	5	--	--
142	Mill Brook at Brimfield, Mass.	1980-88	.02	40	<.10	3	--	--
141	Quinebaug River at Holland, Mass.	1980-88	.01	43	<.10	3	--	--
140	Quinebaug River at East Brimfield, Mass.	1972-88	.02	42	<.10	3	--	--
139	French River at Oxford, Mass.	1980-87	.067	14	--	--	--	--
138	Quinebaug River below East Brimfield Dam, at Sturbridge Mass.	1972-88	.01	41	<.10	3	--	--
137	Tributary to Quinebaug River at Quinebaug, Conn.	1982-88	.18	37	.11	7	--	--
136	French River below Hodges Village Dam, at Oxford, Mass.	1972-87	.07	13	--	--	--	--
135	Little River below Buffumville Dam, at Oxford, Mass.	1972-88	.02	39	<.10	3	--	--
134	Little River at Oxford, Mass.	1972-88	.02	40	<.10	3	--	--
133	Quinebaug River below Westville Dam, at Southbridge, Mass.	1972-88	.04	27	<.30	2	--	--
132	Quinebaug River at Sturbridge, Mass.	1972-88	.05	27	<.30	2	--	--

Table 29. Summary of median concentrations for selected phosphorus species from the U.S. Environmental Protection Agency's computer STorage and RETrieval (STORET) system for selected water-quality stations in the Connecticut, Housatonic, and Thames Rivers study unit, water years 1972-90
—Continued

Map No.	Station name	Period of record (water year)	Phosphorus, total as P (mg/L)		Orthophosphate phosphorus, dissolved as P (mg/L)		Phosphorus, dissolved, as P (mg/L)	
			Median	Samples	Median	Samples	Median	Samples
131	South Fork Little River at Oxford, Mass.	1972-88	0.01	39	<0.10	3	--	--
130	Natchaug River at Mansfield Hollow, Conn.	1972-86	.02	10	--	--	--	--
129	Mount Hope River at Mansfield, Conn.	1980-86	.03	10	--	--	--	--
128	Fenton River at Mansfield, Conn.	1980-86	.02	9	--	--	--	--
127	Naugatuck River at Thomaston, Conn.	1972-85	.27	29	--	--	--	--
126	Northfield Brook at Thomaston, Conn.	1979-88	.01	19	<.10	3	--	--
125	Leadmine Brook at Thomaston, Conn.	1972-87	.01	29	--	--	--	--
124	Naugatuck River at Campville, Conn.	1972-85	.37	28	--	--	--	--
123	Northfield Brook at Thomaston, Conn.	1972-88	.01	20	<.10	3	--	--
122	Confluence of Hop and Shattuck Brooks at Waterbury, Conn.	1972-88	.024	60	<.10	7	--	--
121	Hop Brook Dam at Waterbury, Conn.	1972-87	.04	15	--	--	--	--
120	Welton Brook at Waterbury, Conn.	1974-88	.02	59	<.10	7	--	--

Table 30. Summary of pesticide detections and concentrations in ground water in the Connecticut, Housatonic, and Thames Rivers study unit, 1978-89

[Data from the USGS National Water Information System (NWIS). All data are included in NWIS except pesticide compounds that did not have assigned parameter codes. Analytical data for pesticide compounds without parameter codes (no code) are on file and available at the Connecticut District Office, Hartford, Connecticut. Ground-water data include wells and springs; reporting level is the least concentration detectable by the analytical methods used during the period of record. **Health advisory 70-kilogram adult DWEL:** assumes 2 liters per day water consumption with 100 percent daily contaminant intake from drinking water. µg/L, micrograms per liter; USEPA, U.S. Environmental Protection Agency; MCL, Maximum Contaminant Level; DWEL, Drinking Water Equivalent Level; DDD, dichlorodiphenyldichloroethane; DDE, dichlorodiphenyldichloroethylene; DDT, dichlorodiphenyltrichloroethane; 2,4-D, (2,4-dichlorophenoxy) acetic acid; 2,4-DP, 2-(2,4-dichlorophenoxy) propionic acid; 2,4,5-T, (2,4,5-trichlorophenoxy) acetic acid]

Pesticide name (parameter code)	Reporting level (µg/L)	Number of sites or samples with detections (percentage shown in parentheses)		Concentrations (µg/L)	USEPA standards or guidelines	
		Sites	Samples		Water-quality criteria drinking water MCL (µg/L) and USEPA references	Health advisory 70-kilogram adult, DWEL (µg/L) and USEPA references
Organochlorine insecticides						
Chlordane, total (39350)	0.1	4 (3)	4 (3)	0.1-0.3	[1]2 (Code of Federal Regulations, 1991a,1992a, 1992b)	2 (1992b)
DDD, total (39360)	.001	1 (<1)	1 (<1)	0.01	--	--
DDE, total (39365)	.001	2 (2)	3 (2)	0.001	--	--
DDT, total (39370)	.001	1 (<1)	1 (<1)	0.002	--	--
Dieldrin, total (39380)	.001	7 (5)	13 (8)	0.001-0.13	--	2 (1992a,1992g, 1992c)
Lindane, total (39340)	.001	1 (<1)	5 (3)	0.001-0.005	--	--
Organophosphorus insecticides						
Chlorpyrifos, total (no code)	.01	1 (<1)	1 (<1)	0.01	--	100 (1992f,1992b)
Diazinon, dissolved (39572)	.01	1 (8)	1 (8)	0.01	--	3 (1988b,1989, 1992b)
Diazinon, total (39570)	.01	1 (<1)	1 (<1)	0.07	--	3 (1988b,1989, 1992b)
Phorate, total (39023)	.01	1 (1)	1 (1)	0.01	--	--
Carbamate insecticides						
Oxyamyl, total (82613)	.5	4 (3)	4 (2)	0.5-2.5	200 (Code of Federal Regulations,1992g)	900 (1992b)
Benzoic, chlorophenoxy-acid, and pyridine herbicides						
Dicamba, (banvel), total (82052)	.1	1 (2)	1 (1)	0.10	--	1,000 (1988b,1989, 1992b)
2,4-D, dissolved (39732)	.01	1 (8)	1 (8)	0.03	70 (Code of Federal Regulations,1991a, 1992a,1992b)	400 (1992b)

Table 30. Summary of pesticide detections and concentrations in ground water in the Connecticut, Housatonic, and Thames Rivers study unit, 1978-89—*Continued*

Pesticide name (parameter code)	Reporting level (µg/L)	Number of sites or samples with detections (percentage shown in parentheses)				Concentrations (µg/L)	USEPA standards or guidelines	
		Sites		Samples			Water-quality criteria drinking water MCL (µg/L) and USEPA references	Health advisory 70-kilogram adult, DWEL (µg/L) and USEPA references
2,4-D, total (39730)	.01	7	(5)	8	(4)	0.01 -0.06	70 (Code of Federal Regulations,1991a, 1992a,1992b)	400 (1992b)
2,4-DP, total (82183)	.01	1	(<1)	1	(<1)	0.09	--	--
Picloram, total (39720)	.1	1	(2)	1	(1)	0.10	500 (Code of Federal Regulations,1992g)	500 (1988b,1989, 1992a,1992b)
Silvex, dissolved (39762)	.01	1	(8)	1	(8)	0.01	50 (Code of Federal Regulations,1991a, 1992a,1992b)	50 (1992b)
Silvex, total (39760)	.01	1	(<1)	1	(<1)	0.02	50 (Code of Federal Regulations,1991a, 1992a,1992b)	50 (1992b)
2,4,5-T, dissolved (39742)	.01	1	(8)	1	(8)	0.01	--	350 (1992b)
2,4,5-T, total (39740)	.01	2	(1)	2	(1)	0.06 -0.4	--	350 (1992b)
Acetamide and triazine herbicides								
Alachlor, total (77825)	.1	2	(1)	5	(2)	0.1-11.	[1]2 (Code of Federal Regulations,1991a, 1992a,1992b)	350 (1988c, 1992a,1992b)
Atrazine, total (39630)	.1	32	(22)	64	(24)	0.1-9.7	3 (Code of Federal Regulations,1991a, 1992a,1992b)	3 (1988b,1989, 1992b)
Cyanazine, total (81757)	.1	3	(2)	3	(1)	0.1	--	70 (1992b)
Metolachlor, total (82612)	.1	8	(6)	18	(7)	0.1-26.0	--	5,000 (1988b,1989, 1992a,1992b)
Metribuzin, total (82611)	.1	2	(1)	4	(2)	0.1-1.3	--	900 (1988b,1989, 1992a,1992b)
Prometon, total (39056)	.1	3	(2)	3	(1)	0.1-2.0	--	500 (1988b,1989, 1992b)
Prometryn, total (39057)	.1	1	(<1)	2	(<1)	0.9-1.1	--	--
Propazine, total (39024)	.1	2	(1)	3	(1)	0.1	--	700 (1988b,1989, 1992a,1992b)
Simazine, total (39055)	.1	11	(8)	16	(6)	0.1-10.0	4 (Code of Federal Regulations,1992g)	200 (1992b)

Table 30. Summary of pesticide detections and concentrations in ground water in the Connecticut, Housatonic, and Thames Rivers study unit, 1978-89—*Continued*

Pesticide name (parameter code)	Reporting level (µg/L)	Number of sites or samples with detections (percentage shown in parentheses)		Concentrations (µg/L)	USEPA standards or guidelines	
		Sites	Samples		Water-quality criteria drinking water MCL (µg/L) and USEPA references	Health advisory 70-kilogram adult, DWEL (µg/L) and USEPA references
Uracil and other pesticides						
DCPA plus metabolites, total (no code)	.01	11 (73)	15 (79)	0.01-124	--	2,000(1988b,1989, 1992b)
Diuron, total (no code)	.01	3 (43)	3 (43)	0.02 -0.07	--	70 (1988b,1989, 1992a,1992b)
Terbacil, total (no code)	.1	5 (100)	5 (100)	0.1-2.1	--	400 (1988b,1989, 1992b)
Volatile organic compounds						
1,2-Dibromoethylene, total (39082)	.2	2 (2)	3 (2)	0.2 -0.5	[1]0.05(Code of Federal Regulations,1991a, 1992a,1992b)	--
1,1-Dichloroethane, total (34496)	.2	7 (5)	8 (5)	0.2-4.8	--	--
1,1-Dichloroethylene, total (34501)	.2	5 (4)	5 (3)	0 2-14.0	--	--
1,2-Dichloropropane, total (34541)	.2	8 (6)	10 (6)	0.3-41.0	--	--
Semi-volatile compounds						
1,2-Dichlorobenzene, total (34536)	.2	3 (2)	3 (2)	0.5-3.0	--	--
1,4-Dichlorobenzene, total (34571)	.2	1 (<1)	1 (<1)	0.3	--	--