National Water-Quality Assessment Program

Trends in Streamflow and Nutrient and Suspended-Sediment Concentrations and Loads in the Upper Mississippi, Ohio, Red, and Great Lakes River Basins, 1975–2004

Scientific Investigations Report 2008–5213

U.S. Department of the Interior
U.S. Geological Survey

Cover: Contrast in sediment loads in the Mississippi and St. Croix Rivers at their confluence. (Photograph by David Morrison, Minnesota Pollution Control Agency.)

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By David L. Lorenz, Dale M. Robertson, David W. Hall, and David A. Saad

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KEN SALAZAR, Secretary

U.S. Geological Survey
Suzette M. Kimball, Acting Director

U.S. Geological Survey, Reston, Virginia: 2009

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Suggested citation:
Lorenz, D.L., Robertson, D.M., Hall, D.W., and Saad, D.A., 2009, Trends in streamflow and nutrient and suspended-sediment concentrations and loads in the Upper Mississippi, Ohio, Red, and Great Lakes River Basins, 1975–2004: U.S. Geological Survey Scientific Investigations Report 2008–5213, 81 p.

Foreword

The U.S. Geological Survey (USGS) is committed to providing the Nation with credible scientific information that helps to enhance and protect the overall quality of life and that facilitates effective management of water, biological, energy, and mineral resources (*http://www.usgs.gov/*). Information on the Nation's water resources is critical to ensuring long-term availability of water that is safe for drinking and recreation and is suitable for industry, irrigation, and fish and wildlife. Population growth and increasing demands for water make the availability of that water, now measured in terms of quantity and quality, even more essential to the long-term sustainability of our communities and ecosystems.

The USGS implemented the National Water-Quality Assessment (NAWQA) Program in 1991 to support national, regional, State, and local information needs and decisions related to water-quality management and policy (*http://water.usgs.gov/nawqa*). The NAWQA Program is designed to answer: What is the condition of our Nation's streams and groundwater? How are conditions changing over time? How do natural features and human activities affect the quality of streams and groundwater, and where are those effects most pronounced? By combining information on water chemistry, physical characteristics, stream habitat, and aquatic life, the NAWQA Program aims to provide science-based insights for current and emerging water issues and priorities. From 1991-2001, the NAWQA Program completed interdisciplinary assessments and established a baseline understanding of water-quality conditions in 51 of the Nation's river basins and aquifers, referred to as Study Units (*http://water.usgs.gov/nawqa/studyu.html*).

In the second decade of the Program (2001–2012), a major focus is on regional assessments of water-quality conditions and trends. These regional assessments are based on major river basins and principal aquifers, which encompass larger regions of the country than the Study Units. Regional assessments extend the findings in the Study Units by filling critical gaps in characterizing the quality of surface water and groundwater, and by determining status and trends at sites that have been consistently monitored for more than a decade. In addition, the regional assessments continue to build an understanding of how natural features and human activities affect water quality. Many of the regional assessments employ modeling and other scientific tools, developed on the basis of data collected at individual sites, to help extend knowledge of water quality to unmonitored, yet comparable areas within the regions. The models thereby enhance the value of our existing data and our understanding of the hydrologic system. In addition, the models are useful in evaluating various resource-management scenarios and in predicting how our actions, such as reducing or managing nonpoint and point sources of contamination, land conversion, and altering flow and (or) pumping regimes, are likely to affect water conditions within a region.

Other activities planned during the second decade include continuing national syntheses of information on pesticides, volatile organic compounds (VOCs), nutrients, selected trace elements, and aquatic ecology; and continuing national topical studies on the fate of agricultural chemicals, effects of urbanization on stream ecosystems, bioaccumulation of mercury in stream ecosystems, effects of nutrient enrichment on stream ecosystems, and transport of contaminants to public-supply wells.

The USGS aims to disseminate credible, timely, and relevant science information to address practical and effective water-resource management and strategies that protect and restore water quality. We hope this NAWQA publication will provide you with insights and information to meet your needs, and will foster increased citizen awareness and involvement in the protection and restoration of our Nation's waters.

The USGS recognizes that a national assessment by a single program cannot address all water-resource issues of interest. External coordination at all levels is critical for cost-effective management, regulation, and conservation of our Nation's water resources. The NAWQA Program, therefore, depends on advice and information from other agencies—Federal, State, regional, interstate, Tribal, and local—as well as nongovernmental organizations, industry, academia, and other stakeholder groups. Your assistance and suggestions are greatly appreciated.

Matthew C. Larsen
Associate Director for Water

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Conversion Factors and Datum

Multiply	By	To obtain
	Length	
centimeter (cm)	0.3937	inch (in.)
meter (m)	3.281	foot (ft)
meter (m)	1.094	yard (yd)
kilometer (km)	0.6214	mile (mi)
kilometer (km)	0.5400	mile, nautical (nmi)
	Area	
square kilometer (km^2)	247.1	acre
square kilometer (km^2)	0.3861	square mile (mi^2)
	Volume	
cubic meter (m^3)	35.31	cubic foot (ft^3)
cubic meter (m^3)	1.308	cubic yard (yd^3)
	Flow rate	
cubic meter per second (m^3/s)	22.83	million gallons per day (Mgal/d)
cubic meter per second (m^3/s)	70.07	acre-foot per day (acre-ft/d)
cubic meter per second (m^3/s)	35.31	cubic foot per second (ft^3/s)
	Mass	
kilogram (kg)	2.205	pound avoirdupois (lb)
kilogram per year (kg/yr)	2.205	pound per year (lb/yr)
	Application rate	
kilograms per square kilometer (kg/km^2)	0.8514	pounds per square mile (lb/mi^2)

Horizontal coordinate information is referenced to the North American Datum of 1983 (NAD83).

Concentrations of chemical constituents in water are given either in milligrams per liter (mg/L) or micrograms per liter (μg/L).

Water year is the 12-month period, October 1 through September 30, and is designated by the calendar year in which it ends. Thus, the water year ending September 30, 2003 is called “water year 2003.”

Abbreviations used in report

Abbreviation	Description
River basins	
GL	Great Lakes
MAR	Mississippi-Atchafalaya River
OH	Ohio River, exclusive of the Tennessee River
RR	Red River including the Red, Rainy, and Souris Rivers
UM	Mississippi River upstream from confluence with the Ohio River, exclusive of the Missouri River
Constituents	
DP	Dissolved phosphorus
KJN	Total (whole water) organic plus ammonia nitrogen (Kjeldahl)
NH4	Dissolved ammonia
NO2NO3	Dissolved nitrite plus nitrate
SMAT	Total suspended material
SS	Total suspended sediment
TN	Total (whole water) nitrogen
TP	Total (whole water) phosphorus
Equation symbols	
β	estimated coefficient
C	log concentration
e	base of natural logarithms
F	unit conversion factor
FAC	log of the flow-adjusted concentration
HFV	high-frequency variability component
L	daily load
Q	log streamflow
Q^a	center value for streamflow
Q^*	centered log of streamflow
T^m	midpoint of the period of record
T^a	centering value of time
$\%\triangle FAC/yr$	percent change in flow-adjusted trend per year

Abbreviations used in report—continued

Abbreviation	Description
Other abbreviations	
<	less than
≤	less than or equal to
>	greater than
≥	greater than or equal to
AIC	Akaike information criterion
CAFO	confined animal feeding operations
FAC	flow-adjusted concentration
FA trend	flow-adjusted trend
MDL	method detection limit
NAWQA	National Water-Quality Assessment
NWIS	National Water Information System
NWQL	National Water Quality Laboratory
OA trend	overall trend in concentration
QL	quantitation limit
USEPA	U.S. Environmental Protection Agency
USGS	U.S. Geological Survey

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Abstract

Many actions have been taken to reduce nutrient and suspended-sediment concentrations and the amount of nutrients and sediment transported in streams as a result of the Clean Water Act and subsequent regulations. This report assesses how nutrient and suspended-sediment concentrations and loads in selected streams have changed during recent years to determine if these actions have been successful.

Flow-adjusted and overall trends in concentrations and trends in loads from 1993 to 2004 were computed for total nitrogen, dissolved ammonia, total organic nitrogen plus ammonia, dissolved nitrite plus nitrate, total phosphorus, dissolved phosphorus, total suspended material (total suspended solids or suspended sediment), and total suspended sediment for 49 sites in the Upper Mississippi, Ohio, Red, and Great Lakes Basins. Changes in total nitrogen, total phosphorus, and total suspended-material loads were examined from 1975 to 2003 at six sites to provide a longer term context for the data examined from 1993 to 2004.

Flow-adjusted trends in total nitrogen concentrations at 19 of 24 sites showed tendency toward increasing concentrations, and overall trends in total nitrogen concentrations at 16 of the 24 sites showed a general tendency toward increasing concentrations. The trends in these flow-adjusted total nitrogen concentrations are related to the changes in fertilizer nitrogen applications. Flow-adjusted trends in dissolved ammonia concentrations from 1993 to 2004 showed a widespread tendency toward decreasing concentrations. The widespread, downward trends in dissolved ammonia concentrations indicate that some of the ammonia reduction goals of the Clean Water Act are being met. Flow-adjusted and overall trends in total organic plus ammonia nitrogen concentrations from 1993 to 2004 did not show a distinct spatial pattern. Flow-adjusted and overall trends in dissolved nitrite plus nitrate concentrations from 1993 to 2004 also did not show a distinct spatial pattern.

Flow-adjusted trends in total phosphorus concentrations were upward at 24 of 40 sites. Overall trends in total phosphorus concentrations were mixed and showed no spatial pattern. Flow-adjusted and overall trends in dissolved phosphorus concentrations were consistently downward at all of the sites in the eastern part of the basins studied. The reduction in phosphorus fertilizer use and manure production east of the Mississippi River could explain most of the observed trends in dissolved phosphorus.

Flow-adjusted trends in total suspended-material concentrations showed distinct spatial patterns of increasing tendencies throughout the western part of the basins studied and in Illinois and decreasing concentrations throughout most of Wisconsin, Iowa, and in the eastern part of the basins studied. Flow-adjusted trends in total phosphorus were strongly related to the flow-adjusted trends in suspended materials. The trends in the flow-adjusted suspended-sediment concentrations from 1993 to 2004 resembled those for suspended materials.

The long-term, nonmonotonic trends in total nitrogen, total phosphorus, and suspended-material loads for 1975 to 2003 were described by local regression, LOESS, smoothing for six sites. The statistical significance of those trends cannot be determined; however, the long-term changes found for annual streamflow and load data indicate that the monotonic trends from 1993 to 2004 should not be extrapolated backward in time.

Introduction

Elevated nutrient and suspended-sediment concentrations are the two of the most common contaminant stressors affecting streams throughout the United States (U.S. Environmental Protection Agency, 1996). Excessive nutrients in rivers and streams can result in the overgrowth of benthic algae in shallow areas and areas with fast current and in an overabundance of phytoplankton and macrophytes in deep areas with slow current. High algal and macrophyte biomass, in turn, can cause severe diurnal fluctuations in dissolved oxygen and pH because of biotic production and respiration. Very low dissolved oxygen concentrations often occur when part of the algal or macrophyte population dies and decomposes, which in turn can generate harmful organic materials (Welch and others, 1992). Therefore, high nutrient concentration conditions can lead to an increase in the availability of toxic substances, a decrease in available aquatic habitat, modifications to the composition of the biotic communities, and a decrease in the overall usefulness of the stream (Miltner and Rankin, 1998; Dodds and Welch, 2000). Excessive transport of nutrients has also been linked to eutrophication of downstream lakes and impoundments, outbreaks of *Pfiesteria* in bays and estuaries in various Gulf and Mid-Atlantic States, and hypoxia in the Gulf of Mexico (U.S. Environmental Protection Agency, 2000). Suspended sediment reduces clarity in streams and affects sight-feeding fish. Suspended sediment also interferes with water-treatment processes and recreational uses of streams. Excessive siltation can bury and suffocate fish eggs and bottom-dwelling organisms. In addition to in-stream effects, excessive sediment loading causes sedimentation problems in many downstream lakes and harbors and water-clarity problems in nearshore areas.

Many actions have been taken to reduce nutrient and suspended-sediment concentrations and the amount or load of nutrients and sediment transported in streams. In 1972, the Federal Water Pollution Control Act Amendments, commonly known as the Clean Water Act, were passed, and they established the basic structure for regulating discharges of pollutants into the waters of the United States (U.S. Environmental Protection Agency, 2006a). As result of the Clean Water Act, governmental efforts in the 1970s and 1980s primarily focused on regulating discharges from traditional “point source” facilities, such as municipal sewage plants and industrial facilities. In the 1980s and 1990s, the focus expanded to address nonpoint runoff, in other words, losses from agricultural areas, such as feedlots and fields, and losses from urban areas, such as construction sites and urban storm sewers. Under recommendations of the Clean Water Action Plan released in 1998, the U.S. Environmental Protection Agency (USEPA) developed a national strategy to develop waterbody-specific nutrient criteria for lakes and reservoirs, rivers and streams, wetlands, and estuaries (U.S. Environmental Protection Agency, 1998). The intent of this strategy is to require all States and tribes to establish nutrient standards that, if enforced, will continue to reduce nutrient concentrations and improve the beneficial ecological uses of surface waters. To determine if these actions have been successful, it is important to assess how nutrient and suspended-sediment concentrations and loads in streams have changed during recent years.

Changes in the concentrations of nutrients and suspended sediment in streams are the result of anthropogenic changes in the watershed, such as changes in the levels of sewage treatment or implementation of best management practices, and the result of changes in natural factors, such as streamflow or climatic change. To better detect changes as the result of anthropogenic factors, it is best to remove or factor out the variability associated with the natural factors. One of the most dominant natural factors affecting nutrient and suspended sediment in streams is streamflow. Concentrations of many water-quality constituents increase with increasing streamflow, especially those associated with particulates, whereas concentrations of others can decrease with increasing streamflow because of dilution. Therefore, when data are analyzed for trends, it is important to compensate for variability in streamflow.

Identification and analysis of trends in streamflow is of great interest to water-resource managers, scientists, and the public for a variety of important reasons. Long-term trends in streamflow potentially affect water levels in lakes and reservoirs; the availability of habitat for aquatic organisms; the availability of water for anthropogenic uses including recreation, irrigation, water supply, and waste disposal; aspects of bridge and dam construction; and flooding of low lying land. Trends in streamflow caused by climate may magnify the effects of land-use changes associated with anthropogenic activities, such as when increased precipitation causes flooding that is exacerbated by the increased amount of impervious areas associated with increased urbanization. Therefore, it is important to identify the trends in streamflow in addition to the trends in water quality.

Although many actions have been taken to reduce the concentrations and loads of nutrients and suspended sediments in streams across the country, most studies that have tried to evaluate whether or not these actions have been successful have examined the water quality of only a few streams from a specific area (such as Corsi and others, 2005) or within a specific State (such as Ruppel, 2006, for Wisconsin or Zipper and others, 1998, for Virginia). Only a few studies have examined trends over large geographical areas. For example, Stoddard and others (2003) examined water quality in streams from selected regions of the eastern and midwestern areas of the United States to assess changes in water quality as the result of recent reductions in nitrogen emissions.

To quantify how streamflow and nutrient and suspended-sediment concentrations and loads in streams across the country have changed as result of changes in land use and management practices, and the many actions taken to improve water quality, regional studies were initiated by the National Water-Quality Assessment (NAWQA) Program of the U.S. Geological Survey (USGS). The objectives of the regional studies are to describe changes or trends in streamflow and the associated trends in nutrient and suspended-sediment concentrations and loads that have occurred during a 1993–2004 base period and to try to understand what may have caused any of the substantial changes in water quality. Changes in total nitrogen, total phosphorus, and total suspended-material loads were examined from 1975 to 2003 at six sites to provide a longer term context for the data examined from 1993 to 2004.

The purpose of this report is to describe trends in streamflow and concentrations and loads of selected nutrients including total nitrogen, dissolved ammonia, dissolved nitrite plus nitrate, total organic plus ammonia nitrogen (Kjeldahl), total phosphorus, and dissolved phosphorus; total suspended materials; and total suspended sediment from 1993 to 2004 in streams at 49 selected sites within the north-central part of the

United States. The report describes what may have caused the changes in water quality by examining changes in the major sources of nutrients in this area. Changes in concentrations and loads from 1975 to 2003 for six of these sites, which are distributed throughout the basins studied, are examined to place the trends observed for the 1993–2004 base period into a longer term context.

Study Area and Environmental Setting

The study area comprises four major river basins: the Upper Mississippi, Ohio, Red, and Great Lakes Basins (fig. 1). The Upper Mississippi River (UM) Basin is defined as the Mississippi River and its tributaries upstream from the confluence with the Ohio River, excluding the Missouri River. The Ohio River begins in eastern Pennsylvania at the confluence of the Allegheny and Monongahela Rivers and flows westward until it joins the Mississippi at Cairo, Ill. The Ohio River (OH) Basin includes the Ohio River and its tributaries, excluding the Tennessee River. The Red River (RR) Basin includes all rivers in the United States that drain northward to the southern basin of Lake Winnipeg, including the Red River of the North, the Souris River, and the Rainy River. The Great Lakes (GL) Basin includes all streams in the United States draining into the Great Lakes upstream from Montreal, Canada. Lake Michigan was joined with the Illinois River (UM Basin) through construction of the Chicago Sanitary and Ship Canal in 1900, a project engineered to reverse the flow of the Chicago River in order to divert Chicago wastewaters into the Upper Illinois River and away from the historic receiving water of Lake Michigan.

Factors Affecting Nutrient and Suspended-Sediment Concentrations and Loads

Various factors may cause changes in water-quality concentrations and yields such as climatic factors (precipitation and resulting runoff), land-use practices, and changes in the inputs of nutrients to the basin. To determine how these factors may have affected water quality, the distribution of each of these factors throughout the study area and how they have changed from 1993 to 2004 are described in the following sections.

Precipitation and Runoff

Precipitation and resulting runoff are the major factors that influence transport of nutrients from the land surface to streams. Precipitation ranges from about 38 cm/yr in the western part of the study area to about 150 cm/yr in the eastern part (fig. 2). The largest relative differences in precipitation are in the western part of the study area, and the smallest differences are in the eastern part. In the western part of the area, average precipitation ranges from about 38 to 80 cm/yr over about 400 km (Winterstein and Lorenz, 2007). Evaporation ranges from about 97 to 102 cm/yr in the southwestern part of the study area to about 71 cm/yr in the north-central and southeastern parts of the study area (Farnsworth and others, 1982). These geographical differences in precipitation and evaporation result in a wide range in runoff from less than 2.5 cm/yr in the northwest to about 75 cm/yr in the southeastern part of the study area (fig. 3). This wide range in runoff results in a large variability in average streamflow volume and in loads of various water-quality constituents.

The regional precipitation map (fig. 2) can be used to divide the study area into three different geographic/climatic parts: the western, central, and eastern parts. The western part of the study area includes parts of North Dakota, South Dakota, and northwestern Minnesota. The climate is subhumid continental with warm summers and cold winters, and annual precipitation ranges from approximately 36 to 50 cm. The central part of the study area includes most of the UM Basin in addition to areas draining to the Great Lakes, including parts of Minnesota, Wisconsin, Iowa, Illinois, and Indiana. The central part has a predominantly humid continental climate with annual rainfall ranging from approximately 50 cm in Minnesota to 115 cm in southern Indiana. The eastern part of the study area includes the Upper Ohio River Basin in addition to other drainages to the east and includes Ohio and parts of Pennsylvania and New York. This area also is humid continental and has annual precipitation ranging from approximately 85 cm in Ohio to 115 cm in Pennsylvania and New York. The northern regions of the central and eastern parts of the study area have warm summers and cold winters. The southern regions of the central and eastern parts of the study area have hot summers and cool winters.

Precipitation and associated streamflow generally has increased during the 20th century in the central part of the study area bordering the Great Lakes and UM Basin (Lins, 2005; Hodgkins and others, 2007). Many studies have predicted greater climatic variability in the future that could have far-reaching effects on hydrology, such as earlier and smaller annual snowmelt and increased evapotranspiration; on fish, aquatic vegetation, and agriculture; and, in the event of lower lake levels, disruptions of shipping and commerce (Sousounis and Bisanz, 2000).

Annual precipitation from 1975 to 2005 is shown for five sites in the study area (one site in the western part, two sites in the central part, and two sites in the eastern part) in figure 4, in which the period from 1993 to 2004 is identified. Precipitation in the western part increased during the period from 1993 to 2004 as shown by the data from Grand Forks, N. Dak. During 1993 to 2004, precipitation was variable in the central and eastern parts of the study area, except for a slight decrease in the precipitation at LaCrosse, Wis. The changes from 1993 to 2004 in the western part of the study area appear to be part of a long-term increase in precipitation; whereas, no consistent long-term change in precipitation is apparent in any other part of the study area.

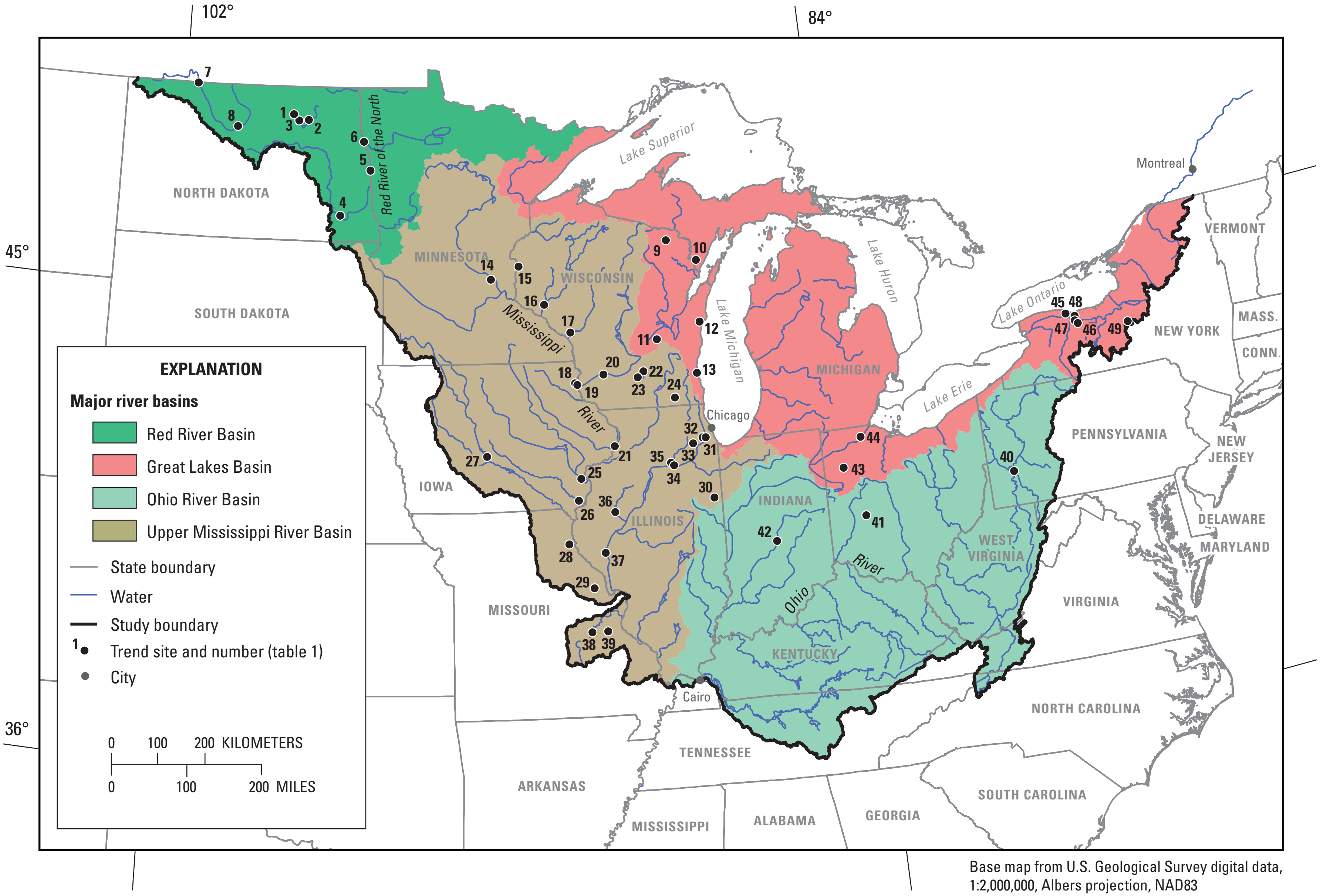


Figure 1. Study area, major river basins, and sites. Sites are described in detail in table 1.

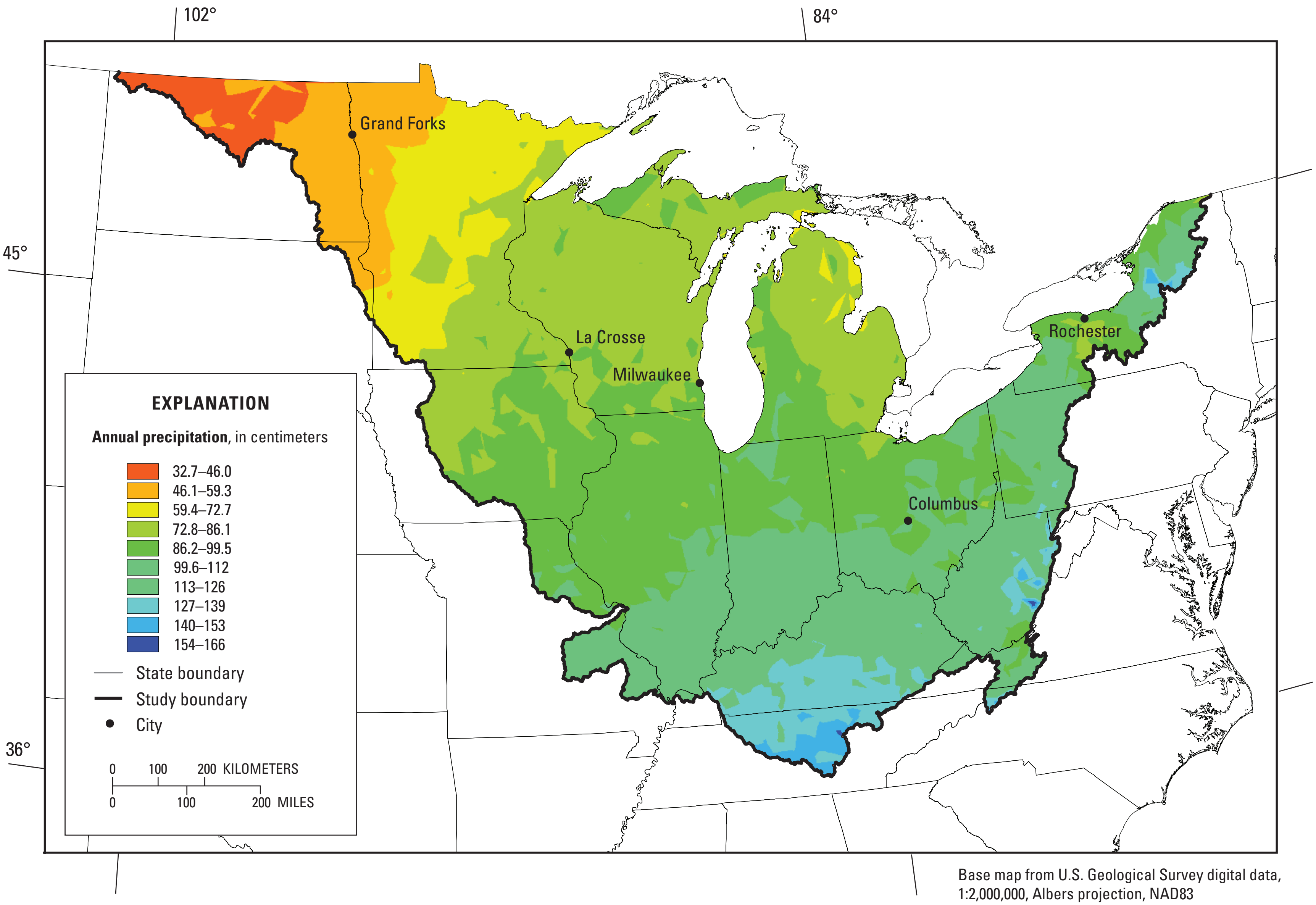


Figure 2. Average annual precipitation across the study area, 1971 to 2000 (National Climatic Data Center, 2002).

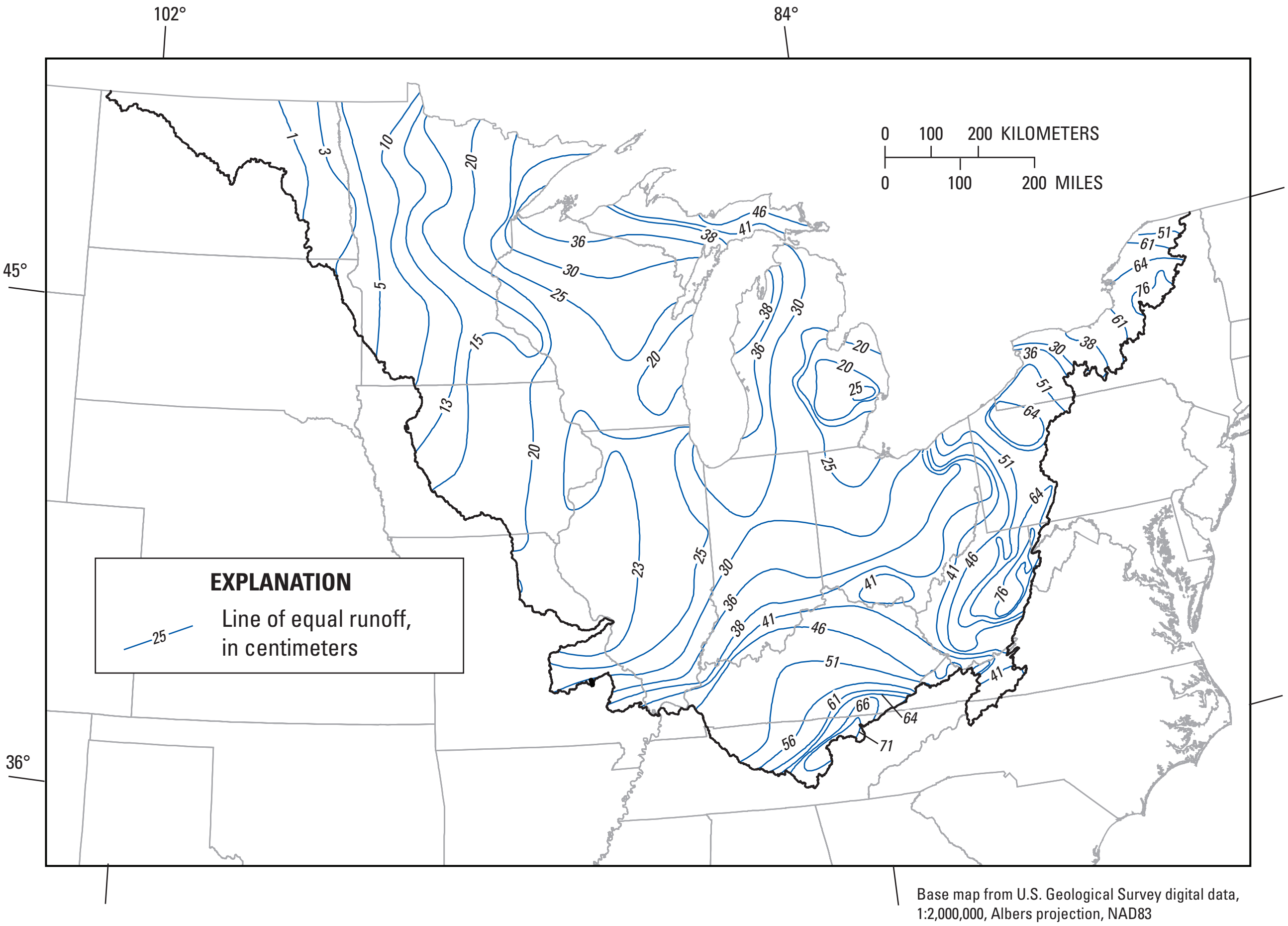


Figure 3. Average annual runoff across the study area, 1951 to 1980 (modified from Gebert and others, 1987).

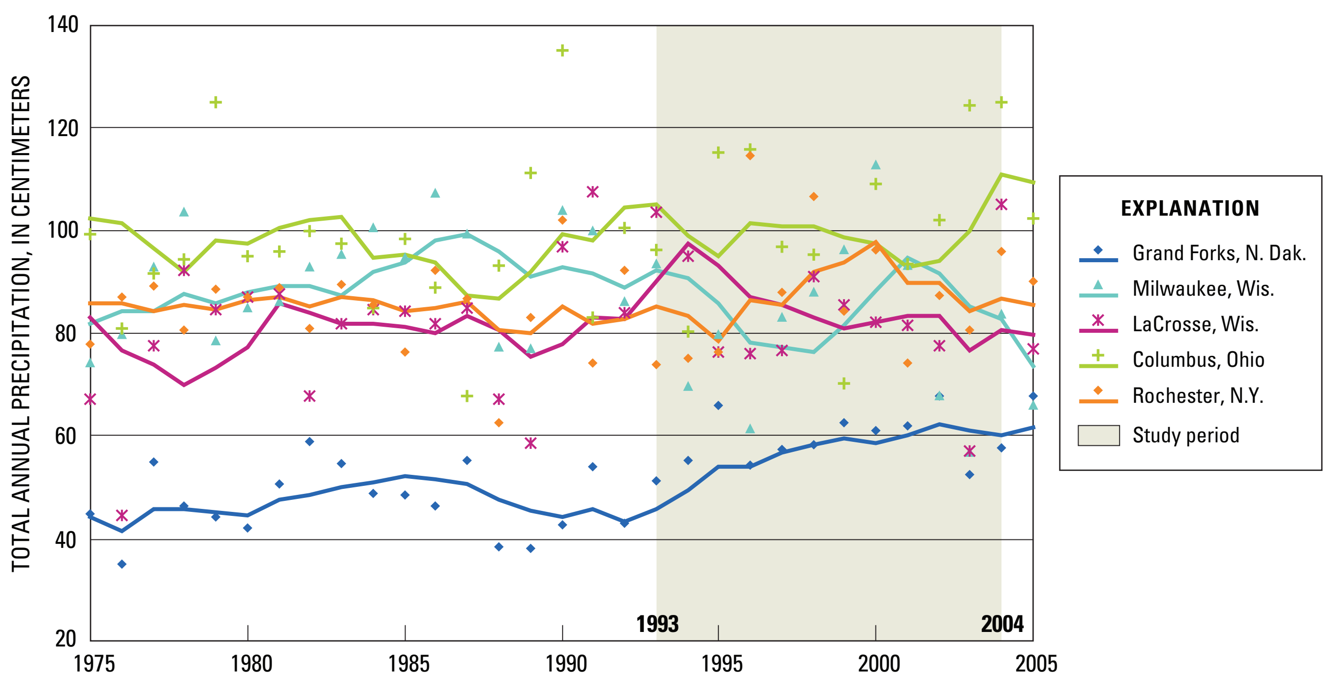


Figure 4. Annual precipitation from 1975 to 2005 at five sites in the study area.

Land Use

Land use across the study area ranges from almost completely forested areas that are relatively unaffected by human activities in the north-central and southeastern parts, to extensively agricultural areas in the central part, to intensely urbanized areas around Chicago, Ill.; Milwaukee, Wis.; Minneapolis, Minn.; Detroit, Mich.; Cleveland, Columbus, and Cincinnati, Ohio; Pittsburgh, Penn.; Indianapolis, Ind.; and St. Louis, Mo. (fig. 5; U.S. Geological Survey, 2000). Population density by county is shown in fig. 6***A*** (Geolytics, 2006). The extensive row-crop agricultural area (fig. 5) is known as the "corn belt," where the major crops are corn and soybeans. Small grains, primarily wheat and barley, are important crops in the western parts of the study area. These differences in land use result in differences in the amount of nutrients applied to the land surface and, potentially, the amount of nutrients that may reach and be transported down a stream and the amount of sediment eroded from the land.

The largest populations are found throughout the center of the study area, especially near the larger cities. Human population censuses, a surrogate for nutrients in wastewater effluent, were available for only 1990 and 2000. From 1990 to 2000, the largest increases in population have occurred in the counties surrounding the largest cities in the central parts of Minnesota, Wisconsin, in the northern part of Lower Michigan, and the southeastern part of the study area, whereas populations have decreased in the far northwestern part of the study area (fig. 6***B***). Changes in population may have caused changes in the magnitude of point-source discharges of nutrients from these areas; however, improvements in wastewater facilities may have offset some water-quality effects attributable to the increased population.

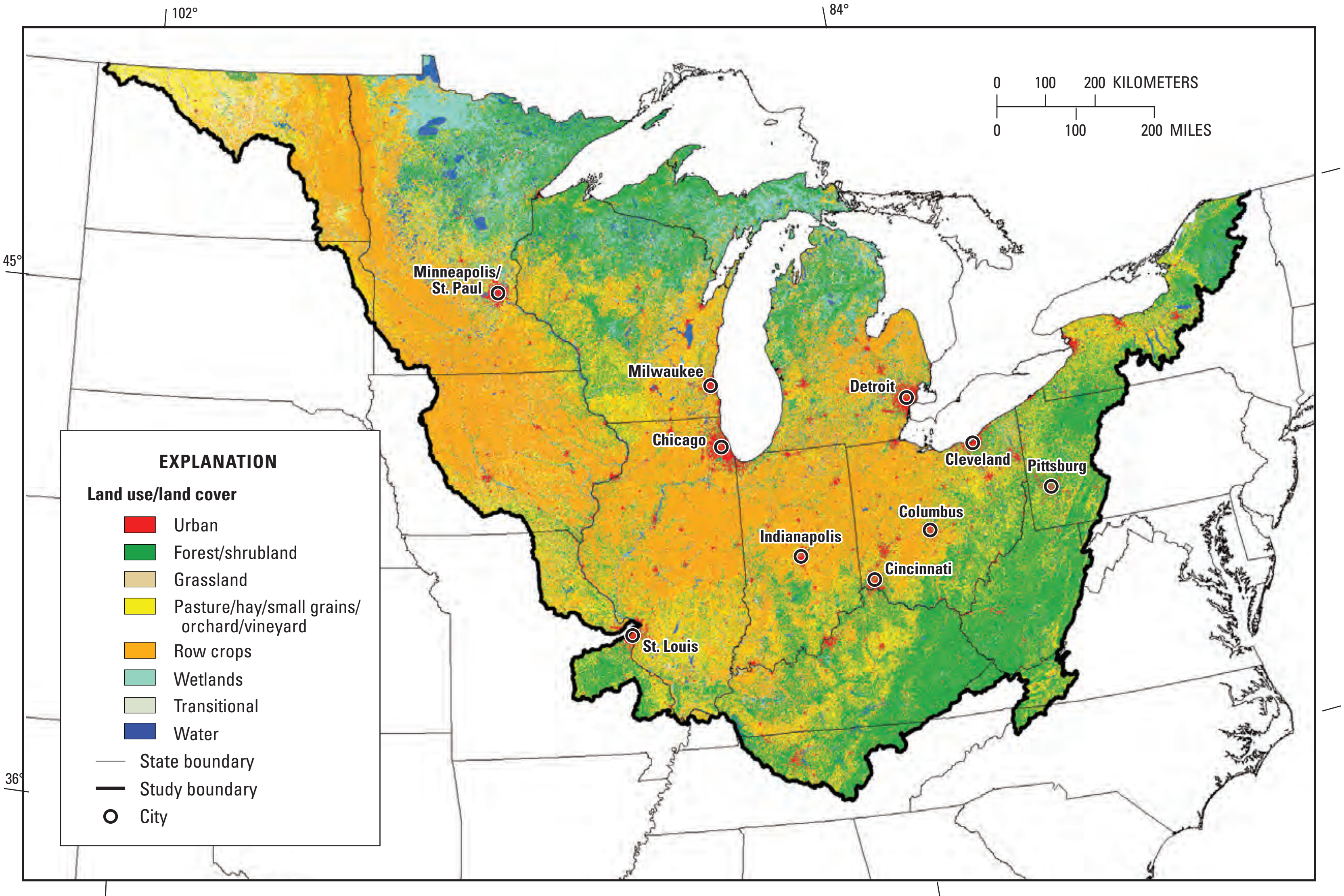
Fertilizers and Manure

Farmers apply fertilizers and manure to increase crop production. When those sources exceed crop needs or have not been incorporated into the soil before rain, nutrients can drain into streams (U.S. Environmental Protection Agency, 2005). Rates of total fertilizer nitrogen application are shown in fig. 7***A*** by county for 1997, which was the middle of the study period (Ruddy and others, 2006). The distribution in nitrogen fertilizer use closely resembles the intensity of agriculture across the study area (fig. 5). The highest fertilizer application rates were in the central and western parts of the study area and encompass nearly all of Indiana, Illinois, and Iowa. The distribution of the rates of total fertilizer phosphorus application (not shown) closely resembles that for nitrogen. Changes in the annual fertilizer application were quite variable; therefore, the changes in fertilizer inputs were obtained by computing the change in the average inputs from 1993–95 to 2002–04. From about 1993 to 2004, rates of nitrogen fertilizer use generally have increased in northwestern North Dakota, and northeastern Minnesota, and most of Wisconsin, whereas rates of nitrogen use generally have decreased in the eastern part of the study area and in the eastern part of North Dakota (fig. 7***B***). From 1993 to 2004, rates of fertilizer phosphorus use generally have increased west of the Mississippi River, except in eastern part of North Dakota, and generally decreased east of the Mississippi River (fig. 7***C***).

The rates of total manure nitrogen application are shown for 1997 by county in figure 8***A*** (Ruddy and others, 2006). The highest application rates were in Iowa, southern Minnesota, southern Wisconsin and central Kentucky. The distribution of the rates of total manure phosphorus application closely resembles that for nitrogen (not shown). Manure data were available for only 1992, 1997, and 2002. From 1992 to 2002, the rates of manure nitrogen application have decreased by more than 15 percent throughout almost all of the study area except near the Iowa-Minnesota border, where the rates of application increased more than 25 percent (fig. 8***B***). From 1992 to 2002, the rates of manure phosphorus application decreased by more than 15 percent throughout almost all of the study area except near the Iowa-Minnesota border, where the rates of application have increased more than 25 percent (fig. 8***C***).

Atmospheric Deposition

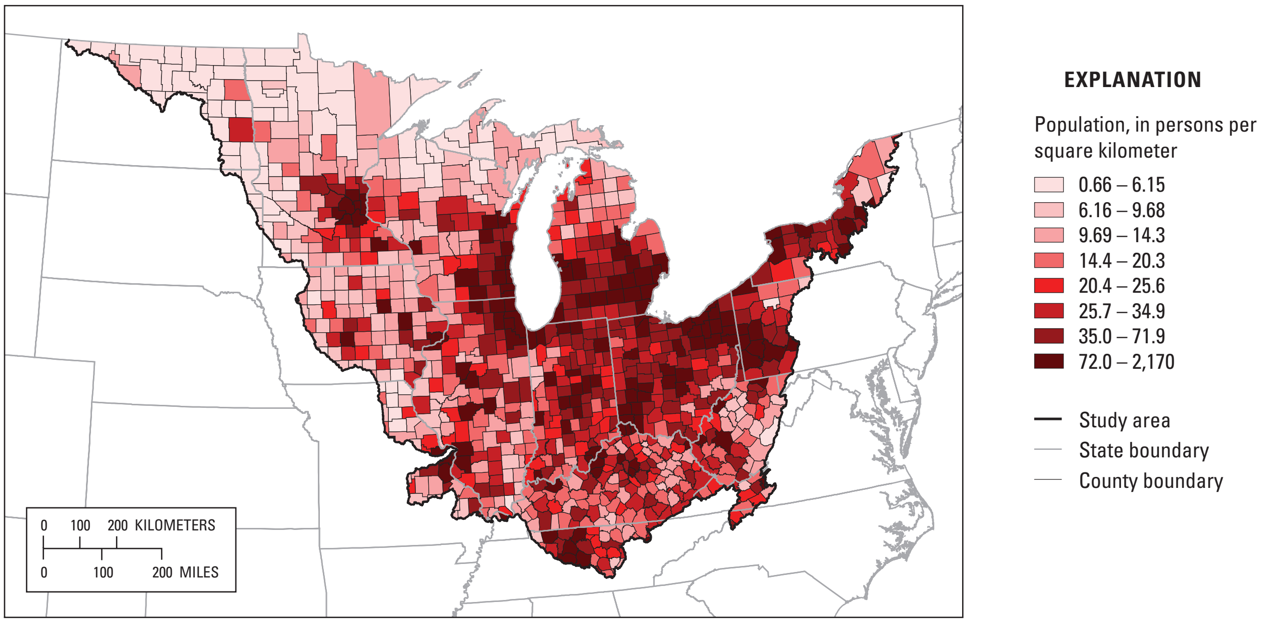
Atmospheric deposition can be an important source of nutrients to the land surface and to streams, especially in forested areas where there are few anthropogenic sources (U.S. Environmental Protection Agency, 2002). Atmospheric deposition of nitrogen can be elevated downwind of some types of agricultural animal production (Langland, 1992) and downwind of areas with fossil fuel combustion (Mason and others, 2002), such as cities. The annual rate of nitrogen input from atmospheric deposition in 1997 ranged from less than 350 kg/km^2 in the northwestern part of the study area to more than 600 kg/km^2 in southern Michigan, northwestern and southern Illinois, northern and southern Indiana, northern and southern Ohio, northeastern and western Kentucky, and the northeastern part of the study area (fig. 9***A***; Ruddy and others, 2006). Because the rates of atmospheric deposition are fairly consistent from year to year, the change from 1993 to 2004 is based on the values for those two years. From 1993 to 2004, the rates of deposition have decreased or changed very little throughout the southern two thirds of the study area and increased in the far northwestern part of the study area (fig. 9***B***).



Base map from U.S. Geological Survey digital data, 1:2,000,000, Albers projection, NAD83

Figure 5. Land use and land cover across the study area (U.S. Geological Survey, 2000), with the 10 largest metropolitan areas identified.

A. **Population, 2000**



B. **Change in population, 1990–2000**

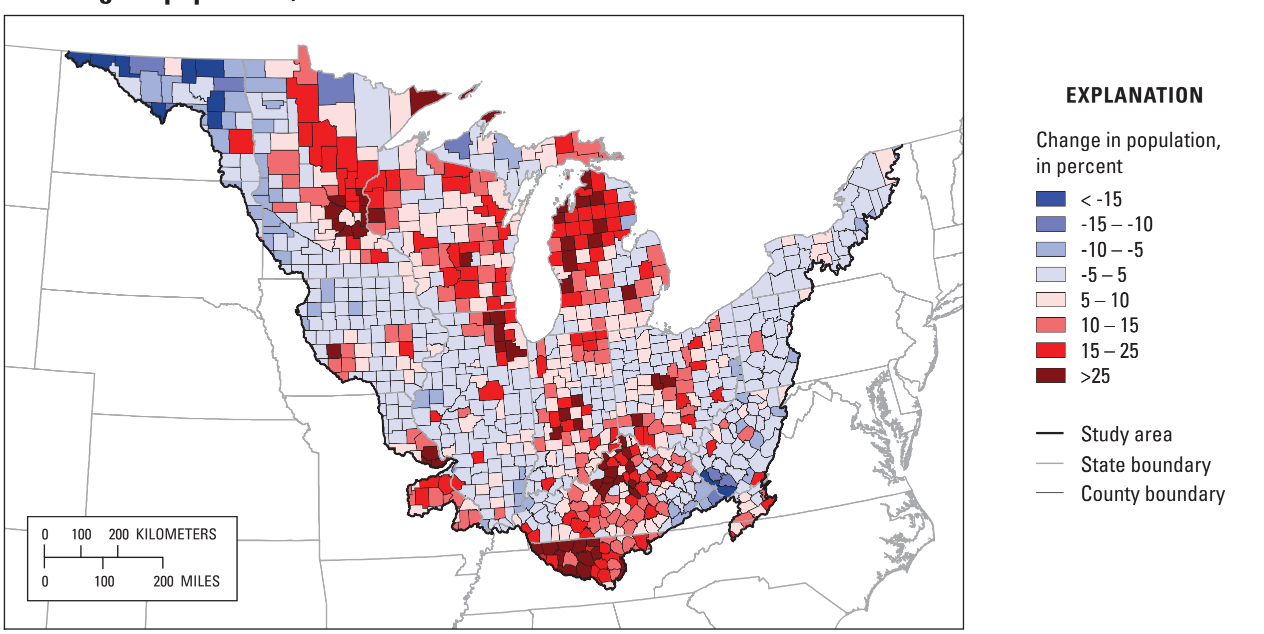


Figure 6. Study area population by ***A***, county in 2000 and ***B***, county-level change from 1990 to 2000 (Geolytics, 2006).

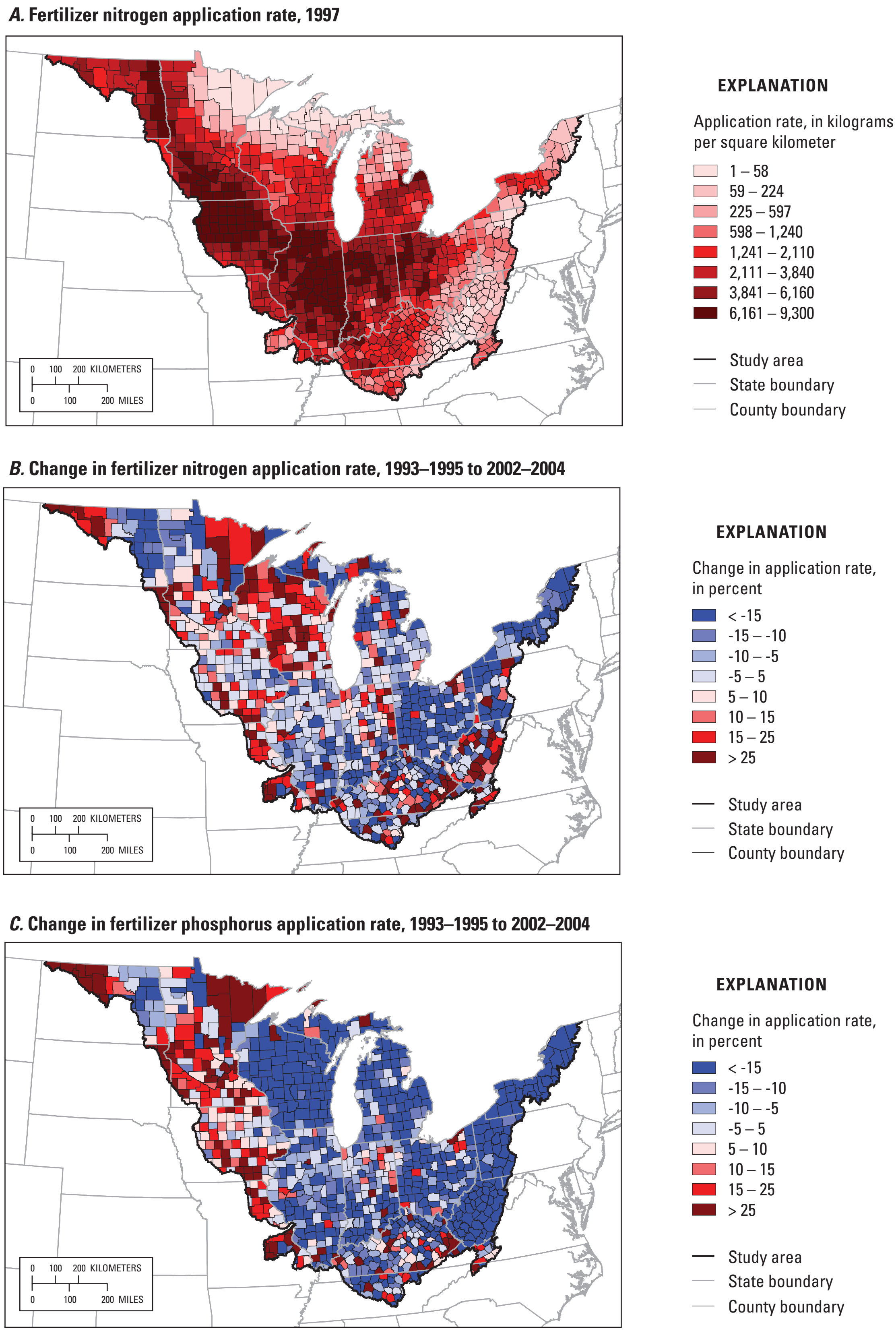


Figure 7. Fertilizer nitrogen application by ***A***, county in 1997 and changes in application rates from 1993 to 2004 for ***B***, nitrogen and ***C***, phosphorus. (Data from Ruddy and others, 2006.)

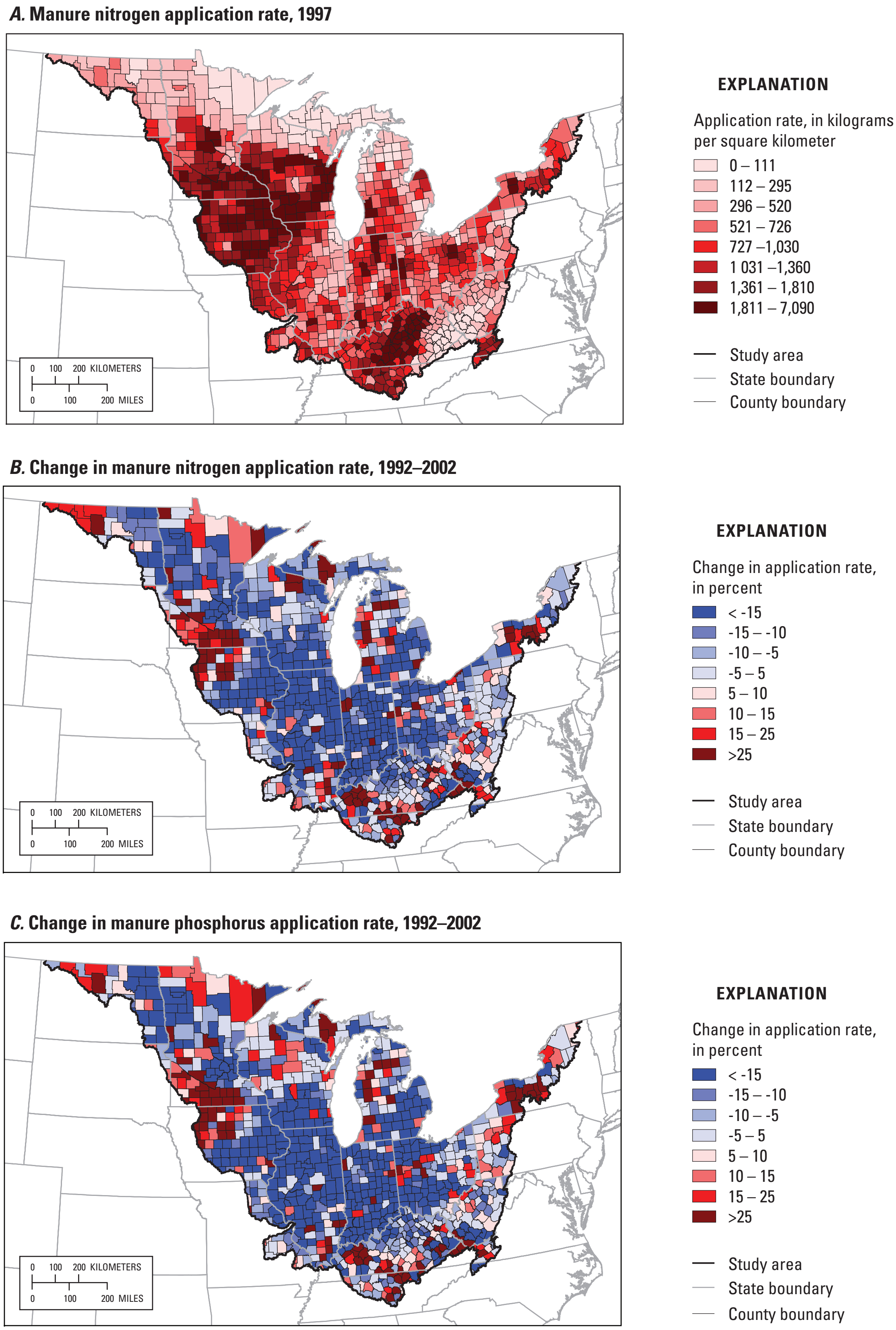
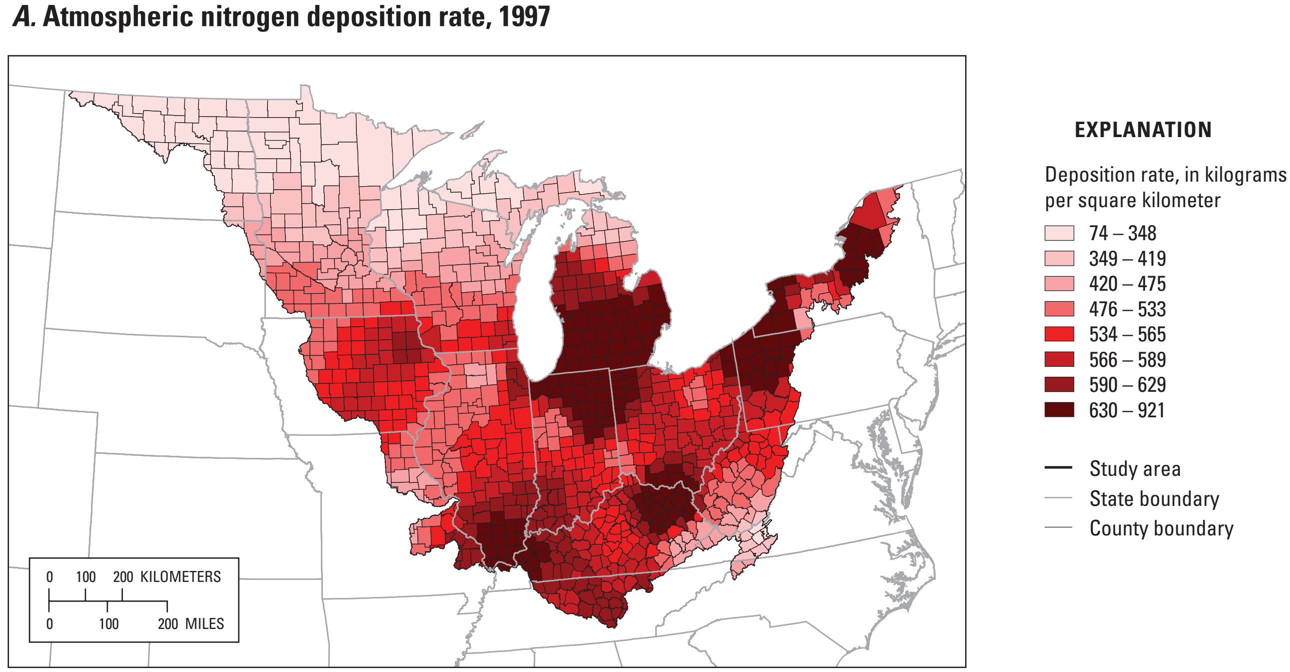


Figure 8. Manure nitrogen application rates by ***A***, county in 1997 and changes in application rates from 1992 to 2002 for ***B***, nitrogen and ***C***, phosphorus. (Data from Ruddy and others, 2006.)



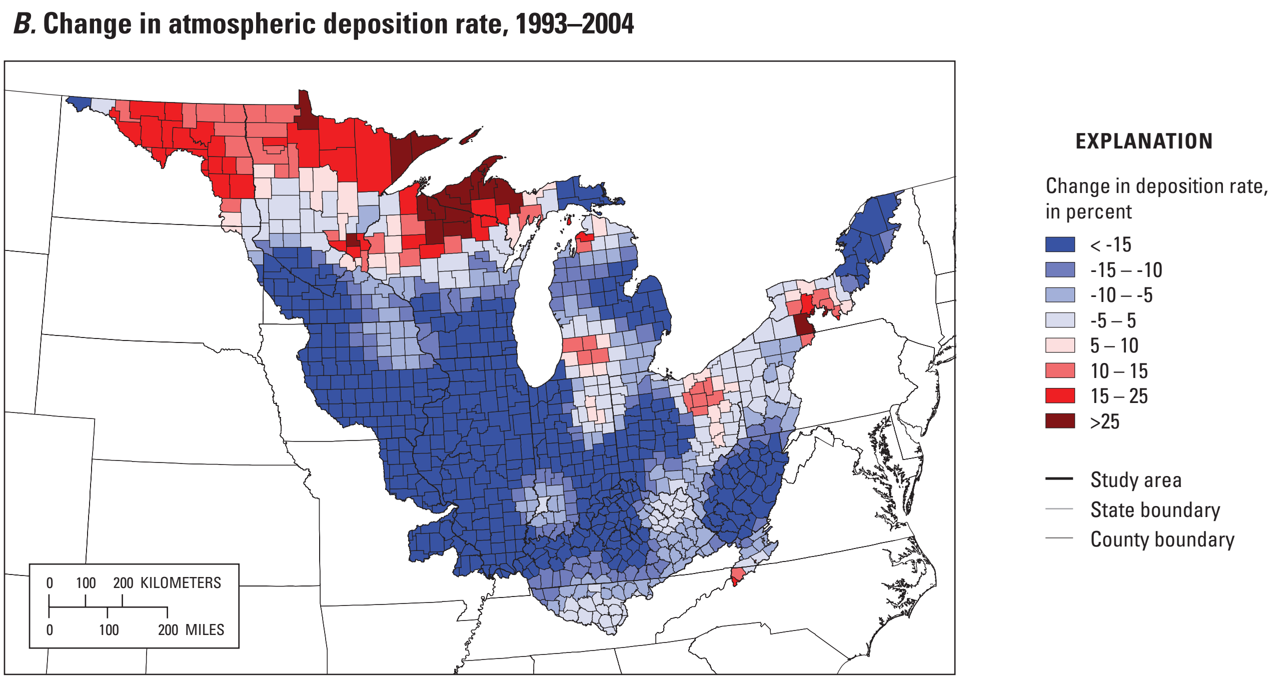


Figure 9. Rates of atmospheric deposition of nitrogen by ***A***, county in 1997 and ***B***, changes in atmospheric deposition rates from 1993 to 2004. (Data from Ruddy and others, 2006.)

Total Nonpoint Sources of Nutrients

The nonpoint sources of nitrogen to streams include fertilizers from agricultural, urban, and suburban areas; manure spread on cropland or from livestock pastures; and atmospheric deposition of nitrogen from fossil fuel combustion and other sources. The pattern of total nitrogen input from nonpoint sources by county for 1997 (fig. 10***A***) closely resembles the amount of row crop agriculture (fig. 5) in the area and application of fertilizer (fig. 7***A***) with highest inputs in the central and western parts of the study area. In general, the input from fertilizers is the dominant source of nitrogen, except in forested areas where the input from atmospheric deposition predominates. The change in the total nonpoint nitrogen input rate from 1993 to 2004 was computed by use of the 1992 and 2002 rates for manure, 3-year average rates around 1993 and 2004 for fertilizers, and rates for 1993 and 2004 for atmospheric deposition. From about 1993 to 2004, the total nonpoint input of nitrogen has increased in western Iowa, southern and northeastern Minnesota, northern Wisconsin, and the western one-half of the Upper Peninsula of Michigan, and decreased in the central and eastern parts of the study area (fig. 10***B***). The changes are not as consistent or widespread as the changes in the individual sources. Most of the total nonpoint-source changes were the result of the changes in fertilizer and manure inputs. In places where these changes were similar—such as increases in western North Dakota, eastern Minnesota, and northern Iowa, and decreases in Ohio—overall changes in nonpoint-source inputs were large. In places where the changes differed in sign, such as in Wisconsin and Illinois, overall changes in nonpoint-source inputs were small.

The distribution in the total input of phosphorus from fertilizers and manure closely resembles that for nitrogen; the input from atmospheric deposition has not been estimated, but is expected to be small compared to other sources of phosphorus. From 1993 to 2004, the total input rates of phosphorus decreased east of the Mississippi River, near the Mississippi River in Missouri and Iowa, and in eastern North Dakota, whereas the input rates increased throughout most of Minnesota, western North Dakota and Iowa, and in and near Tennessee (fig. 10***C***). In most places, the types of the changes in the phosphorus inputs were similar for fertilizers and manure.

The trends in the section “Trends in Nutrients and Suspended-Sediment Concentrations and Loads” are described with respect to patterns shown in figures 5–10 to determine if the changes in the input of nitrogen and phosphorus has affected the water quality, indicated by concentrations and loads, in the streams throughout the study area.

Methods

For this report, a trend is defined as a general change in concentrations or loads over time. Because of the complicating effects of natural and anthropogenic influences on water-quality conditions, it is important to consider three types of trends: (1) flow-adjusted trends in concentrations (FA trend)—the trends in concentrations that would have occurred in the absence of natural variability in streamflow and seasonal effects; (2) total or overall trends in concentrations (OA trend)—the trends that represent changes in the overall effects on the stream ecosystem and include the effects of streamflow and seasonal variability and the effects of anthropogenic activities; and (3) trends in the load of a constituent—trends that directly include the changes in streamflow and the overall changes in concentrations. Changes in constituent loads are most important to downstream water bodies, such as the Gulf of Mexico or Lake Michigan. The following sections describe data, site selection, and trend and load computations.

Water-Quality Data

Water-quality data for trend analyses were limited to the nutrients total nitrogen, dissolved ammonia, total Kjeldahl nitrogen, dissolved nitrite plus nitrate, and total and dissolved phosphorus; and total suspended-material and suspended-sediment data measured in streams in the study area for which sufficient data were available between 1970 and 2004. For the purpose of analysis, suspended sediment was examined as suspended sediment (USGS parameter code 80154) alone and combined with total suspended solids (USGS parameter code 00530) into one constituent, total suspended material (SMAT). All data were compiled from data collected by the USGS and the major sampling agency(s) in each State. USGS data were retrieved from the National Water Information System (NWIS) and the NAWQA Program’s Data Warehouse (*http://water.usgs.gov/nawqa/data*). All of the State-agency data were obtained from USEPA legacy and modernized STOrage and RETrieval databases, known as STORET (*http://www.epa.gov/storet/*), except for data from Illinois, which were collected by the Illinois Environmental Protection Agency and contained in NWIS; Indiana, which were collected by the Indiana Department of Environmental Management (C. Bell, Indiana Department of Environmental Management written commun., 2004) and Wisconsin, which was collected by the Wisconsin Department of Natural Resources (J. Ruppel, Wisconsin Department of Natural Resources written commun., 2004).

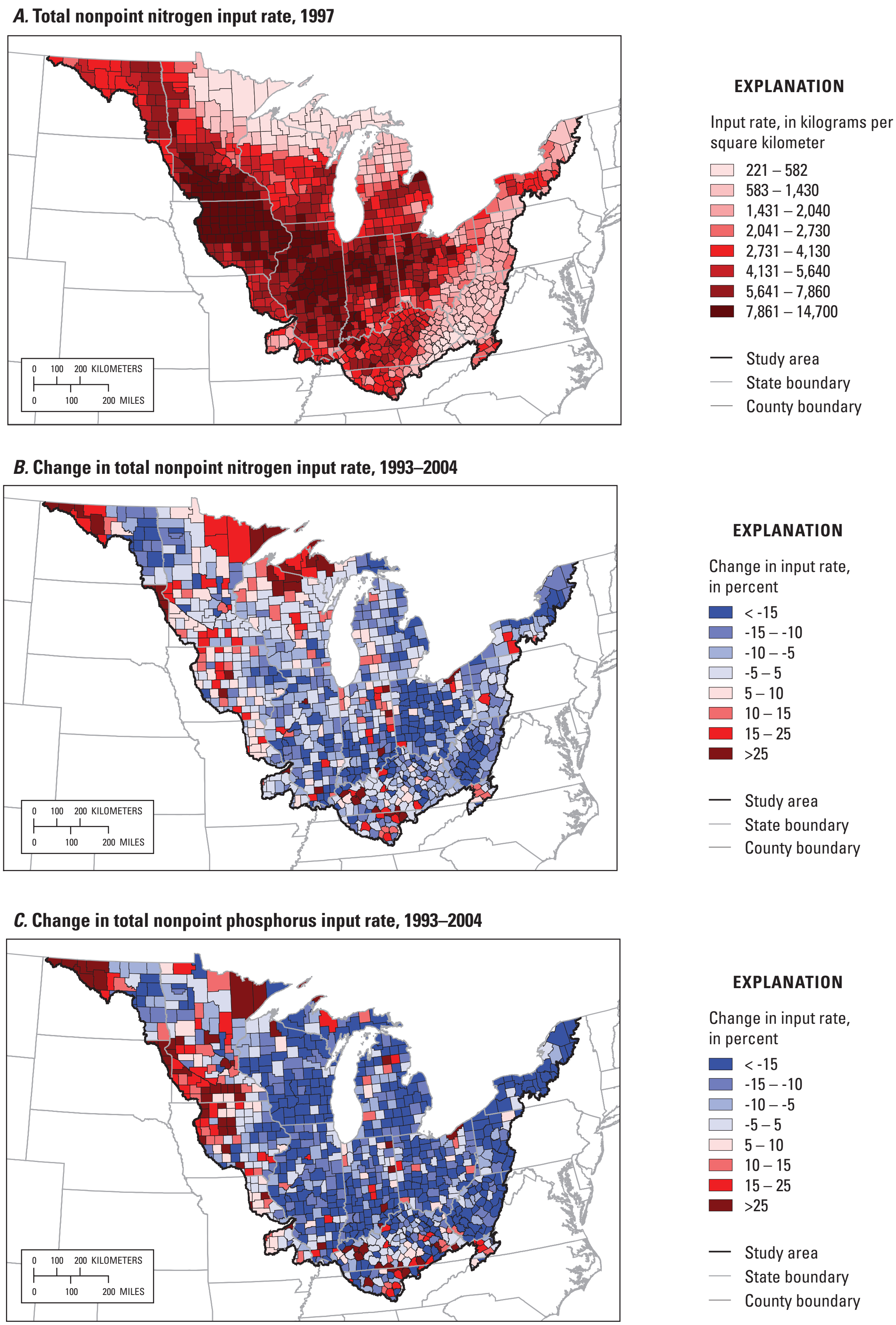


Figure 10. Total annual input rates of nitrogen from nonpoint sources (fertilizers, manure, and atmospheric deposition) by ***A***, county in 1997 and changes in the total input rates from 1993 to 2004 for ***B***, nitrogen and ***C***, phosphorus; atmospheric deposition of phosphorus was assumed to be negligible.

Streamflow Data

For each water-quality site, a nearby streamflow gage was identified. A nearby gage was defined as a gage within the same stream system with a drainage-area ratio, which is the drainage area of the water-quality station divided by the drainage area of the streamflow gage, between about 0.3 and 2.0. A nearby gage with a long record was chosen over a nearby gage with a very short record. At each streamflow gage, all of the available daily streamflow data from 1970 to 2004 were obtained from USGS's Automated Data-Processing System (ADAPS) within NWIS (U.S. Geological Survey, 1998).

Final Site and Sample Selection

After the water-quality data were retrieved and matched to streamflow stations, the water-quality data were reviewed for completeness, quality-assurance samples were removed, and multiple daily samples were reduced to a single sample per day. For all of the multiple daily samples, the value of the constituent reported closest to noon was retained and all others deleted. Final site and sample selection involved three levels of screening. In the first level of screening, only sites with at least 15 samples for at least one constituent from a period of at least 2 years were retained, which resulted in approximately 1,700 sites. Data from the first level of screening were used to describe general patterns in median total nitrogen, total phosphorus, and suspended-material concentrations in streams throughout the entire study area.

A second level of screening was used to identify sites to describe general patterns in median annual yields or loads per unit area. Yields were computed to remove the effects caused by large differences in the size of the streams. Selection requirements for a site to be used to compute median annual yields were that it included at least 25 samples over a period of at least 2 years and at least 5 years of complete daily streamflow records; therefore, a site needed at least 5 years of complete estimated loads to compute a median annual yield. An annual yield was computed only if there were no missing daily flows for that year. The requirement of having 5 years of estimated load data to compute median values was used to reduce the potentially large effects of natural climatic variability. Approximately 700 sites were retained from the second level of screening. Data from the second level of screening were used to describe general patterns in median total nitrogen, total phosphorus, and suspended-material yields throughout the entire study area.

A third level of screening was used to select sites for the trend analyses. In this level, only sites that had at least 10 years of data for at least one constituent between 1993 and 2004, had at least 60 samples, and had a gage or a nearby gage with a drainage-area ratio ranging from 0.9 to 1.1 and streamflow record from 1993 to 2004. A base period of 1993 to 2004 was selected because it is relevant to current conditions and represents the period since the beginning of the USGS NAWQA Program. For the final trend analyses in the base period, 49 sites met the criteria. The sites selected are listed in table 1 and shown on figure 1. In table 1, the sites are divided among the western, central, and eastern parts of the study area. Eight of the sites were in the western part, 31 sites were in the central part, and 10 sites were in the eastern part of the study area.

A fourth level of screening was applied to each of the 49 selected sites for each constituent analyzed. At this level, 10 years of data and at least 60 samples were required for the constituent to be analyzed at the site. For total nitrogen, 24 sites were analyzed; for dissolved ammonia, 36 sites were analyzed; for total Kjeldahl nitrogen, 36 sites were analyzed; for dissolved nitrite plus nitrate, 29 sites were analyzed; for total phosphorus, 41 sites were analyzed; for dissolved phosphorus, 35 sites were analyzed; for SMAT, 42 sites were analyzed; and for total suspended sediment, 22 sites were analyzed.

To put the trends found for 1993 to 2004 into a longer term context, data from a subset of the final 49 sites were analyzed for long-term trends in flow-adjusted concentrations from 1975 to 2004. The trends in the flow-adjusted concentrations at six of these sites, which are distributed throughout the study area (table 1 and fig. 11), are shown graphically for each constituent. One site was in the western part, four sites were in the central part, and one site was in the eastern part of the study area. Of the four sites in the central part, three sites represented relatively small basins, and one site, Mississippi River at Clinton, Iowa, represented a large basin that integrates most of this area. Long-term trends in loads also were examined and shown graphically for these six sites. A description of the drainage basin for each of these six sites follows. Trends for the six long-term sites are presented, in order, on the basis of west to east.

The long-term western site drains the Souris River Basin, which occupies approximately 7,900 km^2 in southeastern Saskatchewan and southwestern Manitoba in Canada and north-central North Dakota in the United States. The Souris River starts in southeastern Saskatchewan, enters the United States and flows southward before looping back into Canada. Most of the drainage basin of the Souris River near Sherwood, N. Dak., is in Canada and contains substantial forested, prairie, and agricultural (mostly grain crops) areas. Three large dams are upstream from the monitoring site: the Rafferty Alameda Dams, built in the 1990s and the Boundary Dam, built in 1957 (The Encyclopedia of Saskatchewan, 2008). The presence of these dams indicates that the flow at this site is intensively managed. A list of publications relating to water quality, land use, and basin characteristics in the Souris River Basin was assembled by the U.S. Geological Survey (2006b).

Table 1. Description of sites used in trend analyses.

[USGS, U.S. Geological Survey; km², square kilometer; agr, agriculture; for, forest; mix, mixed urban, agriculture, and forest; sites in **bold** indicate long-term evaluation sites]

USGS site number	Site name	State	Map number (figure 1)	Drainage area (km²)	Major river basin	Land use / land cover			
						Percent urban	Percent agriculture	Percent forest	Dominant type
				Western part					
05056100	Mauvais near Cando	ND	1	977	Red River	0.59	56.52	2.46	Agr
05056200	Edmore Coulee near Edmore	ND	2	731	Red River	.34	92.72	1.23	Agr
05056239	Starkweather Coulee near Webster	ND	3	544	Red River	.24	84.33	.60	Agr
05058700	Sheyenne River at Lisbon	ND	4	11,370	Red River	.38	76.77	1.46	Agr
05064500	Red River of the North at Halstad	MN	5	46,620	Red River	.63	78.65	5.97	Agr
05082500	Red River of the North at Grand Forks	ND	6	68,117	Red River	.65	72.22	7.39	Agr
05114000	**Souris River near Sherwood**[1]	**ND**	**7**	**7,876**	**Red River**	**.02**	**7.45**	**.00**	**Agr**
05120000	Souris River near Verendrye[1]	ND	8	11,399	Red River	.26	17.54	.09	Agr
				Central part					
04063700	Popple River near Fence	WI	9	360	Great Lakes	0.03	3.36	57.45	For
04067500	Menominee River near McAllister	WI	10	10,181	Great Lakes	.41	6.22	64.07	For
04073468	Green Lake Inlet near Green Lake	WI	11	139	Great Lakes	3.88	78.96	6.54	Agr
04085427	Manitowoc River at Manitowoc	WI	12	1,363	Great Lakes	1.41	78.53	9.16	Agr
04087000	**Milwaukee River at Milwaukee**	**WI**	**13**	**1,803**	**Great Lakes**	**8.30**	**65.11**	**16.45**	**Agr**
05287890	Elm Creek near Champlin	MN	14	223	Upper Mississippi River	8.13	55.32	11.79	Agr
05340500	St. Croix River at St. Croix Falls	WI	15	16,166	Upper Mississippi River	.57	23.25	52.13	For
05369500	**Chippewa River at Durand**	**WI**	**16**	**23,342**	**Upper Mississippi River**	**.62**	**33.13**	**48.89**	**For**
05382000	Black River near Galesville	WI	17	5,389	Upper Mississippi River	.61	39.95	52.60	For
05389400	Bloody Run Creek near Marquette	IA	18	88	Upper Mississippi River	1.02	69.03	27.92	Agr
05389500	Mississippi River at McGregor	IA	19	174,870	Upper Mississippi River	1.74	53.57	26.94	Agr
05407000	Wisconsin River at Muscoda	WI	20	26,943	Upper Mississippi River	1.15	37.84	45.31	For
05420500	**Mississippi River at Clinton**	**IA**	**21**	**221,762**	**Upper Mississippi River**	**1.68**	**52.96**	**28.93**	**Agr**
05427718	Yahara River at Windsor	WI	22	191	Upper Mississippi River	2.37	91.71	3.63	Agr
05427948	Pheasant Branch at Middleton	WI	23	44	Upper Mississippi River	6.10	85.81	4.30	Agr
054310157	Jackson Creek Tributary near Elkhorn	WI	24	11	Upper Mississippi River	19.43	72.77	1.86	Mix
05465500	**Iowa River at Wapello**	**IA**	**25**	**32,383**	**Upper Mississippi River**	**3.12**	**83.63**	**4.80**	**Agr**
05474000	Skunk River at Augusta	IA	26	11,171	Upper Mississippi River	2.91	81.11	7.68	Agr
05481650	Des Moines River near Saylorville	IA	27	15,132	Upper Mississippi River	2.15	88.93	2.96	Agr
05500000	South Fabius River near Taylor	MO	28	1,606	Upper Mississippi River	1.45	67.92	19.96	Agr
05514500	Cuivre River near Troy	MO	29	2,339	Upper Mississippi River	.52	72.68	22.16	Agr
05525500	Sugar Creek at Milford	IL	30	1,155	Upper Mississippi River	.57	97.25	.88	Agr

Table 1. Description of sites used in the analyses.—Continued

[USGS, U.S. Geological Survey; km^2, square kilometer; agr, agriculture; for, forest; mix, mixed urban, agriculture, and forest; sites in **bold** indicate long-term evaluation sites]

USGS site number	Site name	State	Map number (figure 1)	Drainage area (km^2)	Major river basin	Land use / land cover: Percent urban	Percent agriculture	Percent forest	Dominant type
				Central part—continued					
05531500	Salt Creek at Western Springs	IL	31	298	Upper Mississippi River	62.87	2.20	8.01	Urb
05532500	Des Plaines River at Riverside	IL	32	1,632	Upper Mississippi River	38.85	31.98	10.72	Urb
05540275	Spring Brook at 87th St. near Naperville	IL	33	26	Upper Mississippi River	27.11	58.56	3.07	Mix
05543500	Illinois River at Marseilles	IL	34	21,396	Upper Mississippi River	11.97	71.74	7.80	Mix
05552500	Fox River at Dayton	IL	35	6,586	Upper Mississippi River	9.19	67.81	10.68	Agr
05570000	Spoon River at Seveille	IL	36	4,238	Upper Mississippi River	.82	84.50	11.77	Agr
05586100	Illinois River at Valley City	IL	37	69,285	Upper Mississippi River	5.68	78.49	9.44	Agr
07014500	Meramec River near Sullivan	MO	38	3,821	Upper Mississippi River	.54	20.75	77.12	For
07018100	Big River near Richwoods	MO	39	1,904	Upper Mississippi River	1.41	21.22	73.50	For
				Eastern part					
03085000	Monongahela River at Braddock	PA	40	19,008	Ohio River	2.60	20.18	72.91	For
03267900	Mad River at St Paris Pike at Eagle City	OH	41	803	Ohio River	2.40	80.20	14.50	Agr
03353637	Little Buck Creek near Indianapolis	IN	42	44	Ohio River	44.72	41.69	2.99	Urb
04186500	Auglaize River near Fort Jennings	OH	43	860	Great Lakes	2.05	88.53	7.98	Agr
04193500	**Maumee River at Waterville**	**OH**	**44**	**16,399**	**Great Lakes**	**2.61**	**87.09**	**7.36**	**Agr**
0422026250	Northrup Creek at North Greece	NY	45	26	Great Lakes	11.92	54.94	27.74	Mix
04232034	Irondequoit Creek at Railroad Mills near Fishers	NY	46	102	Great Lakes	2.82	58.34	34.10	Agr
0423204920	East Branch Allen Creek at Pittsford	NY	47	18	Great Lakes	18.58	53.32	17.52	Mix
0423205025	Irondequoit Creek at Empire Boulevard in Rochester	NY	48	371	Great Lakes	22.62	41.35	23.75	Mix
04237946	Onondaga Creek Tributary number 6 downstream of mudboil area at Tully	NY	49	1	Great Lakes	.17	33.96	65.87	For

[1]Statistics on land use/land cover represent U.S. portion only.

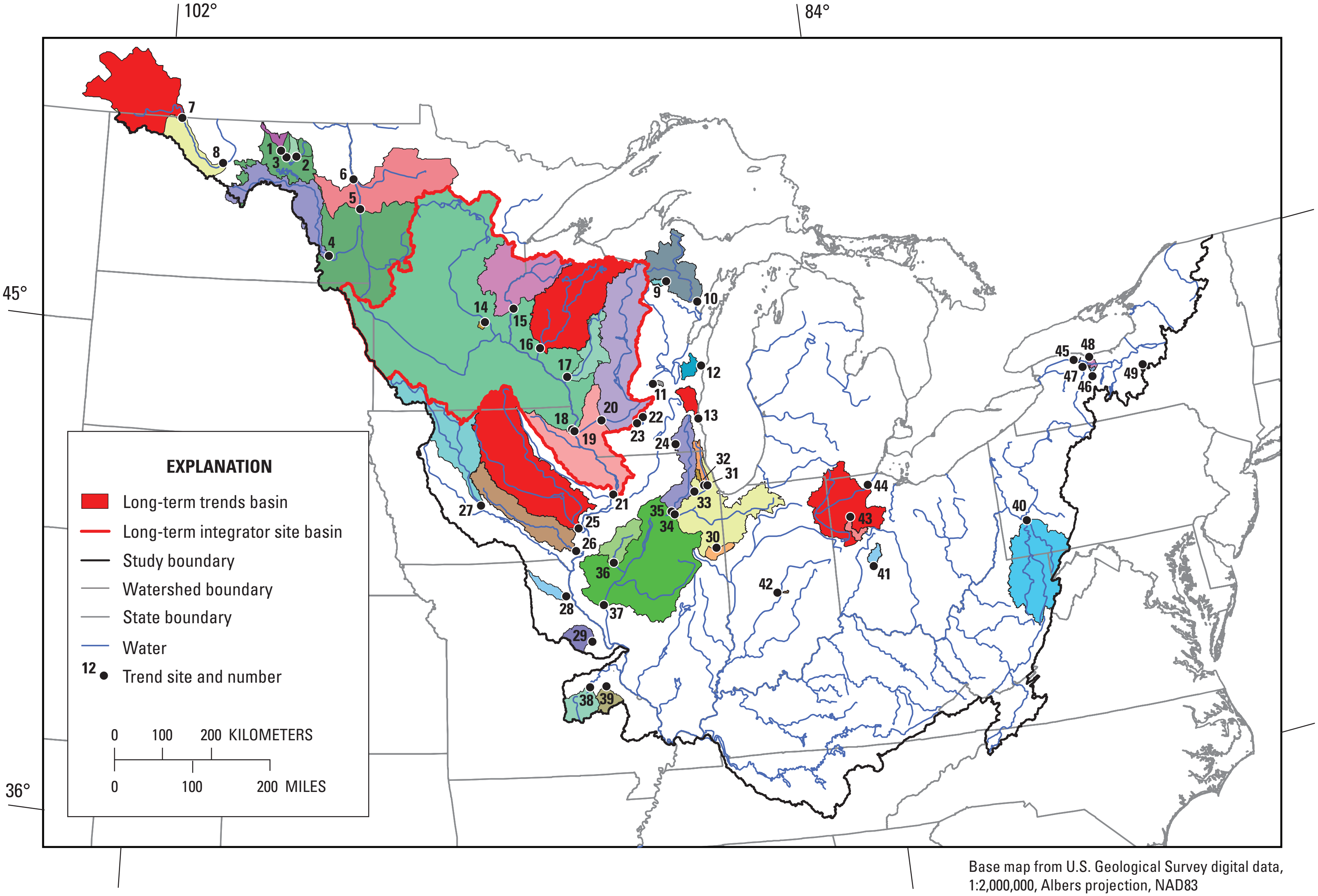


Figure 11. Sites used in the study. Drainage basins of the long-term trend sites are filled in red; the basin of the long-term integrator site on the Mississippi River is outlined in red. Sites are described in detail in table 1.

The four long-term central sites are the Chippewa River at Durand, Wis., the Iowa River at Wapello, Iowa, the Milwaukee River at Milwaukee, Wis., and the Mississippi River at Clinton, Iowa. The Chippewa River Basin is in northwestern Wisconsin and has an area of about 23,300 km^2. The upper part of this basin is largely undeveloped, contains several major parks and many State wildlife areas, and is one of the least-populated parts of the State. Major tributaries to the upper part of the Chippewa River include the Flambeau, Jump, Thornapple, Manitowish, Couderay, and Elk Rivers. The lower part of the basin has areas of agriculture and several moderate sized cities, such as Eau Claire. Major tributaries to the lower part of the Chippewa River include the Red Cedar and Eau Claire Rivers. The lower Chippewa River Basin contains many large areas of undeveloped native prairie and forest with numerous rare and endangered species of plants of animals (Wisconsin Department of Natural Resources, 2006). The site described in this report is located approximately 25 km upstream from the confluence of the Chippewa River and the Mississippi River. The characteristics of the downstream part of basin are described by Voss and Beaster (2001).

The Iowa River at Wapello, Iowa, is located near the confluence of the Iowa River and the Mississippi River. The drainage area at the monitoring site is approximately 32,400 km^2 and extends from east-central Iowa to the Mississippi River. The Cedar River is a major tributary. The population of the basin was approximately 772,000 with 22 percent agricultural land use in 1960 and is expected to reach a population of nearly 2 million with only 4 percent agricultural land use by 2020 as farmland and forest lands are converted to urban, industrial, and residential uses (U.S. Army Corps of Engineers, 2004). Flow at this site is influenced by management of the Coralville Dam and reservoir, substantial irrigation demands, periodic flooding and flood plain management, and construction of agricultural levees. The basin has numerous water-quality problems including habitat degradation throughout the basin and severe problems with sedimentation in the lower parts of this basin (U.S. Army Corps of Engineers, 2004).

The Milwaukee River Basin comprises six basins that collectively cover approximately 1,800 km^2. Grasslands, wetlands, forested lands, and agricultural land account for a large percentage of land in the upper Milwaukee River Basin; however, many of these areas are facing pressure from urban development as the greater Milwaukee metropolitan area continually expands to the northwest. The Milwaukee River and Estuary, downstream from the monitoring site, are listed by the USEPA as being an Area of Concern (U.S. Environmental Protection Agency, 2006b), where 11 water-quality impairments exist. Water-quality problems in the intensely urbanized lower Milwaukee River have caused beach closings, restrictions on fish consumption, restrictions on dredging activities, degradation of phytoplankton and zooplankton, restrictions on drinking-water supplies, extensive eutrophication, and disruption of animal reproductive cycles and health. Recently published reports documenting the water quality in the basin include Schneider and others (2004) and Hall (2006).

The Upper Mississippi River drains parts of Minnesota, Wisconsin, Iowa, and Illinois, and represents an integration of most of the central part of the study area. The Upper Mississippi River is one of the most regulated waterways in the world, with numerous dams, irrigation channels, locks, dredged waterways, and artificial channels. The site at Clinton, Iowa, has a drainage area of approximately 222,000 km^2. The Upper Mississippi River Basin includes all types of land use, including a diverse array of habitats for birds, terrestrial animals, and aquatic species. A summary of water quality in the Upper Mississippi River was published by Sullivan and others (2002).

The long-term site in the eastern part of the study area is in the Maumee River Basin. The river flows through parts of Indiana, Ohio, and Michigan and has the largest drainage area of any Great Lakes river, with approximately 16,400 km^2. Approximately 90 percent of the basin is agricultural, with the remaining area consisting of forest, urban, and industrial land use. The Maumee River Basin is the largest single source of sediment to Lake Erie and is one of the USEPA's "Areas of Concern" because of water-quality problems attributed to agricultural nonpoint-source pollution, contaminated industrial sites, combined sewer overflows, periodic dredging of the shipping channel sediments, and disposal of dredged materials (U.S. Environmental Protection Agency, 2006c).

Data Manipulations

Beginning in 1999, the USGS National Water Quality Laboratory (NWQL) began reporting the quantitation limit (QL) in addition to the method detection limit (MDL) for reporting censored data. The QL is larger than the MDL to reduce the chance of the actual concentration being greater than the MDL and reporting the data as censored at the MDL. Helsel (2005) reported biased results when using censored values at the QL instead of the MDL and offered three methods for modifying the data to avoid biased results. Method 1, which used the MDL instead of the QL, was used for all trend analyses. Table 2 summarizes the adjustments to censored data retrieved from the NWQL. All of the censored data used to describe the general spatial patterns in concentrations and yields were set to one-half of the detection limit.

Other adjustments to the reported concentrations also were made to data retrieved from the NWQL (David K. Mueller, U.S. Geological Survey, electronic commun., November 9, 2004). For dissolved ammonia, all concentrations reported with a value less than 0.02 mg/L prior to September 30, 1997, were set to less than 0.02 mg/L. For total and dissolved phosphorus, all concentrations reported as a value less than 0.03 mg/L prior to September 30, 1998, were set to less than 0.05 mg/L. Total nitrogen concentrations were not always reported in the databases, but individual nitrogen constituents were reported. Therefore, if dissolved nitrite plus nitrate and total Kjeldahl nitrogen concentrations were reported, total nitrogen concentrations were computed from the sum of these two constituents.

Table 2. Adjustments applied to censored data values retrieved from the National Water Quality Laboratory.

[All values in milligrams per liter; <, less than]

Constituent	Time frame	Description of adjustment
Total nitrogen		No adjustments
Dissolved ammonia	Prior to Sept. 30, 1997	Multiple censored values less than 0.02 were set to <0.02
	After Sept. 31, 2000	Censored values (<0.041) were set to <0.021
Kjeldahl nitrogen	Sept. 30, 1998, to Sept. 30, 2000, and after Sept. 30, 2001	Censored values (<0.1) were set to <0.05
	Sept. 31, 2000, to Sept. 30, 2001	Censored values (<0.05) were set to <0.04
Dissolved nitrite plus nitrate	Sept. 30, 2000, to Sept. 30, 2002	Censored values (<0.05) were set to <0.023
	After Sept. 31, 2002	Censored values (<0.05 and <0.06) were set to <0.03
Total phosphorus	Prior to Sept. 30, 1998	Values less than 0.03 were set to <0.05
	Sept. 30, 1999, to Sept. 30, 2000	Censored values (<0.008) set to <0.004
	After Sept. 30, 2000	Censored values (<0.0037) set to <0.0019
Dissolved phosphorus	Prior to Sept. 30, 1998	Values less than 0.03 were set to <0.05
	Sept. 30, 1999, to Sept. 30, 2000	Censored values (<0.008) set to <0.004
	After Sept. 30, 2000	Censored values (<0.0037) set to <0.0019

Basin Boundaries and Land-Use Classification

Basin boundaries for all of the streams selected in the first level of screening to describe general spatial patterns were delineated and digitized from 1:24,000-scale USGS topographic quadrangle maps or 1:100,000-scale digital coverage of the USGS Hydrologic Unit maps (Seaber and others, 1984) refined with digital RF3 stream coverages (Clifford and others, 1993).

Each of the basins of the final 49 sites was classified into one of four land-use categories: urban, agriculture, forest, or mixed on the basis of the dominant land use from the NLCD land use and land cover data in figure 5 (table 1). If urban land use exceeded 10 percent of the basin, but was not the dominant land use, then the class “mixed” was used for that basin.

Trend and Load Computations

A generalized least-squares regression (GLS) method was used to model trends in streamflow. The model used all daily streamflow measurements available over the analysis period. It relates the log of daily streamflow to an intercept, a linear trend term that is measured by decimal time, and sine and cosine seasonal factors (also functions of decimal time). The GLS method accounts for long-term serial correlation of the streamflow data. The results are expressed as percent change per year, and the reference flow is the mean flow for the analysis period.

The regression method, which also is referred to as the rating-curve method, was used to quantify loads and evaluate trends in water quality. This method commonly is used for load estimation when infrequent water-quality data are available over long periods of time (Cohn, 2005). Daily loads (L) were computed on the basis of relations between constituent load and two explanatory variables—streamflow and time. The regression models use the log-transform of load and streamflow to improve the linear relation and account for multiplicative errors. Nine different predefined regression models were used to estimate loads (table 3). Streamflow and time are transformed to produce variables that are statistically independent when squared terms are included, a process called centering. Because a log transformation was used in the models, computed daily loads were adjusted to account for a retransformation bias by use of the minimum variance unbiased estimate procedure (Cohn and others, 1989) or the adjusted maximum likelihood method (Cohn, 2005).

Table 3. Nine predefined regression models for estimating loads.

[L, natural logarithm (log) of the estimated load; Q^*, centered log of streamflow; T^*, centered time (in decimal years); T, fraction of the year (expressed in decimal years); β_i, estimated coefficients]

Model number	Model description
1	$L = \beta_0 + \beta_1 Q^*$
2	$L = \beta_0 + \beta_1 Q^* + \beta_2 Q^{*2}$
3	$L = \beta_0 + \beta_1 Q^* + \beta_2 T^*$
4	$L = \beta_0 + \beta_1 Q^* + \beta_2 \sin(2\pi T) + \beta_3 \cos(2T)$
5	$L = \beta_0 + \beta_1 Q^* + \beta_2 Q^{*2} + \beta_3 T^*$
6	$L = \beta_0 + \beta_1 Q^* + \beta_2 Q^{*2} + \beta_3 \sin(2\pi T) + \beta_4 \cos(2\pi T)$
7	$L = \beta_0 + \beta_1 Q^* + \beta_2 T^* + \beta_3 \sin(2\pi T) + \beta_4 \cos(2\pi T)$
8	$L = \beta_0 + \beta_1 Q^* + \beta_2 Q^{*2} + \beta_3 T^* + \beta_4 \sin(2\pi T) + \beta_5 \cos(2\pi T)$
9	$L = \beta_0 + \beta_1 Q^* + \beta_2 Q^{*2} + \beta_3 T^* + \beta_4 T^{*2} + \beta_5 \sin(2\pi T) + \beta_6 \cos(2\pi T)$

Computation of Loads for General Spatial Patterns

To describe the general patterns in yields or loads per unit area, annual loads, which are calculated by summing daily loads, were estimated by use of the Fluxmaster program (Schwarz and others, 2006). To simplify processing for this part of the analysis, model 7 (table 3) was used to compute loads for all constituents at all sites rather than calibrating a model for each basin. All of the water-quality data from 1970 to 2004 with available corresponding daily streamflow values were used in the analysis to estimate the regression coefficients. Daily loads were estimated for each site from 1970 to 2004. Total annual loads for general spatial patterns were computed for all water years, defined as the period from October 1 of the previous calendar year through September 30 of the current year, that had no missing daily streamflow values. Median annual loads were then computed for each site that had at least 5 years of data. Yields were then computed by dividing the median annual load by the drainage area of the basin.

The general spatial patterns of concentrations are expressed as median midmonthly concentrations, which are intended to reduce the variability of the random sampling of the constituent. The number of samples collected at each site was quite variable and ranged from quarterly to over a hundred in a year. Therefore, to obtain representative statistical summaries that were not biased by intensively sampled periods, the data were subsampled to monthly intervals. For each site, only one sample per constituent per month per year was used. The value included in the statistical summaries was the one collected closest to the middle of the month (midmonthly value). The midmonthly median is the median of all midmonthly values at a site.

Computations of Trends and Detailed Loads for 1993 to 2004

A version of LOADEST (Runkel and others, 2004) converted to run in S-PLUS (Insightful, 2005) was used to compute FA trends and loads. For each constituent at each site, the best model (table 3) identified by the Akaike information criterion (AIC) (Akaike, 1974) was selected and validated using all of the data from water year 1993 through water year 2004 (October 1, 1992, through September 30, 2004). The validation included checking the residuals for a linear fit, uniform scatter around the fit, normality of distribution, and linearity with all explanatory variables. If the model failed validation, then regression models were examined individually until a valid model was identified, or if no valid model was found, the analysis was dropped. For the analysis of trends, a time term was added to the model if it was not in the selected model.

The FA trend in concentrations at a site can be estimated from the coefficients in the equation chosen in the regression method. Consider the seven-parameter model shown by model 9 in table 3. With adjustment to the estimated parameters, this could be a concentration model because $L = C + Q + F$, where C is the log concentration, Q is the log streamflow, and F is a units conversion factor. Only β_0 and β_1 need adjustment, as follows:

$$\beta'_0 = \beta_0 - F - Q^a, \tag{1}$$

and

$$\beta'_1 = \beta_1 - 1, \tag{2}$$

where

Q^a is the center value for streamflow and

β_2 in model 9 is not changed because Q is orthogonal (uncorrelated) to Q^{*2}.

No other terms need adjustment because there is no streamflow component in them. The linear time terms (β_3 and β_4) in model 9 describe the FA trend. This assumes that the “true” model for FA is quadratic in log streamflow and first order Fourier seasonal component.

The FA trend in concentrations expressed as an average percentage change per year is computed from the coefficients from the linear time terms, as follows:

$$\%\triangle FAC/yr = 100\left(e^{(2\beta_4(T^m - T^a)+\beta_3)} - 1\right), \tag{3}$$

where

$\%\triangle FAC/yr$ is the percent change in FA trend per year,

e is the base of the natural logarithms,

β_4 is the coefficient for the second-order linear time term in model 9 and zero otherwise,

T^m is the midpoint of the period of record,

T^a is the centering value of time, and

β_3 is the coefficient for linear time.

OA trends in concentrations and trends in loads are defined as the changes in model-estimated concentrations and loads over the period of the water-quality record, divided by the length of that record. The model-estimated concentrations and loads are determined by fitting separate trend models for streamflow and for the water-quality constituent. The streamflow model used the GLS method. The water-quality model, estimated from all water-quality measurements collected during the analysis period, is one of the models in table 3 that include a linear decimal time term; these model numbers are 3, 5, 7, 8, or 9. The OA trend in the log of water-quality concentration is determined by streamflow and time trend components of the water-quality model, where the effect of the trend of streamflow is based on the coefficient for streamflow in the water-quality model. The OA trends in concentrations and loads are obtained by transforming the model-estimated, water-quality trend from log space to real space, computing the percentage of change corresponding to the first and last dates of the water-quality period of record, and normalizing by the length of this period. The significance of the trend was estimated by the p-value of a t-test using standard errors of estimated typical values. A more complete description of the methods used to compute trends is given in Sprague and others (2006), where the OA trend is referred to as nonflow-adjusted trend.

The results for nutrients and suspended materials are reported in estimates of unit trends—trends expressed in the units of concentration or load; for example, milligrams per liter per year (mg/L/yr) or kilograms per year (kg/yr). The unit trends in concentrations and loads are determined by multiplying the estimated trends in concentrations and loads by the appropriate reference value for concentration or load. For example, if the estimated trend was 10 percent per year and the reference concentration was 3 mg/L, then the unit trend would be 0.10/yr * 3 mg/L = 0.3 mg/L/yr. The reference concentration is obtained by evaluating the water-quality model at reference conditions consistent with the trend in water quality at the beginning of the water-quality period of record. These conditions include setting the log of streamflow to its smoothed trend value corresponding to the first day of the water-quality period, setting the trend term to the decimal equivalent of the first day of the water-quality period, and setting the sine and cosine seasonal factors to their average values over the full water-quality period. The log value of the reference concentration is transformed to real space and a multiplicative retransformation factor is applied to correct for statistical bias arising from sample error in the water-quality model coefficients. The reference load is computed similarly, except the logarithm of streamflow trend, as determined by the linear streamflow equation evaluated at the starting date of the water-quality period, is added to the log value of the reference concentration prior to transformation to real space; in addition, a multiplicative constant is applied to convert the result to appropriate load units. The same reference concentration is used to derive the FA and OA trends in concentrations; the same reference load is used in the evaluation of trends in loads.

The FA trends in concentrations, being independent of streamflow conditions, are best used to evaluate changes in water quality arising from changes in pollution sources or management activities within a basin. Conversely, the OA trend is indicative of the water-quality changes that riverine habitats have actually experienced. If there is no trend in streamflow over time, then the two estimates of trend are equivalent. Because the water-quality model used to derive these trends includes streamflow as a predictor, the estimates of trend are immune to potential biases arising from preferential water-quality sampling during high-streamflow events. Care should be taken, however, in interpolating or extrapolating these trend estimates within or beyond a site's period of record or in making comparisons of trend across sites that have different periods of record. Due to the possible nonlinearity of trends, such as arising from nonlinear specifications of the water-quality model's streamflow or trend components, trends within the water-quality period, or trends experienced outside this period, could be quite different from the trends reported for this study.

The attained p-values, the probability of obtaining a statistic as likely as the computed statistic when the null hypothesis is true, associated with each observation are based on a two-sided test, which means that either an upward or a downward trend could be identified (Helsel and Hirsch, 2002). The null hypothesis for a two-tailed trend test is that the trend is zero. The alternative hypothesis is that the trend is not zero. For a particular criterion for rejecting the null hypothesis, the critical region of the test statistic is in the tails of the distribution. For example, if the criterion is set to 0.05, a value in the inner 95 percent of the test statistic would not result in rejecting the null hypothesis, but a value in either the upper or lower 2.5 percent would result in rejecting the null hypothesis and accepting the alternative hypothesis that there is a trend in the data. To characterize regional patterns in trends, a 0.1 criterion was used to reject the null hypothesis.

When making multiple tests of trends, the effects of those multiple tests on accepting or rejecting the null hypothesis must be considered. For example, use of a criterion of 0.05 for rejecting the null hypothesis would result in an incorrect conclusion in 5 percent of the tests. For a single test, that could be an acceptable risk. For the 49 sites that were analyzed in this study, it could be expected at two or three sites where a trend was identified (the null hypothesis of no trend is rejected) that no real trend would exist. The reverse case also is possible: for several sites, there could be a real trend but the null hypothesis was not rejected because the trend was not large enough to detect given the noise of the observed data. The probabilities for the reverse case were not determined for this study.

Computations of Trends and Detailed Loads for 1975 to 2004

To place trends found for 1993–2004 into a longer term context, all of the sites with sufficient data, defined as sites with a minimum of approximately 200 observations from 1975 to 2004 and that had data that covered most of the extended period with some gaps, were analyzed for long-term changes in flow-adjusted concentrations (FAC). In addition, data from the six long-term site subset described in detail previously in the section "Final Site and Sample Selection" were analyzed for long-term changes in loads from 1975 to 2004. Long-term changes in FAC and loads were assessed using the QWTREND program (Vecchia, 2003a, 2005) and the LOADEST program (Runkel and others, 2004). QWTREND uses a time-series model for estimating trends in FAC. The basic form of the model is

$$FAC = Intercept + Annual + Seasonal + Trend + HFV \qquad (4)$$

where

FAC is the log of the flow-adjusted concentration;
Intercept is the intercept term;
Annual is the annual streamflow-related anomaly composed of 5-year and 1-year moving average streamflow anomalies;

Seasonal is the seasonal streamflow-related anomalies, composed of a 3-month moving average streamflow anomaly and second-order Fourier terms that describe seasonal variation;

Trend is the trend component that can include trend terms based on time or ancillary variables that explain the trend; and

HFV is the high-frequency variability component that includes the high-frequency component of streamflow, which is the daily fluctuations after the longer term anomalies have been removed, and periodic autoregressive coefficients to remove the non-random fluctuation in the observed concentration data.

Vecchia (2000) describes the estimation of the time-series parameters, and Vecchia (2003b) describes the computation of the anomalies.

The suggested minimum data requirements for QWTREND (Vecchia, 2000) are (1) minimum water-quality record length of 15 years, (2) average of at least four samples per year, (3) at least 10 samples within each quarter of the year, (4) less than 10 percent censored data, and (5) complete streamflow record for the water-quality record for the period of interest plus the preceding 5 years. These minimum data requirements do not guarantee convergence of the program, therefore, an average of at least 6.5 samples per year were required for this analysis. The long-term analysis covered 30 years; therefore, about 200 samples were required for each site. For some sites, slightly less than 200 samples were used because not all constituents were quantified on any given sampling day.

QWTREND was used to determine when changes in the trend during the 1975–2003 period were statistically significant. Once these changes were identified, the nine different predefined regression models (table 3) were examined for each trend period and the best load model identified by AIC was computed using LOADEST. The time variable in the model was then used to identify changes in the FAC. Total annual loads, computed by summing daily loads, were then computed for 1975–2003 by use of the various models specified for FAC and the use of LOADEST. Annual loads were calculated for each calendar year. Because streamflow data were not assembled for the period from October through December 2004, the annual loads for 2004 were not estimated. The LOESS smoothing procedure in S-PLUS (Insightful, 2005) was then used to identify the time-weighted central tendency, or moving average, of the total annual loads.

It should be noted that the methodology used to evaluate long-term trends is insensitive to changes in the variability of streamflow or to changes in the unexplained variability of water quality. Both of these changes potentially result in trends in water quality arising from nonlinearity in the specification of the water-quality model. Accommodation of the effects of these uncertainties awaits future research.

Explanation of Observed Trends

Implications from the analysis of the influence of human activity on trends in constituents were generally based on broad regional patterns rather than the results of specific analyses. These general tendencies were not analyzed on a statistical basis—no test for regional trend was made.

In addition to comparing patterns in the changes in concentrations and loads with the changes in the inputs, a regression approach was used to relate likely sources of nutrients to the observed trends. Changes in fertilizer inputs (fig. 7) were used as an ancillary variable in the trend analysis of the agricultural sites, and annual inputs from fertilizers and atmospheric deposition (figs. 7 and 9) were used as ancillary variables in the analysis of trends of the forested sites. For these ancillary variables, the data were log-transformed and included in the regression models as a substitute for the time term. The model used for each site was similar to the water-quality model and is shown in tables presented for each constituent in the section "Trends in Nutrient and Suspended-Sediment Concentrations and Loads." If the p-value of the ancillary variable was less than 0.10, then it was deemed significant and a plausible candidate to explain the trend in the context of human activity. Because manure input data were available only at 5-year intervals, those data could not be used in the regression model approach. To assess the change in concentration with the change in manure input for agricultural sites, the changes in manure input were compared to the changes in concentration at each site, and the significance of the relation was assessed using the p-value from the Spearman rank correlation test (Helsel and Hirsch, 2002).

Trends in Streamflow

Daily streamflows were obtained for the USGS gaging stations at the 49 sites and were analyzed for upward or downward monotonic trends from 1993 to 2004. The p-values in table 4 indicate the significance of the trend term and are used in figure 12 to demonstrate the significance of the trend ($p < 0.05$, $0.05 \leq p < 0.10$, and $p \geq 0.10$). The trends in streamflow throughout the study area showed a general pattern: small changes in streamflow in the western part, significant decreases in streamflow throughout most of the central part, and either no change or slight increases in streamflow in the eastern part of the study area (fig. 12 and summarized in table 4). In the western part of the study area, only one of the eight stations, the most western site, had a significant trend. In the central part, 18 of 31 stations had significant decreases in streamflow. Two of the 10 stations in the eastern part of the study area had significant increases in streamflow. The changes in streamflow are consistent with the changes in precipitation that occurred throughout this area during this period (fig. 4).

Table 4. Summary of results for analyses of trends in streamflow at selected sites in the study area, 1993 to 2004.

[USGS, U.S. Geological Survey; ID, identification; m^3/s, cubic meter per second; ft^3/s, cubic foot per second; %/yr, percent per year; sites in **bold** indicate long-term evaluation sites]

USGS site number	State-ID number (figure 1)	Reference flow (m^3/s)	Reference flow (ft^3/s)	Trend (%/yr)	p-value
			Western part		
05056100	ND-1	0.30	10.6	-4.90	0.435
05056200	ND-2	.53	18.7	-6.87	.169
05056239	ND-3	.56	19.9	-6.50	.169
05058700	ND-4	7.22	255	-2.47	.387
05064500	MN-5	62.8	2,220	-1.70	.406
05082500	ND-6	120	4,250	-2.63	.173
05114000	**ND-7**	**.17**	**5.89**	**45.28**	**.085**
05120000	ND-8	1.17	41.4	5.35	.496
			Central part		
04063700	WI-9	2.64	93.3	-2.99	0.004
04067500	WI-10	75.2	2,660	-.03	.977
04073468	WI-11	.85	29.9	-2.29	.321
04085427	WI-12	4.19	148	-2.71	.256
04087000	**WI-13**	**10.2**	**362**	**-1.44**	**.386**
05287890	MN-14	.71	25.2	-5.44	.028
05340500	WI-15	116	4,080	-1.09	.282
05369500	**WI-16**	**206**	**7,270**	**-1.25**	**.285**
05382000	WI-17	45.9	1,620	-3.41	.002
05389400	IA-18	.79	27.8	-4.36	.000
05389500	IA-19	1,320	46,600	-2.97	.001
05407000	WI-20	264	9,330	-2.26	.006
05420500	**IA-21**	**1,860**	**65,800**	**-3.22**	**.000**
05427718	WI-22	.73	25.7	-2.73	.001
05427948	WI-23	.09	3.18	-.51	.677
054310157	WI-24	.04	1.38	.26	.917
05465500	**IA-25**	**286**	**10,100**	**-4.84**	**.005**
05474000	IA-26	63.8	2,260	-5.89	.012
05481650	IA-27	99.5	3,520	-6.63	.000
05500000	MO-28	4.05	143	-6.27	.020
05514500	MO-29	3.53	125	-.51	.924
05525500	IL-30	4.56	161	-2.94	.440
05531500	IL-31	3.84	136	-.48	.676
05532500	IL-32	16.3	576	-1.56	.261
05540275	IL-33	.18	6.40	-3.43	.119
05543500	IL-34	302	10,700	-3.84	.001
05552500	IL-35	60.0	2,120	-3.65	.008
05570000	IL-36	30.1	1,060	-5.68	.008
05586100	IL-37	785	27,800	-4.71	.001
07014500	MO-38	35.4	1,250	-4.89	.000
07018100	MO-39	14.7	521	-4.61	.002

Table 4. Summary of results for analyses of trends in streamflow at selected sites in the study area, 1993 to 2004.—Continued

[USGS, U.S. Geological Survey; ID, identification; km², square kilometer; m³/s, cubic meter per second; ft³/s, cubic foot per second; %/yr, percent per year; sites in **bold** indicate long-term evaluation sites]

USGS site number	State-ID number (figure 1)	Reference flow		Trend (%/yr)	p-value
		(m³/s)	(ft³/s)		
		Eastern part			
03085000	PA-40	210	7,410	2.48	0.275
03267900	OH-41	4.76	168	26.87	.000
03353637	IN-42	.27	9.42	-2.79	.364
04186500	OH-43	2.37	83.9	-.34	.928
04193500	**OH-44**	**59.7**	**2,110**	**.57**	**.847**
0422026250	NY-45	.20	7.00	5.90	.031
04232034	NY-46	.83	29.4	-.91	.464
0423204920	NY-47	.15	5.28	-1.99	.274
0423205025	NY-48	3.04	108	-2.04	.153
04237946	NY-49	.02	.69	.31	.841

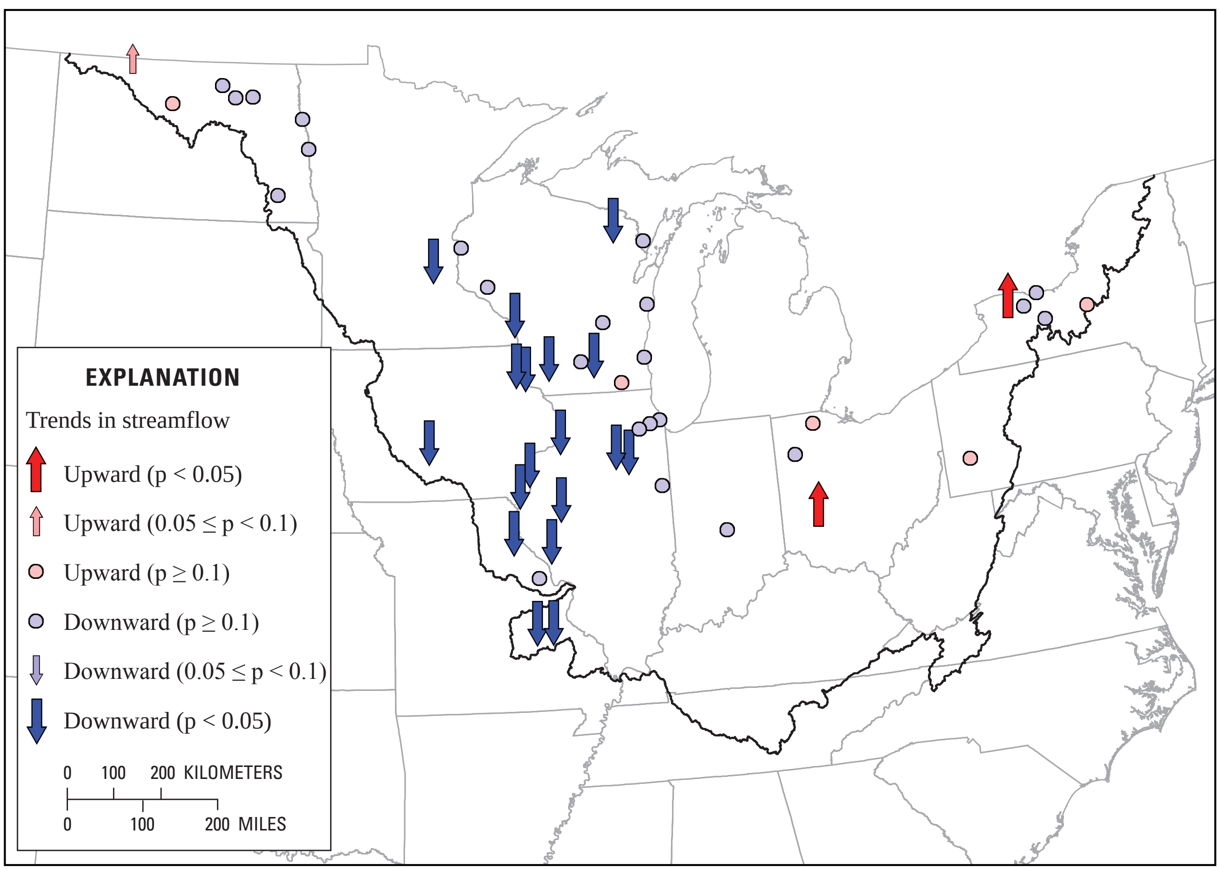


Figure 12. Trends in streamflow from 1993 to 2004 throughout the study area.

Long-term records in streamflow from 1975 to 2003 can be used to understand trends in streamflow from 1993 to 2004. The shorter term trends can be compared to the longer term trends and variability. Long-term trends in streamflow were examined at 6 of the 49 sites (fig. 13). For each site, mean annual streamflows were computed as the average of the mean daily streamflows, and then a LOESS smoothing procedure was used to indicate the long-term trends in mean annual streamflows.

Streamflow trends from 1993 to 2004 observed at each of the six long-term sites agree with the general trend for that period as shown in figure 13. For all sites except the Souris River, the range and variability of streamflow during the short-term period matched the range and variability of streamflow during the long-term period. The Souris River shows a period from 1984 to 1995 when the annual mean daily streamflow never exceeded 4 m^3/s. There could be some effect from that period on the trends estimated at that site.

Trends in Nutrient and Suspended-Sediment Concentrations and Loads

For each constituent, the trends in concentrations and loads are summarized in a table and shown spatially in a figure. The FA and OA trends in concentrations and trends in loads represent the monotonic trend from 1993 to 2004 and are expressed as a change in percentage per year and as a change in milligrams per liter per year or kilograms per year. The p-values in the tables indicate the significance of the trend term and are used in the figure to demonstrate the significance of the trend ($p < 0.05$, $0.05 \leq p < 0.10$, and $p \geq 0.10$). The reference concentration and load in the tables are typical values for that site in the beginning of the period and do not represent a summary statistic such as a mean or median value.

Total Nitrogen

General Patterns in Total Nitrogen Concentrations and Yields

From 1970 to 2004, median midmonthly total nitrogen (TN) concentrations across the study area ranged from 0.102 to 206 mg/L. The overall mean and median values of the median concentrations at these sites were 2.65 and 1.70 mg/L, respectively. The highest concentrations were in the western, southwestern, and south-central parts of the study area (fig. 14***A***). The lowest concentrations were mostly in northern Wisconsin and Michigan and in the northeastern and southeastern parts of the study area. The highest concentrations were in areas with intensive agriculture, and the lowest concentrations were in forested areas (fig. 5). Median midmonthly TN concentrations at the 24 trend sites, based on water-quality data from 1993 to 2004, ranged from 0.314 to 6.22 mg/L. The distribution in concentrations at these sites was consistent with that in figure 14***A***. The mean and median values of the median concentrations at the trend sites were 2.07 and 1.41 mg/L, respectively.

Median annual TN yields, expressed as loads per unit area, ranged from about 2.36 to 18,300 kg/km^2 based on water-quality and streamflow data from 1970 to 2004. The overall mean and median values of the median yields at these sites were 1,280 and 771 kg/km^2, respectively. The highest yields were throughout the middle of the study area (fig. 14***B***). The lowest yields were throughout the northern part of the study area. The highest yields were in areas with intense agriculture and moderate runoff. The areas with the highest runoff had relatively low TN concentrations, such as in northern Wisconsin and southeastern parts of the study area. The major difference in the distributions between concentrations and yields was in the northwestern part of the study area, which had high concentrations but low yields. Median annual TN yields at the trend sites ranged from 12 to 3,310 kg/km^2 (table 5), based on water-quality and streamflow data from 1993 to 2004. The mean and median values of the median yields from each site were 753 and 446 kg/km^2, respectively. The distribution in yields was consistent with yields shown in figure 14***B***.

Trends in Total Nitrogen Concentrations

From 1993 to 2004, the FA trends in TN concentrations at 19 of the 24 sites that had sufficient data for analysis show a tendency toward increasing concentrations (table 5; fig. 15***A***). The 19 sites with increasing TN concentrations were distributed throughout the entire study area; however, only 9 of these sites had significant upward trends ($p < 0.10$) and were all located in the central part of the study area. These upward FA trends indicate that the effects of anthropogenic activities are continuing to increase in these areas and add additional TN to the streams. Only two of the five sites that had downward trends in concentrations had significant trends; those sites were Spring Brook in northern Illinois in the central part, and Red River of the North in the western part of the study area.

When the OA trends in TN concentrations were examined for the period 1993–2004, 16 of the 24 sites showed a general tendency toward increasing concentrations (table 5; fig. 15***B***); however, only three of these sites had significant upward trends—two sites in southern Wisconsin in the central part of the study area and one site in North Dakota the western part of the study area. Of the eight sites with a downward trend, only two sites had significant trends—the same two sites that had downward FA trends. The sites with downward trends in concentrations were scattered throughout the study area, indicating no regional trend.

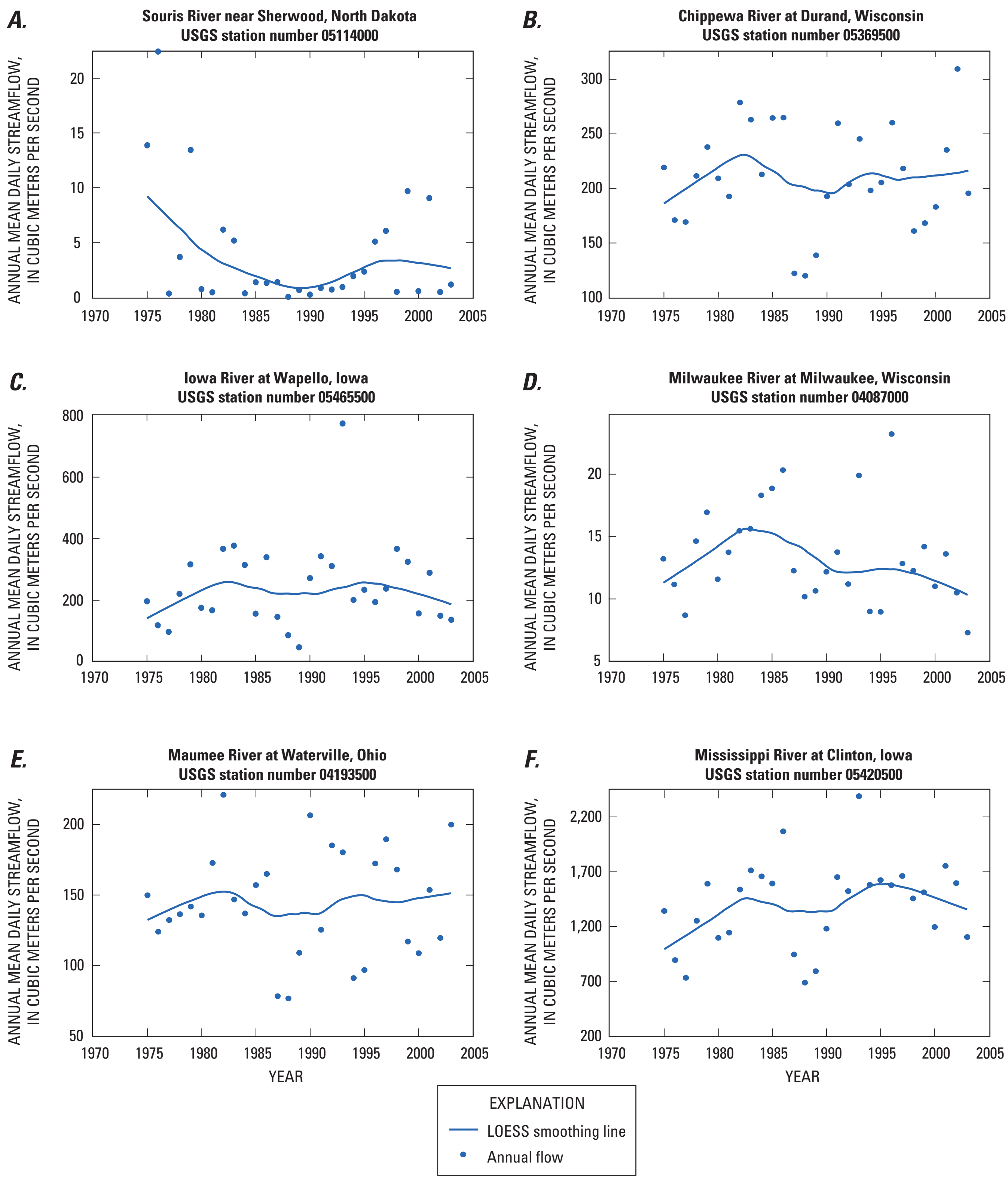
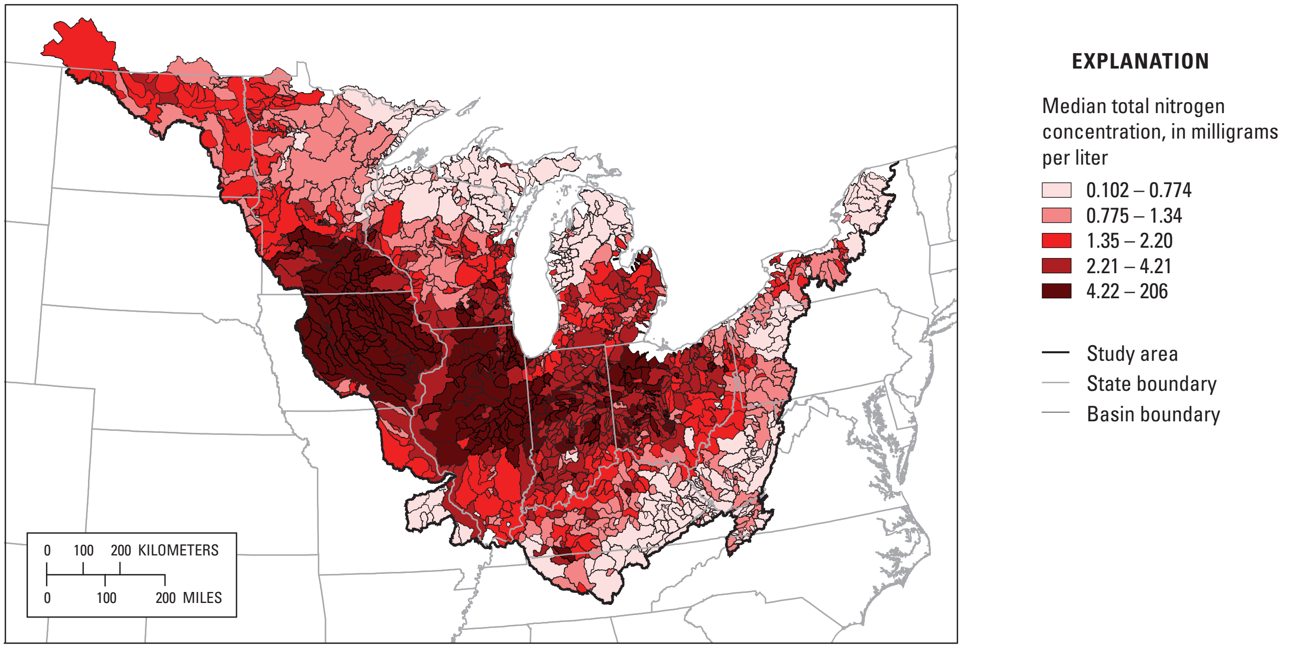


Figure 13. Long-term trends in streamflow from 1975 to 2003 at six selected sites in the study area.

A. **Median total nitrogen concentration**



B. **Median annual total nitrogen yield**

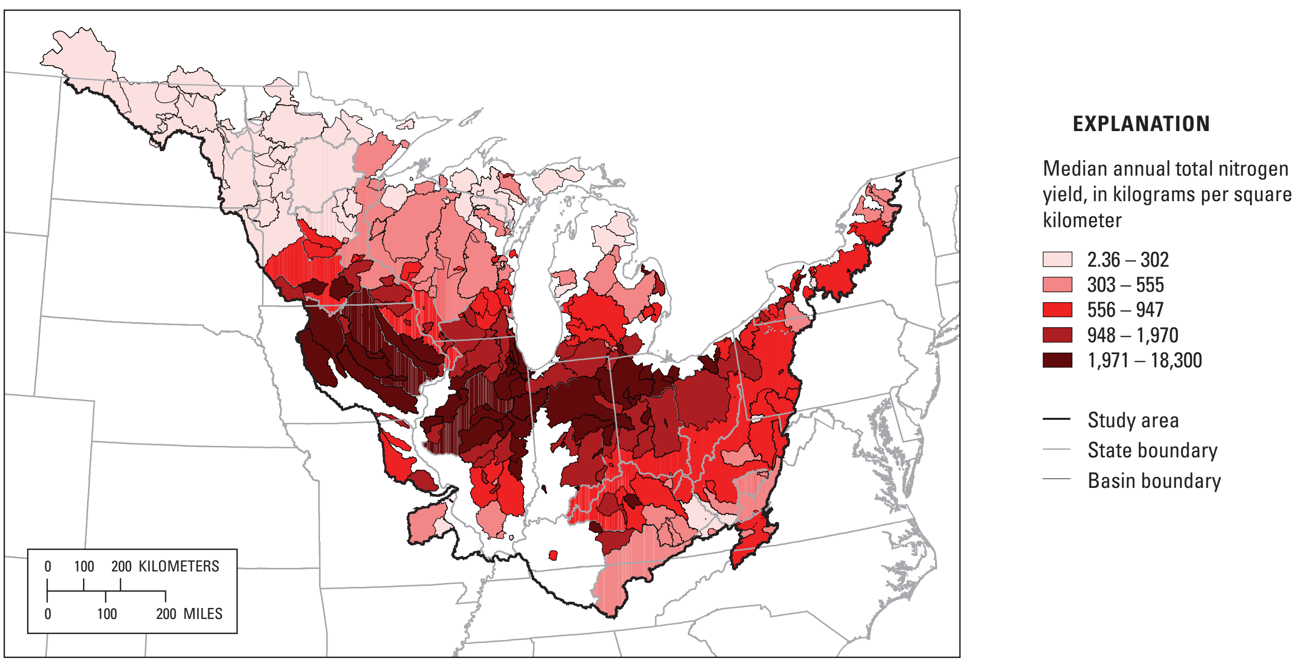


Figure 14. Spatial patterns from 1970 to 2004 in ***A***, median total nitrogen concentrations and ***B***, median annual total nitrogen yields throughout the study area.

Table 5. Summary of results for analyses of flow-adjusted trends, overall trends, and trends in loads for total nitrogen (TN, USGS parameter code 00600), 1993 to 2004.

[USGS, U.S. Geological Survey; ID, identification; mg/L, milligram per liter; mg/L/yr, milligram per liter per year; %/yr, percent per year; kg, kilogram; kg/km^2, kilogram per square kilometer; kg/yr, kilogram per year; NA, not available; NC, not calculated; sites in **bold** indicate long-term evaluation sites]

			Concentration							Annual loads					
				Flow-adjusted trend			Overall trend								
USGS site number	State-ID number (figure 1)	Water-quality model (table 3)	Reference concentration (mg/L)	Trend (mg/L/yr)	Trend (%/yr)	p-value	Trend (mg/L/yr)	Trend (%/yr)	p-value	Median annual load (kg)	Median yield (kg/km^2)	Reference load (kg)	Trend (kg/yr)	Trend (%/yr)	p-value
Western part															
05056100	ND-1	9	1.284	0.061	4.76	0.110	0.086	6.71	0.080	NA	NA	12,500	-250	-1.96	0.784
05056200	ND-2	7	1.727	.037	2.15	.325	.046	2.66	.223	NA	NA	28,800	-1,800	-6.35	.249
05056239	ND-3	7	1.749	.025	1.45	.637	.032	1.81	.570	NA	NA	31,800	-1,900	-6.08	.251
05058700	ND-4	3	1.233	.025	2.05	.499	.015	1.18	.689	747,000	116	281,000	-4,600	-1.65	.714
05082500	ND-6	7	1.513	-.062	-4.08	.075	-.067	-4.43	.038	8,460,000	860	5,940,000	NC	-5.77	.022
05114000	**ND-7**	**9**	2.012	**-.024**	**-1.20**	**.298**	**-.029**	**-1.42**	**.256**	**106,000**	**13.5**	**11,000**	**3,820**	**34.64**	**.119**
05120000	ND-8	7	1.474	.007	.45	.742	.011	.72	.623	136,000	12.0	55,600	3,590	6.45	.463
Central part															
04063700	WI-9	9	0.447	0.006	1.43	0.061	0.001	0.14	0.860	61,400	171	37,200	-1,100	-2.93	0.052
04067500	WI-10	3	.428	-.001	-.02	.991	-.001	-.03	.986	1,330,000	131	1,020,000	-670	-.07	.978
04087000	**WI-13**	**7**	1.735	**.045**	**2.61**	**.000**	**.043**	**2.48**	**.000**	**1,030,000**	**570**	**562,000**	**3,520**	**.63**	**.764**
05287890	MN-14	9	1.251	.024	1.96	.032	-.005	-.41	.743	72,100	323	25,700	-1,500	-5.80	.068
05340500	WI-15	9	.539	.001	.15	.943	-.001	-.28	.893	3,180,000	197	1,970,000	-27,000	-1.37	.566
05369500	**WI-16**	**7**	1.213	**.022**	**1.85**	**.003**	**.024**	**2.01**	**.002**	**10,000,000**	**428**	**7,900,000**	**36,500**	**.46**	**.724**
05382000	WI-17	7	1.243	.018	1.43	.061	.013	1.08	.138	2,400,000	446	1,800,000	-50,000	-2.79	.043
05407000	WI-20	7	1.354	.023	1.67	.104	.020	1.46	.111	12,300,000	456	11,300,000	-140,000	-1.20	.328
05420500	**IA-21**	**5**	2.096	**.054**	**2.56**	**.079**	**.010**	**.46**	**.730**	**131,000,000**	**591**	**118,000,000**	**-3,500,000**	**-2.98**	**.058**
05465500	**IA-25**	**8**	5.804	**.250**	**4.30**	**.008**	**.061**	**1.04**	**.519**	**63,300,000**	**1,950**	**52,700,000**	**-2,400,000**	**-4.60**	**.093**
05540275	IL-33	7	1.599	-.043	-2.72	.002	-.053	-3.32	.000	20,800	810	9,200	-500	-5.43	.016
05586100	IL-37	9	3.978	.149	3.75	.016	.060	1.51	.292	137,000,000	1,970	89,300,000	-3,800,000	-4.25	.051
07018100	MO-39	3	.307	.007	2.42	.609	-.003	-.91	.826	606,000	318	125,000	-6,600	-5.30	.147

Table 5. Summary of results for analyses of flow-adjusted trends, overall trends, and trends in loads for total nitrogen (TN, USGS parameter code 00600), 1993 to 2004.—Continued

[USGS, U.S. Geological Survey; ID, identification; mg/L, milligram per liter; mg/L/yr, milligram per liter per year; %/yr, percent per year; kg, kilogram; kg/km², kilogram per square kilometer; kg/yr, kilogram per year; NA, not available; NC, not calculated; sites in **bold** indicate long-term evaluation sites]

USGS site number	State-ID number (figure 1)	Water-quality model (table 3)	Concentration							Annual loads					
				Flow-adjusted trend			Overall trend								
			Reference concentration (mg/L)	Trend (mg/L/yr)	Trend (%/yr)	p-value	Trend (mg/L/yr)	Trend (%/yr)	p-value	Median annual load (kg)	Median yield (kg/km²)	Reference load (kg)	Trend (kg/yr)	Trend (%/yr)	p-value
Eastern part															
03085000	PA-40	5	0.869	0.028	3.26	0.118	0.025	2.86	0.163	13,000,000	682	5,810,000	356,000	6.13	0.058
03353637	IN-42	7	1.048	-.003	-.27	.749	-.010	-.97	.386	30,200	686	8,650	-310	-3.53	.342
04186500	OH-43	7	3.390	.080	2.37	.421	.077	2.26	.470	2,850,000	3,310	260,000	4,820	1.86	.754
04193500	**OH-44**	**9**	2.908	**.087**	**2.98**	**.126**	**.093**	**3.20**	**.145**	**38,400,000**	**2,340**	**5,300,000**	**210,000**	**3.96**	**.419**

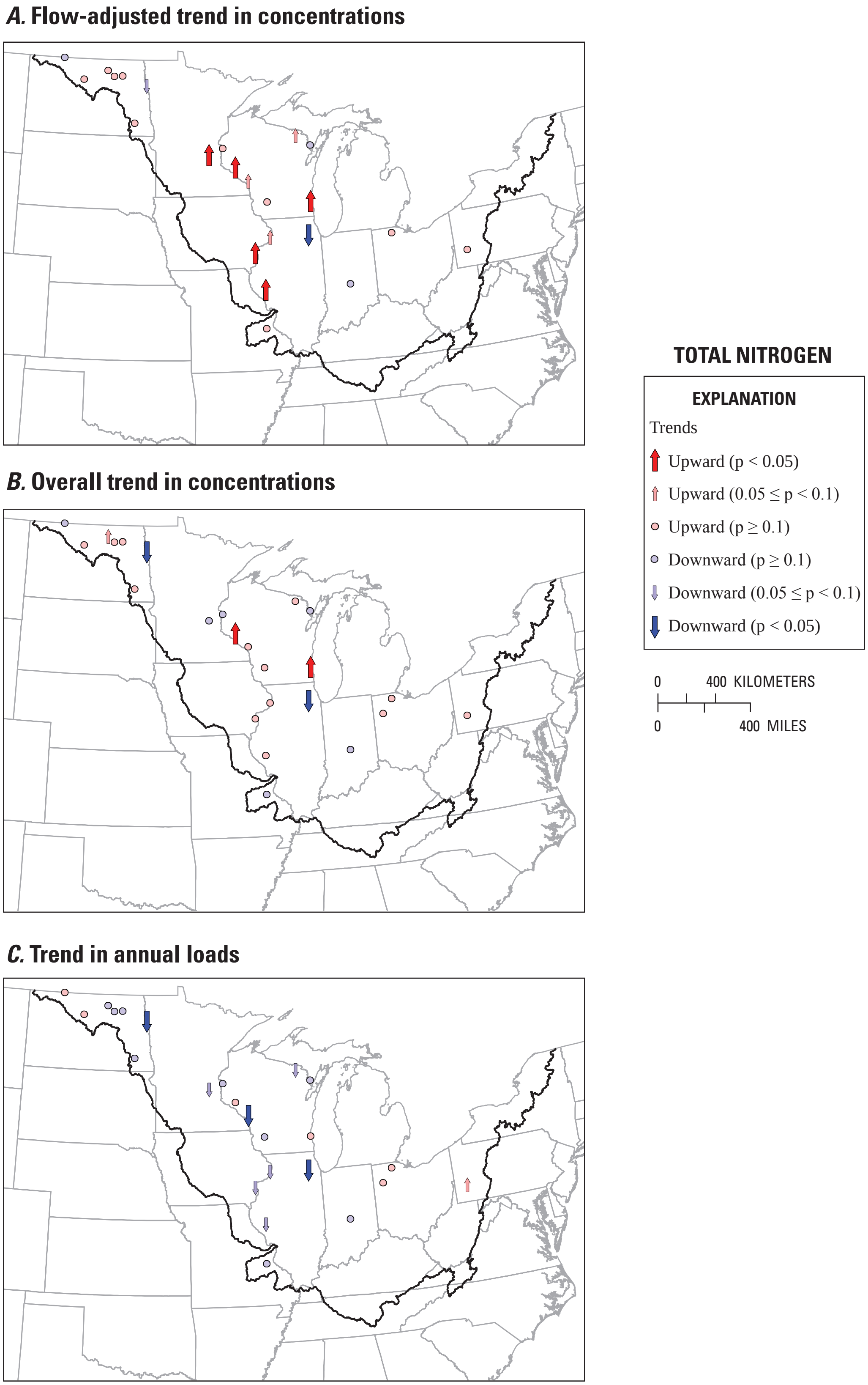


Figure 15. Trends in total nitrogen from 1993 to 2004 for ***A***, flow-adjusted concentrations, ***B***, overall concentrations, and ***C***, annual loads throughout the study area.

The differences between FA trends and OA trends in TN concentrations are attributed to the trends in streamflow (fig. 12). The strong downward trends in streamflow in the central part of the study area moderated the upward FA trends in that area. Of the eight sites that had significant upward FA trends in the central part of the study area, six sites had significant downward trends in streamflow, which generally resulted in insignificant OA trends or trends less than about 3 percent per year for those sites where the trends were significant. Sites in the eastern part of the study area did not have significant trends in streamflow and, therefore, the FA and OA trends were similar. The OA trends at sites in the western part of the study area also were not strongly affected by trends in streamflow, except for the Red River of the North (05082500), which had a downward trend in streamflow and a downward FA trend that resulted in a strong downward OA trend in concentrations, significant at $p < 0.05$.

To place the 1993 to 2004 trends in concentration into a longer term context, FA trends were computed using QWTREND from 1975 to 2004 for all of the sites with sufficient data (table 6). The FA trends in TN concentrations had small deviations from average conditions for the Chippewa River (05369500) and Iowa River (05465500) sites, both in the central part of the study area (fig. 16; table 6). The other four long-term sites had larger deviations from the long-term average flow-adjusted concentrations. The Souris River (05114000) site, in the western part, had a downward shift between 1984 and 1992, when no samples were collected. The Milwaukee River (04087000) had three distinct periods: an upward FA trend from 1975 to 1986, followed by a downward FA trend until 1991, and an upward FA trend from 1992 to 2004. The only long-term site in the eastern part, the Maumee River (04193500), had no FA trend until 1988, after which the flow-adjusted concentration increased for 2 years, which was followed by a downward FA trend to 2004. The Mississippi River integrator site (05420500) had a significant upward FA trend from 1975 to 1983, after which the flow-adjusted concentration was nearly constant. No consistent long-term FA trends in TN concentrations were found at these sites; changes generally alternated between upward and downward trends. Those fluctuating trends imply that local factors are more important that regional factors for long-term trends.

Trends in Total Nitrogen Loads

From 1993 to 2004, TN loads at 17 of the 24 sites that had sufficient data for analysis (table 5; fig. 15***C***) indicated a spatial tendency toward decreasing loads. Most of the sites with a downward trend were in the central or western parts of the study area. Eight of these sites had significant downward trends in loads ($p < 0.10$); most of these sites had significant downward trends in streamflow as well (fig. 12). The only exception was the site in eastern North Dakota, which had a significant downward trend in load and no trend in streamflow. Of the seven sites with upward trends in TN loads, only one site, the site in Pennsylvania, had a significant trend.

Table 6. Summary of flow-adjusted trends in total nitrogen concentrations (TN, USGS parameter code 00600) estimated by QWTREND for long-term evaluation sites, 1975 to 2004.

[USGS, U.S. Geological Survey; mg/L, milligram per liter]

USGS site number	Trend period	Description of flow-adjusted concentration
05114000	January 1975–August 1983	Constant value of 1.88 mg/L
	September 1983–February 1993	Gap in sampling
	March 1993–December 2004	Upward trend from 1.30 to 1.66 mg/L
05369500	January 1975–December 2004	Upward trend from 1.14 to 1.42 mg/L
05465500	January 1975–December 2004	Downward trend from 6.93 to 6.07 mg/L
04087000	January 1975–December 1986	Upward trend from 1.86 to 2.00 mg/L
	January 1987–December 1991	Downward trend from 2.00 to 1.74 mg/L
	January 1992–Dec 2004	Upward trend from 1.74 to 2.18 mg/L
04193500	January 1975–December 1987	Constant value of 5.25 mg/L
	January 1988–December 1989	Upward trend from 5.25 to 6.35 mg/L
	January 1990–December 2004	Downward trend from 6.35 to 4.02 mg/L
05420500	January 1975–December 1982	Upward trend from 1.60 to 2.38 mg/L
	January 1983–December 2004	Downward trend from 2.38 to 2.30 mg/L Gap in sampling from October 1986 to May 1991 Gap in sampling from August 1992 to October 1995

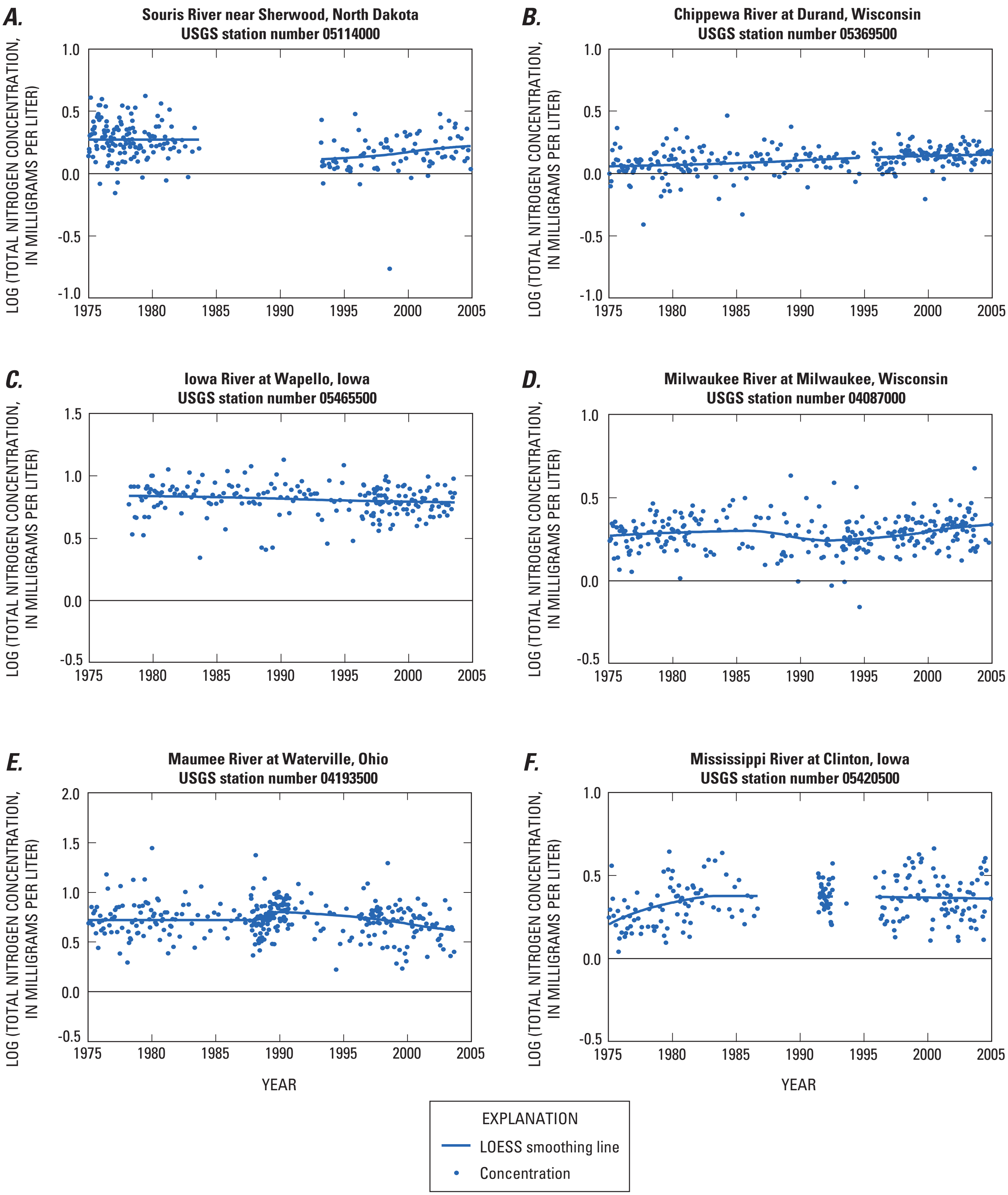


Figure 16. Long-term flow-adjusted trends in total nitrogen concentrations from 1975 to 2004 at six selected sites in the study area.

A comparison of the trends in loads and FA trends in concentrations indicates that the trends in loads are driven primarily by streamflow rather than changes in concentrations (fig. 15***C***). However, the effects of the trends in streamflow can augment the FA trends in concentration to cause a larger OA trend, such as the site in eastern North Dakota (fig. 15***B***).

Annual loads from 1975 to 2003 at the six long-term sites were used to place these short-term trends in load into a longer term context (fig. 17). The long-term changes at all six sites closely resembled the long-term trend in streamflow (fig. 13), which again indicates the importance of changes in streamflow in driving the trends in loads. This relation between streamflow and TN loads was consistent at all scales, from the relatively small Souris River, N. Dak., with a maximum mean annual streamflow of approximately 23.2 m^3/s, to the Mississippi River at Clinton, Iowa, with a maximum mean annual streamflow of 2,390 m^3/s.

Total nitrogen loads at the Souris River, N. Dak., decreased from the mid-1970s to approximately 1990, when loads increased slightly until 2003. That pattern in loads could have been affected by the construction of dams within the basin. Loads at the Milwaukee River, Wisconsin, generally increased from 1975 to 1985, then decreased until 1991, and then remained relatively constant through 2004. Loads at the Maumee River, Ohio, increased from 1975 to about 1982, then decreased slightly through 2003. Loads at the remaining four sites generally increased from 1975 to about 1982, then stayed relatively constant until about 1993, and then increased (Chippewa River) or decreased (Iowa and Mississippi Rivers) slightly until 2003.

Dissolved Ammonia

General Patterns in Dissolved Ammonia Concentrations and Yields

Median midmonthly dissolved ammonia (NH4) concentrations at 36 of the 49 trend sites that had sufficient data for analysis ranged from 0.010 to 0.150 mg/L, based on data from 1993 to 2004. The mean and median values of the median NH4 concentrations at these sites were 0.044 and 0.032 mg/L, respectively. The highest concentrations were measured at urban and agricultural sites in central Wisconsin and in central Illinois. Sites with high NH4 concentrations did not form regional patterns, but rather were spatially intermixed with sites with substantially lower concentrations. Although the lowest concentrations were irregularly distributed through the study area, the forested eastern sites in New York had the lowest concentrations.

Median annual NH4 yields at the trend sites ranged from 1.41 to 118 kg/km^2 (table 7), based on data from 1993 to 2004. The mean and median values from the median yields at these sites were 25.4 and 18.5 kg/km^2, respectively. The highest ammonia yields were at Salt Creek, Ill.; Des Plaines River, Ill.; and Northrup Creek, N.Y. The lowest yields were at the Souris River near Sherwood, N. Dak., the Souris River near Verendrye, N. Dak., and at Irondequoit Creek, N.Y.

Trends in Dissolved Ammonia Concentrations

The FA trends in NH4 concentrations from 1993 to 2004 showed a widespread spatial tendency toward decreasing concentrations (table 7; fig. 18***A***). Of the 36 sites with sufficient streamflow and NH4 concentration data for trend analysis, 32 sites had downward trends in concentrations. Of these 32 sites, 24 sites had significant downward trends ($p < 0.10$). Only four sites had upward FA trends, and only one site, Northrup Creek, N.Y. (0422026250), had a significant upward FA trend in concentrations; however, the magnitude of this upward trend was only 0.002 mg/L/yr.

When the OA trends in NH4 concentrations from 1993 to 2004 were examined, the results were similar to those of the FA trends. Concentrations at 32 of the 36 sites with sufficient data indicated a tendency toward decreasing concentrations (table 7; fig. 18***B***), and at 23 of these sites the downward trends were statistically significant ($p < 0.10$). Of the four sites with upward OA trends in concentrations, the only site with a significant trend was again Northrup Creek, N.Y.

Unlike trends for TN, almost no differences between FA and OA trends in NH4 concentrations were attributed to trends in streamflow. In general, streamflow was not important in determining the significance in the OA trend at most sites; however, the significance level between the FA and OA trends did change at three sites with downward trends in NH4 concentrations.

Trends in Dissolved Ammonia Loads

From 1993 to 2004, 32 of the 36 sites with sufficient data for analysis showed a tendency toward decreasing NH4 loads (table 7; fig. 18***C***). These sites with a downward trend were distributed across all three parts of study area. Twenty-six of these sites had significant downward trends in loads ($p < 0.10$). Most of these sites had significant downward FA trends in both concentration (table 7) and streamflow (table 4).

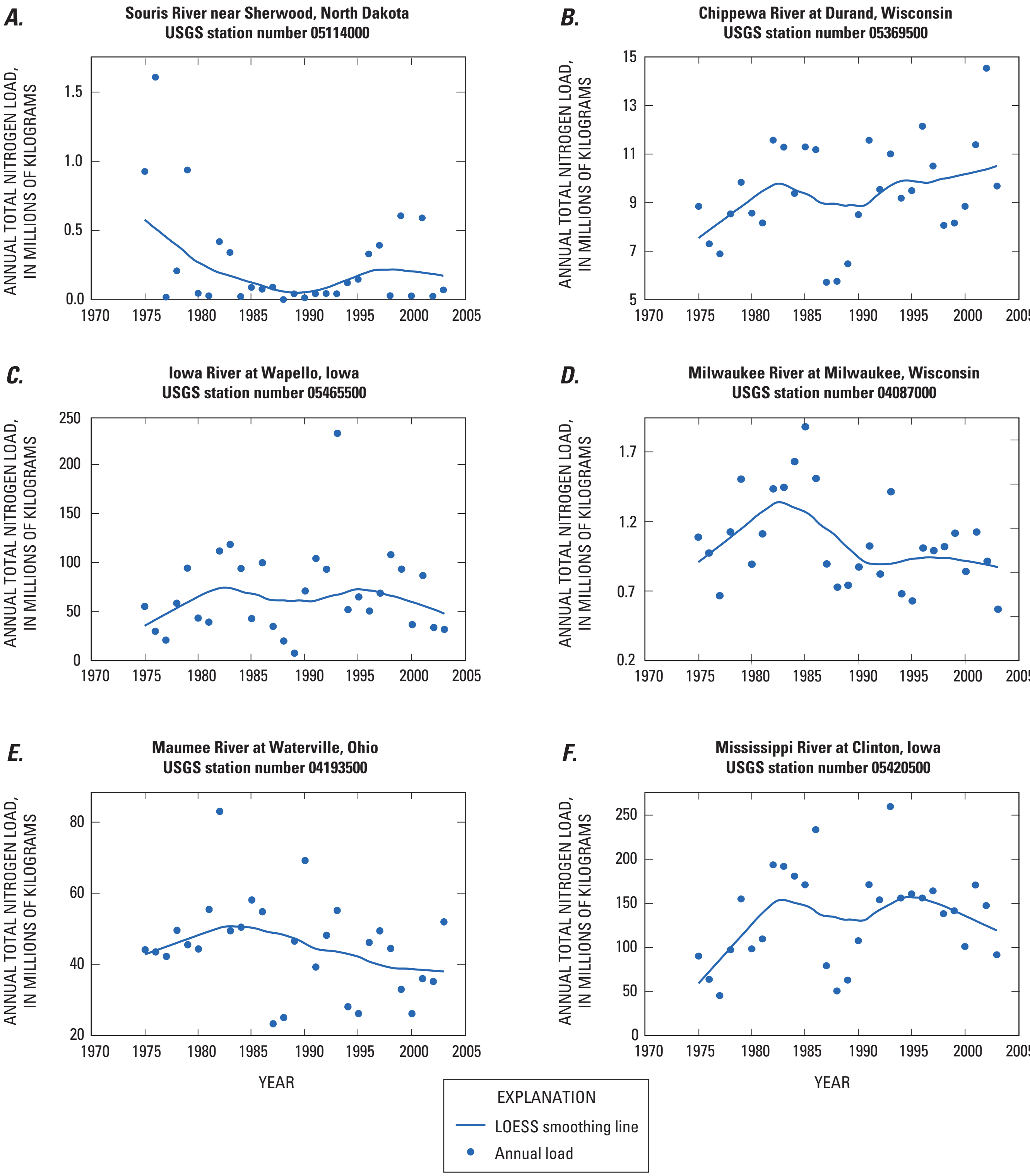


Figure 17. Long-term trends in total nitrogen loads from 1975 to 2003 at six selected sites in the study area.

Table 7. Summary of results for analyses of flow-adjusted trends, overall trends, and trends in loads for dissolved ammonia (NH4, USGS parameter code 00608), 1933 to 2004.

[USGS, U.S. Geological Survey; ID, identification; mg/L, milligram per liter; mg/L/yr, milligram per liter per year; %/yr, percent per year; kg, kilogram; kg/km², kilogram per square kilometer; NA, not available; NC, not calculated; sites in **bold** indicate long-term evaluation sites]

USGS site number	State-ID number (figure 1)	Water-quality model (table 3)	Concentration: Reference concentration (mg/L)	Concentration: Flow-adjusted trend: Trend (mg/L/yr)	Concentration: Flow-adjusted trend: Trend (%/yr)	Concentration: Flow-adjusted trend: p-value	Concentration: Overall trend: Trend (mg/L/yr)	Concentration: Overall trend: Trend (%/yr)	Concentration: Overall trend: p-value	Annual loads: Median annual load (kg)	Annual loads: Median yield (kg/km²)	Annual loads: Reference load (kg)	Annual loads: Trend (kg/yr)	Annual loads: Trend (%/yr)	Annual loads: p-value
							Western part								
05056100	ND-1	7	0.067	-0.004	-5.68	0.082	-0.004	-5.24	0.141	NA	NA	662	-48.0	-7.23	0.103
05056200	ND-2	7	.040	-.002	-4.48	.197	-.002	-4.84	.140	NA	NA	654	-52.0	-7.97	.072
05056239	ND-3	7	.044	-.003	-6.75	.019	-.003	-7.07	.009	NA	NA	780	-66.0	-8.47	.018
05058700	ND-4	7	.066	-.003	-3.92	.324	-.003	-4.47	.229	58,400	9.05	15,100	-860	-5.71	.179
05082500	ND-6	7	.092	-.008	-8.20	.000	-.008	-8.29	.000	301,000	30.6	362,000	-31,000	-8.47	.000
05114000	**ND-7**	**8**	**.093**	**-.004**	**-4.27**	**.197**	**-.006**	**-6.50**	**.053**	**11,100**	**1.41**	**587**	**25.8**	**4.40**	**.634**
05120000	ND-8	8	.090	-.004	-4.56	.166	-.005	-5.06	.124	16,200	1.42	3,550	-100	-2.87	.587
							Central part								
04063700	WI-9	9	0.038	-0.002	-4.66	0.000	-0.001	-3.68	0.003	2,660	7.38	3,160	-170	-5.42	0.000
04067500	WI-10	9	.036	-.001	-.43	.855	-.001	-.43	.856	58,700	5.76	85,100	-390	-.46	.856
04085427	WI-12	7	.071	-.004	-5.19	.098	-.003	-4.58	.144	16,900	12.4	9,440	-560	-5.89	.033
04087000	**WI-13**	**8**	**.037**	**-.001**	**-3.17**	**.051**	**-.001**	**-3.17**	**.046**	**30,000**	**16.6**	**11,900**	**-490**	**-4.08**	**.040**
05287890	MN-14	8	.183	-.012	-6.32	.000	-.012	-6.30	.000	4,350	19.5	3,980	-320	-7.96	.000
05340500	WI-15	7	.028	.001	.29	.931	.001	.20	.952	188,000	11.6	107,000	-990	-.93	.781
05369500	**WI-16**	**9**	**.052**	**.001**	**-.64**	**.746**	**-.001**	**-1.19**	**.543**	**507,000**	**21.7**	**338,000**	**-7,700**	**-2.27**	**.336**
05382000	WI-17	9	.063	-.001	-.23	.932	-.002	-2.59	.278	132,000	24.6	91,500	-4,600	-5.00	.036
05407000	WI-20	7	.046	-.001	-2.79	.366	-.002	-3.46	.207	498,000	18.5	384,000	-19,000	-4.84	.058
05420500	**IA-21**	**3**	**.106**	**-.008**	**-7.66**	**.000**	**-.008**	**-7.52**	**.000**	**4,120,000**	**18.6**	**6,250,000**	**-500,000**	**-7.94**	**.000**
054310157	WI-24	9	.215	-.066	-30.70	.000	-.066	-30.7	.000	NA	NA	236	NC	-3.66	.000
05465500	**IA-25**	**9**	**.024**	**-.002**	**-7.35**	**.003**	**-.002**	**-7.59**	**.000**	**859,000**	**26.5**	**221,000**	**-18,000**	**-8.26**	**.000**
05474000	IA-26	7	.067	-.005	-6.71	.021	-.005	-7.38	.003	254,000	22.8	138,000	-11,000	-8.28	.000
05500000	MO-28	8	.069	-.004	-5.90	.000	-.005	-6.89	.000	40,100	24.9	9,130	-740	-8.13	.000
05525500	IL-30	9	.033	-.002	-6.94	.000	-.002	-7.09	.000	42,000	36.3	4,800	-370	-7.62	.007

Table 7. Summary of results for analyses of flow-adjusted trends, overall trends, and trends in loads for dissolved ammonia (NH4, USGS parameter code 00608), 1933 to 2004.—Continued

[USGS, U.S. Geological Survey; ID, identification; mg/L, milligram per liter; mg/L/yr, milligram per liter per year; %/yr, percent per year; kg, kilogram; kg/km², kilogram per square kilometer; NA, not available; NC, not calculated; sites in **bold** indicate long-term evaluation sites]

USGS site number	State-ID number (figure 1)	Water-quality model (table 3)	Concentration							Annual loads					
				Flow-adjusted trend			Overall trend								
			Reference concentration (mg/L)	Trend (mg/L/yr)	Trend (%/yr)	p-value	Trend (mg/L/yr)	Trend (%/yr)	p-value	Median annual load (kg)	Median yield (kg/km²)	Reference load (kg)	Trend (kg/yr)	Trend (%/yr)	p-value
Central part—continued															
05531500	IL-31	7	0.192	-0.011	-5.76	0.000	-0.011	-5.75	0.000	35,000	118	23,400	-1,400	-5.91	0.000
05532500	IL-32	8	.167	-.012	-6.91	.000	-.012	-6.93	.000	157,000	96.3	86,400	-6,200	-7.24	.000
05540275	IL-33	3	.080	-.004	-5.24	.000	-.004	-5.41	.000	772	30.1	461	-31	-6.68	.000
05586100	IL-37	7	.112	-.008	-6.76	.000	-.007	-6.48	.000	2,020,000	29.1	2,780,000	-210,000	-7.68	.000
07014500	MO-38	9	.018	-.001	-7.63	.003	-.001	-7.83	.001	24,300	6.36	20,700	-1,700	-8.28	.000
07018100	MO-39	9	.016	-.001	-5.49	.018	-.001	-5.81	.006	20,600	10.8	7,290	-530	-7.31	.000
Eastern part															
03085000	PA-40	7	0.103	-0.004	-3.44	0.054	-0.004	-3.67	0.036	332,000	17.5	688,000	-15,000	-2.24	0.349
03353637	IN-42	9	.028	-.002	-7.44	.000	-.002	-7.49	.000	769	17.5	247	-19.0	-7.85	.000
04186500	OH-43	7	.058	-.005	-8.53	.000	-.005	-8.55	.000	35,400	41.1	4,160	-360	-8.58	.004
04193500	**OH-44**	**7**	**.056**	**-.004**	**-6.69**	**.001**	**-.004**	**-6.63**	**.002**	**618,000**	**37.7**	**106,000**	**-6,900**	**-6.50**	**.031**
0422026250	NY-45	7	.032	.002	7.75	.000	.002	7.48	.000	1,810	69.3	202	NC	18.46	.000
04232034	NY-46	7	.003	.001	1.82	.687	.001	1.48	.738	249	2.45	83	.34	.40	.929
0423204920	NY-47	8	.019	.001	2.52	.167	.001	2.87	.128	285	15.8	89	.27	.30	.902
0423205025	NY-48	9	.022	-.001	-5.95	.000	-.001	-5.94	.000	4,260	11.5	2,110	-14	-6.77	.000

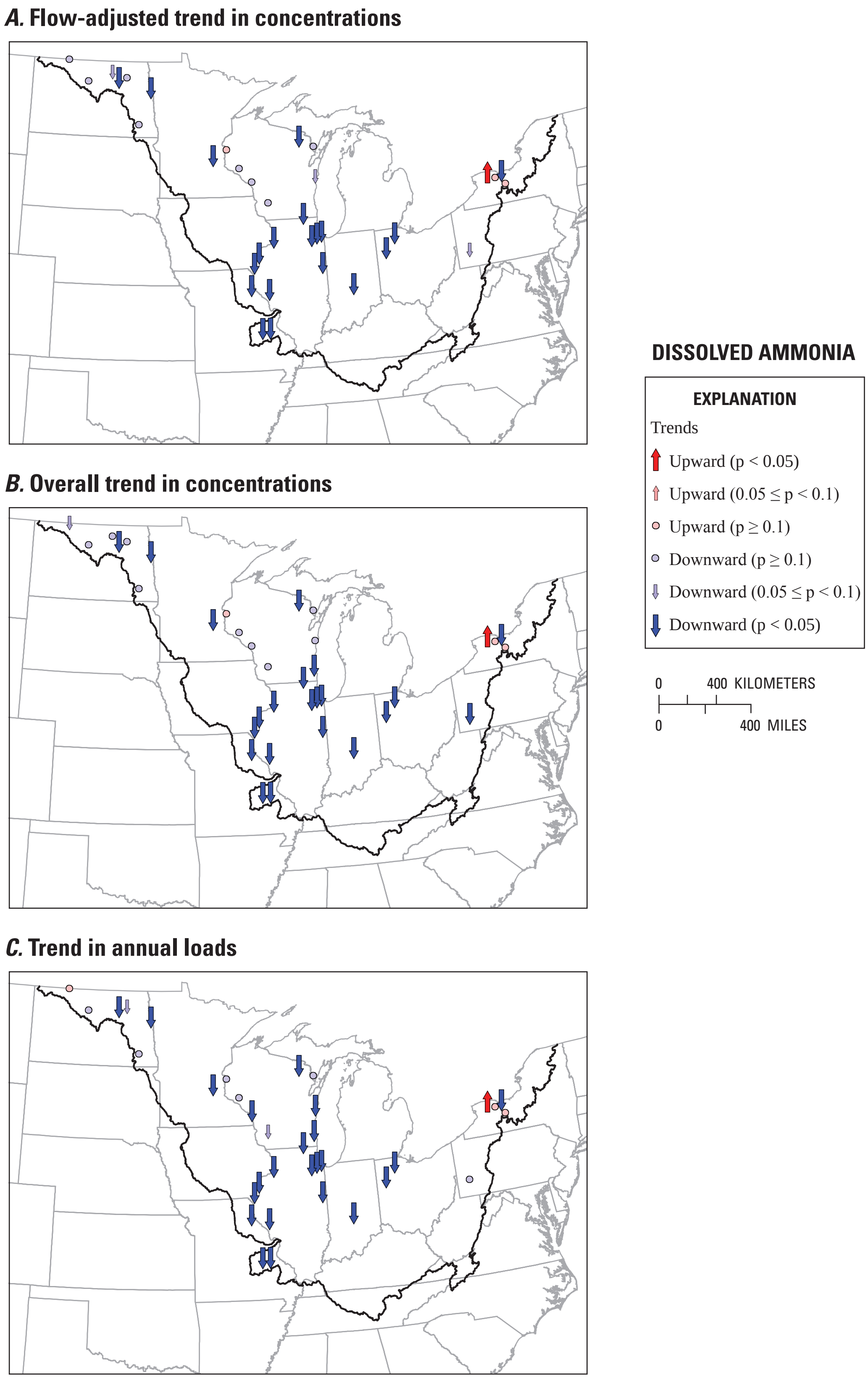


Figure 18. Trends in dissolved ammonia from 1993 to 2004 for ***A***, flow-adjusted concentrations, ***B***, overall concentrations, and ***C***, annual loads throughout the study area.

Total Kjeldahl Nitrogen

General Patterns in Total Kjeldahl Nitrogen Concentrations and Yields

Median midmonthly Kjeldahl nitrogen (KJN) concentrations at 36 of 49 trend sites that had sufficient data for analysis, based on data from 1993 to 2004, ranged from 0.200 to 1.50 mg/L. The mean and median values of the median KJN concentrations at these sites were 0.88 mg/L. Concentrations were uniformly high throughout the western parts of the study area and sporadically high at isolated sites throughout the rest of the study area. The lowest concentrations were at sites in eastern Ohio and eastern Pennsylvania. The highest concentrations were in areas with intensive agriculture, and the lowest concentrations were in forested areas.

Median annual KJN yields at the trend sites ranged from 9.13 to 796 kg/km^2 (table 8), based on data from 1993 to 2004. The mean and median values from the median yields at these sites were 313 and 261 kg/km^2, respectively. The highest yields were from two sites in Illinois and one site in Missouri. The lowest yields were throughout the northwestern part of the study area, including three sites in North Dakota and three sites in northern Wisconsin.

Trends in Total Kjeldahl Nitrogen Concentrations

The FA trends in KJN concentrations from 1993 to 2004 showed no distinct spatial pattern. Of the 36 sites that had sufficient streamflow and KJN concentration data for trend analysis (table 8; fig. 19***A***), 17 sites had tendency toward decreasing concentrations, and 19 sites had a tendency toward increasing concentrations. Seven sites had significant downward trends, and six sites had significant upward trends ($p < 0.10$).

Of the 36 sites that had sufficient streamflow and KJN concentration data from 1993 to 2004 for analysis (table 8; fig. 19***B***), 18 sites had downward OA trends in concentrations, and 18 sites had upward trends. Of these, nine sites had significant downward trends, and six sites had significant upward trends ($p < 0.10$). When the OA trends in KJN concentrations were examined no distinct spatial patterns were evident; however, the tendencies and statistical results were slightly different than those of the FA trends. The differences in the results of the FA and OA trend analyses were attributable to differences in the trends in streamflow (fig. 12).

Trends in Total Kjeldahl Nitrogen Loads

From 1993 to 2004, KJN loads at the 36 trend sites that had sufficient data for analysis showed a tendency toward decreasing loads at 30 sites and a tendency toward increasing loads at 6 sites (table 8; fig. 19***C***). The sites with downward trends in KJN loads were distributed across all three parts of study area, and three of the six sites with upward trends were in the eastern part of the study area. Thirteen of the sites with downward trends in loads had significant trends, whereas only one of the six sites with upward trends in loads had a significant trend ($p < 0.10$).

Dissolved Nitrite plus Nitrate Nitrogen

General Patterns in Dissolved Nitrite plus Nitrate Nitrogen Concentrations and Yields

Median midmonthly dissolved nitrite plus nitrate nitrogen (NO2NO3) concentrations at 29 of the 49 trend sites ranged from 0.05 to 8.11 mg/L, based on data from 1993 to 2004. The mean and median values from the median NO2NO3 concentrations at these sites were 2.21 and 0.65 mg/L, respectively. The highest concentrations were in the central and eastern parts of the study area, and the lowest concentrations were in the north-central and western parts of the study area. The highest concentrations were in areas with intensive agriculture, and the lowest concentrations were in forested areas.

Median annual NO2NO3 yields at the trend sites ranged from 1.80 to 4,190 kg/km^2, based on data from 1993 to 2004 (table 9). The mean and median values of the median yields at these trend sites were 774 and 364 kg/km^2, respectively. The highest yields were at two sites in Ohio and one site each in Illinois and Iowa. The lowest yields were from six sites from throughout the northwest and north-central parts of the study area, three sites in North Dakota and three sites in northern Wisconsin.

Trends in Dissolved Nitrite plus Nitrate Nitrogen Concentrations

The FA trends in NO2NO3 concentrations from 1993 to 2004 showed no distinct spatial pattern (fig. 20***A***). Of the 29 sites that had sufficient streamflow and NO2NO3 concentration data for analysis (table 9 and fig. 20***A***), 12 sites had downward trends, and 17 sites had upward trends. Of these sites, three sites (one in North Dakota, one in Illinois, and one in Indiana) had significant downward trends, and four sites (one in Iowa, one in Illinois, and two in Wisconsin) had significant upward trends ($p < 0.10$).

When the OA trends in NO2NO3 concentrations from 1993 to 2004 were examined, no distinct spatial patterns were evident; however, tendencies and trend results were somewhat different than those of the FA trends. These differences in the results of the FA and OA trend analyses are attributed to differences in streamflow trends (fig. 12). Of the 29 sites that had sufficient streamflow and NO2NO3 concentration data for analysis (table 9; fig. 20***B***), 13 sites had downward OA trends, and 16 sites had upward OA trends. Of these sites, four sites had significant downward trends, and three sites had significant upward trends ($p < 0.10$).

Table 8. Summary of results for analyses of flow-adjusted trends, overall trends, and trends in loads for total Kjeldahl nitrogen (KJN, USGS parameter code 00625), 1993 to 2004.

[USGS, U.S. Geological Survey; ID, identification; mg/L, milligram per liter; mg/L/yr, milligram per liter per year; %/yr, percent per year; kg, kilogram; kg/km^2, kilogram per square kilometer; kg/yr, kilogram per year; NA, not available; NC, not calculated; sites in **bold** indicate long-term evaluation sites]

			Concentration							Annual loads					
				Flow-adjusted trend			Overall trend								
USGS site number	State-ID number (figure 1)	Water-quality model (table 3)	Reference concentration (mg/L)	Trend (mg/L/yr)	Trend (%/yr)	p-value	Trend (mg/L/yr)	Trend (%/yr)	p-value	Median annual load (kg)	Median yield (kg/km^2)	Reference load (kg)	Trend (kg/yr)	Trend (%/yr)	p-value
Western part															
05056100	ND-1	9	1.266	0.017	1.37	0.525	0.036	2.84	0.328	NA	NA	12,400	-450	-3.66	0.552
05056200	ND-2	7	1.497	.039	2.61	.031	.066	4.44	.009	NA	NA	25,400	-1,500	-5.96	.266
05056239	ND-3	9	1.126	-.009	-.79	.767	.005	.44	.881	NA	NA	20,900	-1,300	-6.45	.163
05058700	ND-4	3	1.017	-.002	-.19	.936	-.006	-.60	.795	511,000	79.3	233,000	-6,800	-2.93	.433
05064500	MN-5	3	1.170	-.058	-4.97	.004	-.060	-5.09	.001	3,400,000	345	2,330,000	-140,000	-5.82	.006
05082500	ND-6	3	1.029	-.033	-3.24	.011	-.036	-3.49	.003	5,470,000	555	4,050,000	-210,000	-5.10	.011
05114000	**ND-7**	**9**	**1.955**	**-.024**	**-1.25**	**.276**	**-.034**	**-1.72**	**.185**	**92,300**	**11.7**	**10,800**	**NC**	**32.9**	**.122**
05120000	ND-8	5	1.092	.017	1.55	.244	.016	1.46	.266	104,000	9.13	41,700	3,190	7.65	.367
Central part															
04063700	WI-9	9	0.411	0.004	1.04	0.228	-0.002	-0.47	0.619	55,200	153	34,200	-1,100	-3.32	0.035
04067500	WI-10	9	.323	-.003	-.87	.510	-.003	-.88	.516	1,060,000	104	769,000	-7,000	-.90	.636
04085427	WI-12	3	1.421	.020	1.40	.460	.016	1.14	.530	337,000	248	187,000	-3,700	-1.96	.530
04087000	**WI-13**	**8**	**.821**	**.004**	**.52**	**.424**	**.005**	**.62**	**.340**	**406,000**	**225**	**267,000**	**-2,500**	**-.92**	**.591**
05287890	MN-14	9	1.071	.023	2.17	.001	.003	.33	.728	104,000	467	24,400	-1,300	-5.38	.075
05340500	WI-15	9	.387	.009	2.25	.293	.006	1.51	.481	2,410,000	149	1,410,000	3,510	.25	.923
05369500	**WI-16**	**9**	**.544**	**.006**	**1.03**	**.222**	**.005**	**.96**	**.244**	**5,130,000**	**220**	**3,540,000**	**-15,000**	**-.43**	**.775**
05382000	WI-17	8	.657	.010	1.54	.241	-.001	-.09	.940	1,400,000	261	952,000	-33,000	-3.50	.046
05407000	WI-20	7	.673	.020	2.99	.046	.016	2.43	.064	6,610,000	245	5,610,000	-27,000	-.48	.756
05420500	**IA-21**	**9**	**1.625**	**-.030**	**-1.83**	**.075**	**-.035**	**-2.17**	**.025**	**53,500,000**	**241**	**95,700,000**	**-4,400,000**	**-4.60**	**.000**
05465500	**IA-25**	**7**	**1.076**	**.024**	**2.19**	**.278**	**.041**	**3.82**	**.066**	**11,800,000**	**366**	**9,860,000**	**-320,000**	**-3.29**	**.141**
05500000	MO-28	8	.859	-.001	-.04	.965	-.015	-1.75	.099	900,000	560	115,000	-7,800	-6.81	.021
05514500	MO-29	7	.942	-.039	-4.17	.011	-.040	-4.25	.019	1,000,000	427	106,000	-4,900	-4.59	.412
05525500	IL-30	8	.538	-.002	-.42	.929	-.005	-1.00	.833	527,000	456	62,800	-2,600	-4.13	.524
05531500	IL-31	9	1.458	-.030	-2.08	.036	-.030	-2.05	.038	237,000	796	178,000	-4,300	-2.43	.074
05532500	IL-32	9	1.586	-.036	-2.29	.049	-.035	-2.21	.058	946,000	580	814,000	-28,000	-3.42	.030

Table 8. Summary of results for analyses of flow-adjusted trends, overall trends, and trends in loads for total Kjeldahl nitrogen (KJN, USGS parameter code 00625), 1993 to 2004.—Continued

[USGS, U.S. Geological Survey; ID, identification; mg/L, milligram per liter; mg/L/yr, milligram per liter per year; %/yr, percent per year; kg, kilogram; kg/km², kilogram per square kilometer; kg/yr, kilogram per year; NA, not available; NC, not calculated; sites in **bold** indicate long-term evaluation sites]

USGS site number	State-ID number (figure 1)	Water-quality model (table 3)	Concentration							Annual loads					
				Flow-adjusted trend			Overall trend								
			Reference concentration (mg/L)	Trend (mg/L/yr)	Trend (%/yr)	p-value	Trend (mg/L/yr)	Trend (%/yr)	p-value	Median annual load (kg)	Median yield (kg/km²)	Reference load (kg)	Trend (kg/yr)	Trend (%/yr)	p-value
							Central part—continued								
05540275	IL-33	3	0.589	0.011	1.81	0.155	0.007	1.24	0.322	7,980	311	3,400	-92.0	-2.71	0.329
05586100	IL-37	8	1.150	.013	1.13	.345	.012	1.06	.346	25,300,000	365	28,000,000	-1,200,000	-4.41	.010
07014500	MO-38	8	.172	-.002	-1.14	.695	-.005	-2.72	.251	599,000	157	193,000	-12,000	-6.11	.004
07018100	MO-39	8	.203	-.001	-.67	.833	-.004	-1.78	.562	485,000	255	95,100	-5,200	-5.47	.043
							Eastern part								
03085000	PA-40	8	0.228	0.004	1.96	0.690	0.004	1.83	0.709	3,810,000	200	1,510,000	68,500	4.52	0.435
03353637	IN-42	9	.223	.011	4.92	.003	.008	3.67	.057	14,100	319	1,840	-5.30	-.29	.952
04186500	OH-43	8	.773	-.007	-.90	.602	-.007	-.94	.597	355,000	413	59,400	-750	-1.25	.777
04193500	**OH-44**	**8**	**1.065**	**.022**	**2.03**	**.093**	**.022**	**2.09**	**.090**	**8,560,000**	**522**	**1,950,000**	**54,200**	**2.78**	**.457**
0422026250	NY-45	9	1.111	.027	2.46	.001	.029	2.60	.000	12,700	484	6,990	NC	10.24	.002
04232034	NY-46	9	.621	.009	1.50	.260	.005	.87	.568	22,200	218	16,300	-23.0	-.14	.953
0423204920	NY-47	9	.781	-.010	-1.27	.129	-.011	-1.45	.090	6,590	365	3,690	-120	-3.16	.115
0423205025	NY-48	9	.831	-.018	-2.19	.000	-.020	-2.40	.000	84,200	227	80,600	-3,200	-3.96	.007

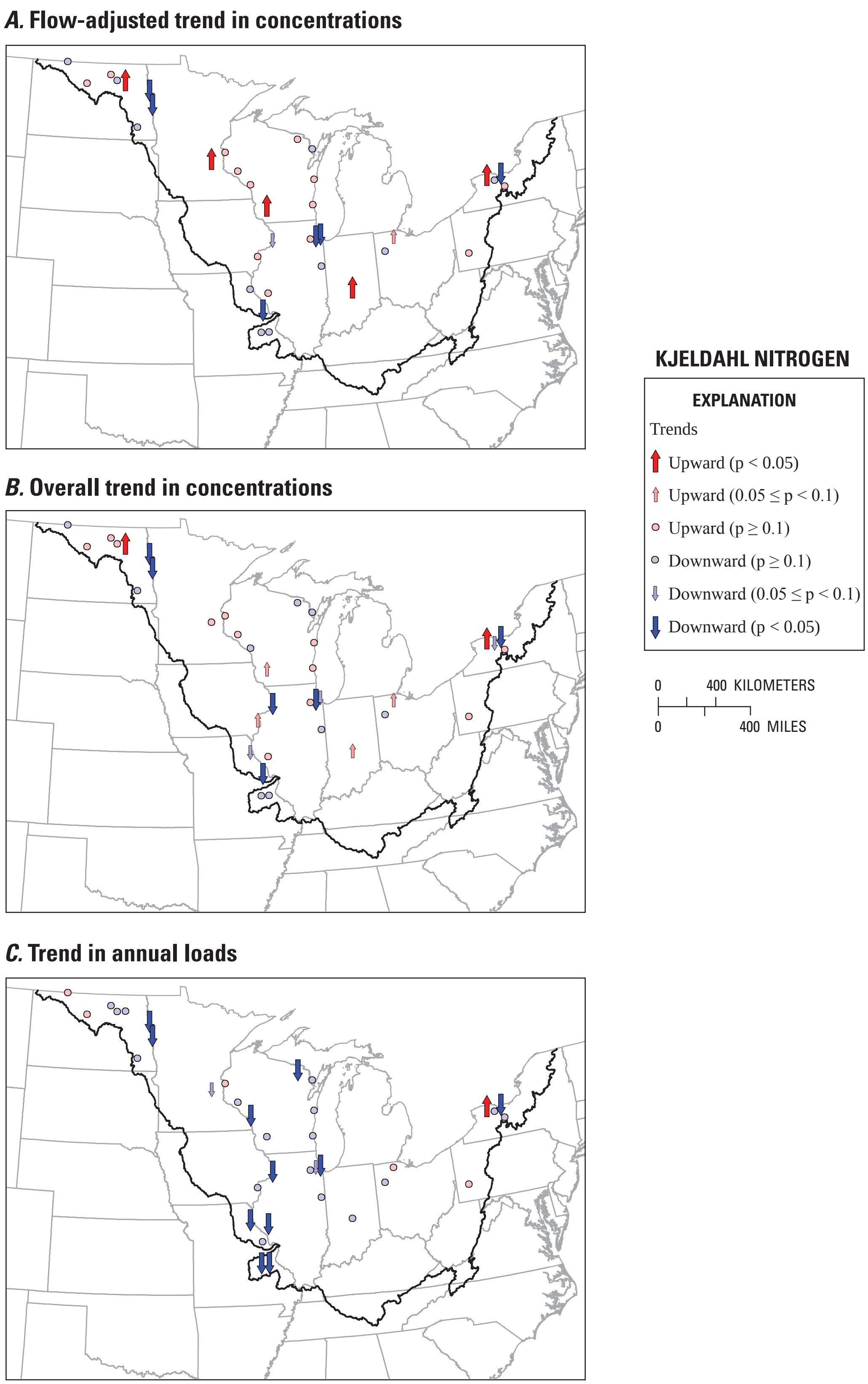


Figure 19. Trends in total Kjeldahl nitrogen from 1993 to 2004 for ***A***, flow-adjusted concentrations, ***B***, overall concentrations, and ***C***, annual loads throughout the study area.

Table 9. Summary of results for analyses of flow-adjusted trends, overall trends, and trends in loads for dissolved nitrite plus nitrate nitrogen (NO3, USGS parameter code 00631), 1993 to 2004.-

[USGS, U.S. Geological Survey; ID, identification; mg/L, milligram per liter; mg/L/yr, milligram per liter per year; %/yr, percent per year; kg, kilogram; kg/km², kilogram per square kilometer; kg/yr, kilogram per year; NA, not available; NC, not calculated; sites in **bold** indicate long-term evaluation sites]

USGS site number	State-ID number (figure 1)	Water-quality model (table 3)	Concentration							Annual loads					
				Flow-adjusted trend			Overall trend								
			Reference concentration (mg/L)	Trend (mg/L/yr)	Trend (%/yr)	p-value	Trend (mg/L/yr)	Trend (%/yr)	p-value	Median annual load (kg)	Median yield (kg/km²)	Reference load (kg)	Trend (kg/yr)	Trend (%/yr)	p-value
							Western part								
05056100	ND-1	8	0.018	0.003	15.00	0.272	0.003	16.76	0.251	NA	NA	173	4.3	2.47	0.852
05056200	ND-2	8	.036	-.001	-4.00	.553	-.002	-5.39	.354	NA	NA	537	-43.0	-8.08	.140
05056239	ND-3	7	.043	-.002	-5.67	.249	-.003	-7.37	.103	NA	NA	704	-60.0	-8.55	.081
05058700	ND-4	7	.151	.012	7.65	.456	.007	4.51	.623	279,000	43.2	34,000	254	.75	.936
05064500	MN-5	7	.436	.046	10.60	.208	.043	9.93	.203	3,380,000	344	866,000	54,600	6.31	.416
05082500	ND-6	8	.421	-.014	-3.26	.474	-.018	-4.25	.293	3,990,000	405	1,640,000	-93,000	-5.64	.165
05114000	**ND-7**	**7**	**.046**	**-.001**	**-.76**	**.834**	**.001**	**3.12**	**.561**	**14,100**	**1.80**	**237**	**NC**	**61.71**	**.122**
05120000	ND-8	7	.153	-.009	-6.10	.062	-.009	-5.86	.089	58,300	5.11	5,690	-240	-4.16	.535
							Central part								
04063700	WI-9	7	0.044	0.001	1.57	0.447	0.001	1.54	0.439	7,330	20.4	3,680	-74.0	-2.01	0.310
04067500	WI-10	7	.088	-.002	-2.58	.172	-.002	-2.59	.170	397,000	39.0	210,000	-5,500	-2.61	.247
04085427	WI-12	7	.231	.027	11.51	.304	.024	10.24	.330	518,000	380	30,300	1,310	4.32	.650
04087000	**WI-13**	**8**	**.669**	**.056**	**8.40**	**.000**	**.043**	**6.37**	**.019**	**827,000**	**458**	**215,000**	**8,360**	**3.88**	**.356**
05287890	MN-14	7	.181	.004	2.39	.351	-.003	-1.56	.536	24,100	108	3,840	-240	-6.19	.086
05340500	WI-15	7	.155	.006	3.57	.144	.006	3.60	.135	919,000	56.9	598,000	12,500	2.08	.414
05369500	**WI-16**	**7**	**.554**	**.013**	**2.32**	**.053**	**.015**	**2.78**	**.028**	**4,700,000**	**201**	**3,610,000**	**40,500**	**1.12**	**.452**
05382000	WI-17	7	.528	.009	1.61	.223	.012	2.26	.090	969,000	180	767,000	-16,000	-2.08	.166
05407000	WI-20	7	.645	-.015	-2.26	.461	-.013	-2.04	.483	7,470,000	277	5,370,000	-200,000	-3.79	.147
05420500	**IA-21**	**8**	**2.052**	**-.049**	**-2.41**	**.253**	**-.106**	**-5.16**	**.003**	**111,000,000**	**502**	**117,000,000**	**-7,700,000**	**-6.56**	**.000**
05465500	**IA-25**	**8**	**3.982**	**.481**	**12.07**	**.006**	**.040**	**1.00**	**.786**	**71,400,000**	**2,210**	**35,900,000**	**-1,600,000**	**-4.50**	**.261**
05474000	IA-26	5	5.885	.360	6.11	.195	-.328	-5.57	.174	18,600,000	1,660	11,900,000	-920,000	-7.72	.059
05514500	MO-29	7	.293	.007	2.37	.752	.006	2.06	.794	3,510,000	1,500	32,600	478	1.47	.896
05540275	IL-33	7	.883	-.046	-5.24	.000	-.051	-5.74	.000	12,600	490	5,070	-350	-6.88	.001
05586100	IL-37	9	3.364	.077	2.28	.038	-.009	-.27	.802	119,000,000	1,720	78,100,000	-3,900,000	-4.97	.011
07018100	MO-39	9	.013	.006	43.56	.060	.002	13.46	.317	195,000	102	5,140	96.7	1.88	.854

Table 9. Summary of results for analyses of flow-adjusted trends, overall trends, and trends in loads for dissolved nitrite plus nitrate nitrogen (NO3, USGS parameter code 00631), 1993 to 2004.—Continued

[USGS, U.S. Geological Survey; ID, identification; mg/L, milligram per liter; mg/L/yr, milligram per liter per year; %/yr, percent per year; kg, kilogram; kg/km², kilogram per square kilometer; kg/yr, kilogram per year; NA, not available; NC, not calculated; sites in **bold** indicate long-term evaluation sites]

USGS site number	State-ID number (figure 1)	Water-quality model (table 3)	Concentration							Annual loads					
				Flow-adjusted trend			Overall trend								
			Reference concentration (mg/L)	Trend (mg/L/yr)	Trend (%/yr)	p-value	Trend (mg/L/yr)	Trend (%/yr)	p-value	Median annual load (kg)	Median yield (kg/km²)	Reference load (kg)	Trend (kg/yr)	Trend (%/yr)	p-value
Eastern Part															
03085000	PA-40	5	0.672	0.016	2.45	0.282	0.013	1.97	0.380	9,320,000	490	4,500,000	224,000	4.99	0.125
03267900	OH-41	5	3.893	-.032	-.81	.354	.056	1.43	.146	1,290,000	1,610	585,000	NC	30.26	.000
03353637	IN-42	8	.857	-.030	-3.50	.000	-.032	-3.77	.000	13,900	315	7,460	-400	-5.34	.056
04186500	OH-43	8	1.973	.020	1.04	.799	.016	.81	.863	3,600,000	4,190	141,000	623	.44	.952
04193500	**OH-44**	**8**	**3.085**	**-.077**	**-2.48**	**.405**	**-.066**	**-2.15**	**.557**	**46,100,000**	**2,810**	**5,760,000**	**-99,000**	**-1.72**	**.760**

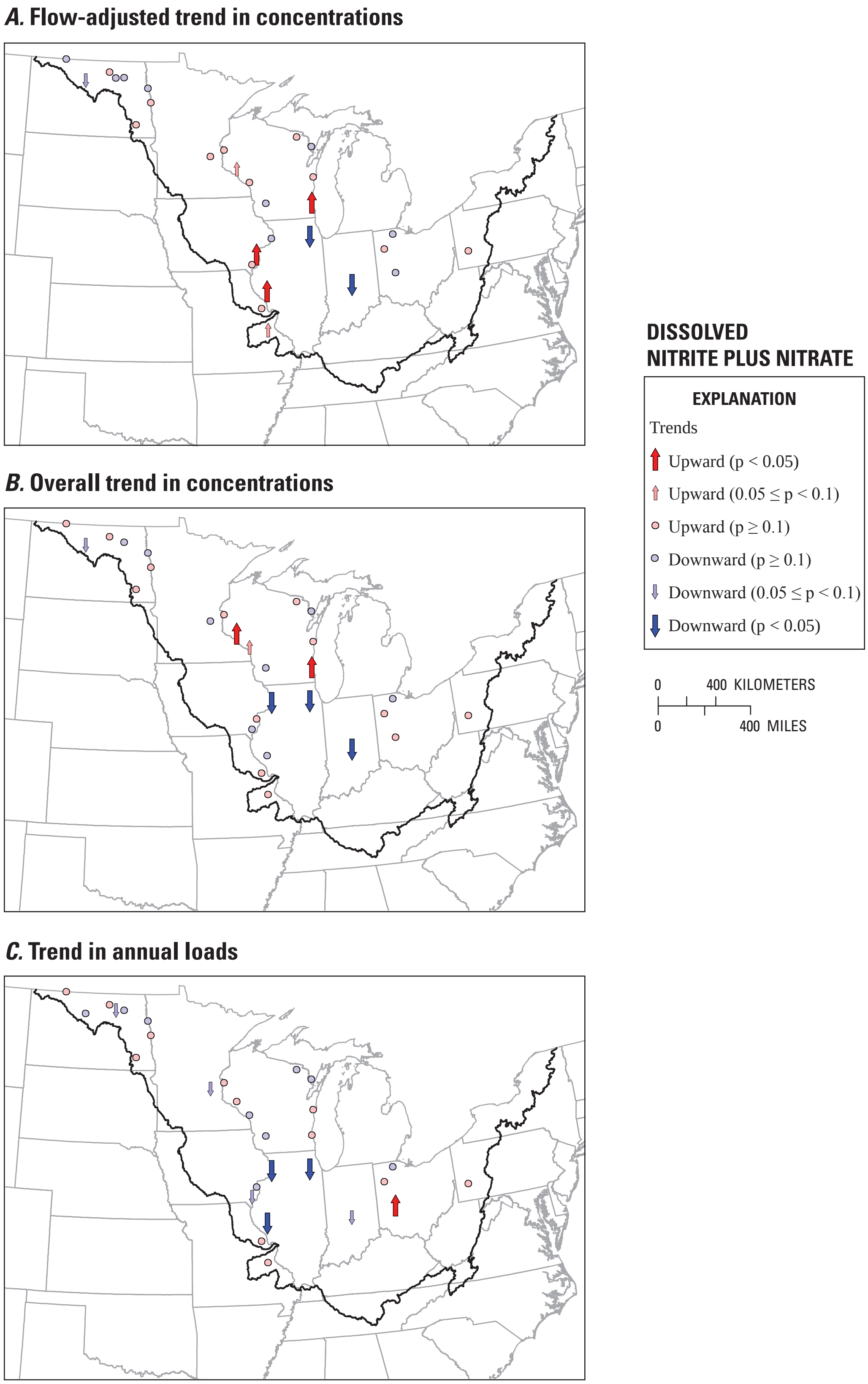


Figure 20. Trends in dissolved nitrite plus nitrate from 1993 to 2004 for ***A***, flow-adjusted concentrations, ***B***, overall concentrations, and ***C***, annual loads throughout the study area.

Trends in Dissolved Nitrite plus Nitrate Loads

Of the 29 sites that had sufficient data from 1993 to 2004 for analysis, 16 sites had a tendency toward decreasing NO2NO3 loads, and 13 sites had a tendency toward increasing loads (table 9; fig. 20***C***). Seven of the sites with downward trends in loads had significant downward trends, whereas only one of the 12 sites with upward trends had a significant upward trend ($p < 0.10$).

Total Phosphorus

General Patterns in Total Phosphorus Concentrations and Yields

Median midmonthly total phosphorus (TP) concentrations ranged from 0.005 to 4.45 mg/L, based on water-quality data from all sites from 1970 to 2004. The overall mean and median of the median TP concentrations were 0.17 and 0.11 mg/L, respectively. High concentrations occurred throughout the study area, but especially at sites along a belt that spans North Dakota, southern Minnesota, Iowa, Illinois, Indiana, and Ohio (fig. 21***A***). The lowest concentrations occurred in forested areas of northern Wisconsin, Michigan, and Minnesota and along the eastern edge of the study area (figs. 5 and 21). The highest concentrations generally occurred in areas with large amounts of agriculture and urban land use. The spatial pattern in TP concentrations was similar to that for TN; however, sporadically high and low concentrations were found throughout the study area. Median midmonthly TP concentrations ranged from 0.028 to 1.49 mg/L at the 41 trend sites that had sufficient data for analysis from 1993 to 2004. The mean and median values of the median concentrations at these sites were 0.22 and 0.16 mg/L, respectively. The distribution in concentrations of these sites was consistent with that in fig. 21***A***.

Median annual TP yields ranged from 0.421 to 1,950 kg/km^2, based on water-quality and streamflow data from all sites from 1970 to 2004. The mean and median values of the median TP yields at these sites were 82.2 and 48.8 kg/km^2, respectively. The highest yields were throughout the southern one-half of the study area, especially through the middle of the study area (fig. 21***B***). The lowest yields were throughout the northern one-half of the study area. Similar to TN, the highest TP yields were in areas with intense row crop agriculture and moderate runoff. The areas with the highest runoff (fig. 3) had relatively low TP concentrations, such as in northern Wisconsin and southeastern parts of the study area. The major difference in the distributions in TP concentrations and yields was in the northwestern part of the study area, which had high concentrations but low yields, which can be attributed to lower runoff than other agricultural areas. Based on 1993 to 2003 data, median annual TP yields at the trend sites ranged from 2.65 to 649 kg/km^2 (table 10), which is within the range found for the entire data set. The distribution in yields was consistent with that in fig. 21***B***. The mean and median values of the median yields at these trend sites were 86.3 and 44.8 kg/km^2, respectively.

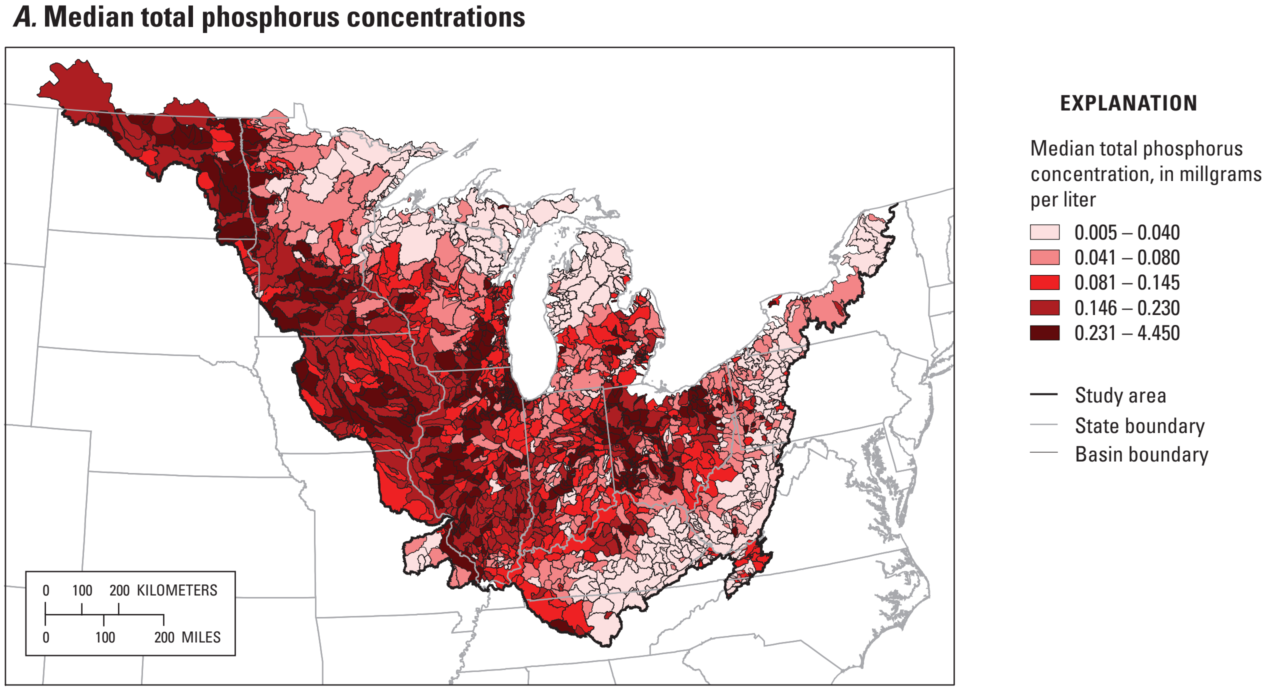
Trends in Total Phosphorus Concentrations

From 1993 to 2004, FA trends in TP concentrations at 24 of the 40 sites that had sufficient data for analysis indicated a tendency toward increasing concentrations (table 10; fig. 22***A***). The 24 sites with increasing TP concentrations were distributed throughout the entire study area; however, most of these sites were in the central part and the eastern side of the western part of the study area. Twelve of the 24 sites had significant upward trends ($p < 0.10$). These upward FA trends indicate that the effects of anthropogenic activities continue to increase in these areas and add additional TP to the streams. Concentrations at 7 of the 16 sites with downward trends were significant ($p < 0.10$), indicating that the effects of anthropogenic activities were decreasing.

The OA trends in TP concentrations from 1993 to 2004 showed a general tendency toward increasing concentrations at 21 of the 41 sites (table 10; fig. 22***B***); however, the upward trends were significant at only 7 of these sites ($p < 0.10$). Of the 20 sites with downward trends, only 7 sites had significant trends. The sites with significant upward or downward trends were scattered throughout the study area.

The differences between FA trends and OA trends in TP concentrations are attributed to the trends in streamflow. The major difference was in the central part of the study area—several sites in Minnesota, Wisconsin, Illinois, Iowa, and Missouri had significant downward trends in streamflow (fig. 12). In this area, only four of the nine sites that had statistically significant FA trends in TP concentrations had statistically significant OA trends in concentrations. Very few changes occurred in the other parts of the study area.

To place the 1993 to 2004 trends in TP concentrations into a longer term context, FACs were computed from 1975 to 2004 for the six sites with sufficient data and examined for trends with QWTREND (table 11; fig. 23). Three sites had little or no trend from 1975 to 2004; the Iowa River (05465500) and Mississippi River (05420500) showed no trend, and the Chippewa River (05369500) had a downward trend from 1975 to 1985 and no trend after that. The Souris River (05114000) site had no trend from 1975 to 1979, then a downward trend until 1988 and an upward trend through 2004. The Milwaukee River (04087000) had a downward trend from 1975 through 1995 and then an upward trend though 2004. The Maumee River (04193500) had a very strong downward trend from 1975 through 1982, followed by an upward trend until 1989, and another downward trend through 2004. The flow-adjusted concentration in the Maumee River in 2004 was less than one-half of the flow-adjusted concentration in 1975 (table 11).



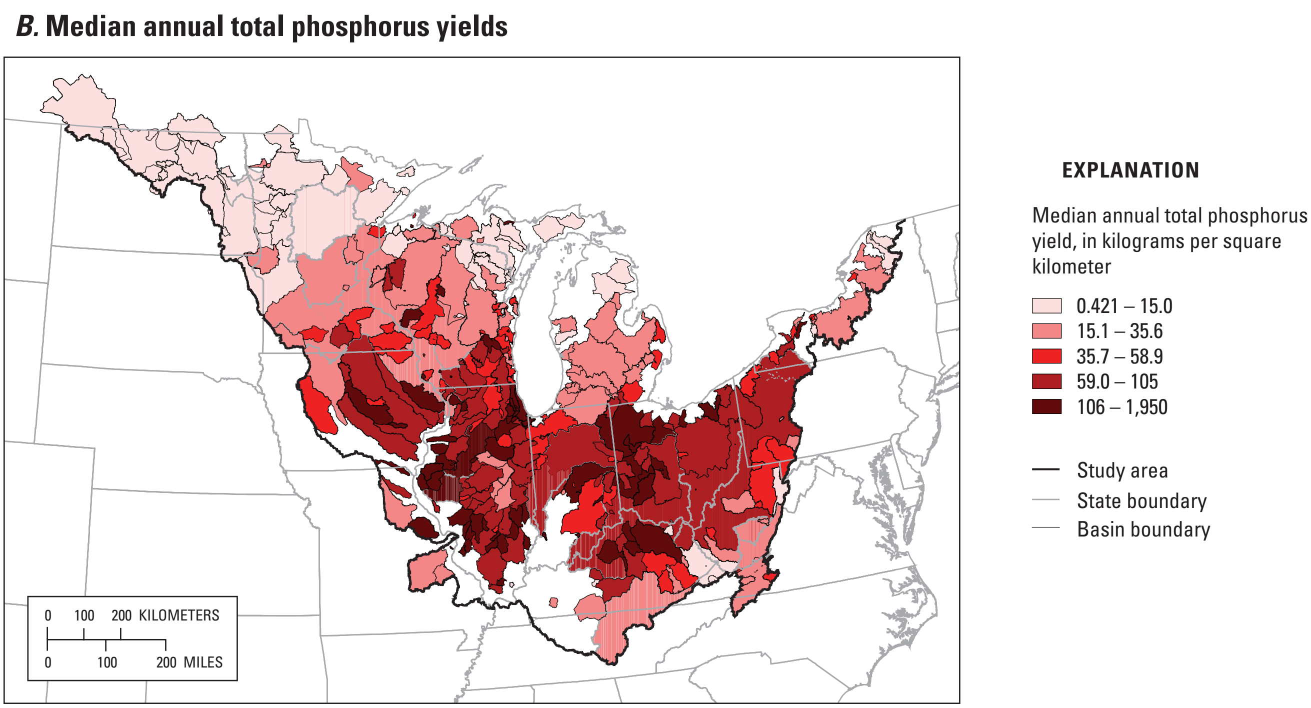


Figure 21. Spatial patterns from 1970 to 2004 in ***A***, median total phosphorus concentrations and ***B***, median annual total phosphorus yields throughout the study area.

Table 10. Summary of results for analyses of flow-adjusted trends, overall trends, and trends in loads for total phosphorus (TP, USGS parameter code 00665), 1993 to 2004.

[USGS, U.S. Geological Survey; ID, identification; mg/L, milligram per liter; mg/L/yr, milligram per liter per year; %/yr, percent per year; kg, kilogram; kg/km², kilogram per square kilometer; kg/yr, kilogram per year; NA, not available; NC, not calculated; sites in **bold** indicate long-term evaluation sites]

			Concentration							Annual loads					
				Flow-adjusted trend			Overall trend								
USGS site number	State-ID number (figure 1)	Water-quality model (table 3)	Reference concentration (mg/L)	Trend (mg/L/yr)	Trend (%/yr)	p-value	Trend (mg/L/yr)	Trend (%/yr)	p-value	Median annual load (kg)	Median yield (kg/km²)	Reference load (kg)	Trend (kg/yr)	Trend (%/yr)	p-value
Western part															
05056100	ND-1	7	0.362	0.001	0.11	0.960	0.001	0.28	0.898	NA	NA	3,430	-160	-4.79	0.455
05056200	ND-2	5	.302	.011	3.66	.183	.015	4.83	.095	NA	NA	5,040	-300	-5.88	.309
05056239	ND-3	3	.231	.008	3.39	.370	.005	2.05	.581	NA	NA	4,100	-250	-6.01	.314
05058700	ND-4	3	.187	.002	.89	.854	-.001	-.11	.980	141,000	21.8	42,600	-1,100	-2.58	.632
05064500	MN-5	7	.186	.019	10.14	.033	.017	9.15	.032	1,200,000	122	369,000	21,000	5.67	.279
05082500	ND-6	5	.151	.012	7.83	.062	.008	5.42	.159	1,510,000	153	590,000	6,710	1.14	.807
05114000	**ND-7**	**9**	**.385**	**-.010**	**-2.70**	**.061**	**-.011**	**-2.91**	**.059**	**20,800**	**2.65**	**2,130**	**NC**	**25.80**	**.185**
05120000	ND-8	9	.263	-.003	-1.09	.483	-.002	-.93	.550	30,400	2.67	9,980	379	3.80	.630
Central part															
04063700	WI-9	3	0.025	-0.001	-2.00	0.374	-0.001	-2.83	0.220	2,410	6.70	2,090	-100	-4.87	0.028
04067500	WI-10	1	.027	NA	NA	NA	-.001	-.01	.977	79,100	7.76	64,800	-25.0	-.04	.977
04073468	WI-11	8	.131	-.003	-2.00	.000	-.003	-2.13	.000	4,050	29.2	3,480	-130	-3.86	.079
04085427	WI-12	7	.184	-.002	-.85	.739	-.002	-1.25	.609	40,700	29.9	24,100	-870	-3.60	.269
04087000	**WI-13**	**8**	**.091**	**.001**	**.63**	**.463**	**.001**	**.55**	**.519**	**49,500**	**27.5**	**29,400**	**-290**	**-.98**	**.617**
05287890	MN-14	9	.135	.004	2.68	.016	.001	.47	.713	8,420	37.8	3,080	-160	-5.33	.091
05340500	WI-15	5	.028	.003	11.71	.003	.003	9.71	.011	225,000	13.9	101,000	7,800	7.69	.076
05369500	**WI-16**	**7**	**.084**	**.001**	**.57**	**.510**	**.001**	**.24**	**.792**	**737,000**	**31.6**	**545,000**	**-5,700**	**-1.05**	**.528**
05382000	WI-17	9	.124	.003	2.34	.013	.001	1.17	.198	259,000	48.0	180,000	-4,900	-2.73	.086
05407000	WI-20	7	.086	-.001	-.75	.635	-.001	-1.18	.397	709,000	26.3	713,000	-22,000	-3.15	.036
05420500	**IA-21**	**9**	**.183**	**-.001**	**-.07**	**.952**	**-.002**	**-.99**	**.361**	**9,230,000**	**41.6**	**10,800,000**	**-420,000**	**-3.86**	**.002**
05427718	WI-22	9	.143	.001	.02	.974	-.005	-3.34	.003	7,280	38.2	3,290	-160	-5.01	.001
05427948	WI-23	5	.377	-.020	-5.32	.000	-.021	-5.44	.000	3,700	83.5	1,070	-60.0	-5.62	.000
054310157	WI-24	8	.130	-.002	-1.69	.054	-.002	-1.60	.228	1,000	89.3	159	-2.20	-1.39	.677
05465500	**IA-25**	**3**	**.227**	**.014**	**6.22**	**.002**	**.009**	**4.11**	**.022**	**2,570,000**	**79.4**	**2,070,000**	**-65,000**	**-3.14**	**.251**
05474000	IA-26	9	.142	.013	9.26	.008	.005	3.53	.281	1,160,000	104	293,000	-14,000	-4.92	.218

Table 10. Summary of results for analyses of flow-adjusted trends, overall trends, and trends in loads for total phosphorus (TP, USGS parameter code 00665), 1993 to 2004.—Continued

[USGS, U.S. Geological Survey; ID, identification; mg/L, milligram per liter; mg/L/yr, milligram per liter per year; %/yr, percent per year; kg, kilogram; kg/km², kilogram per square kilometer; kg/yr, kilogram per year; NA, not available; NC, not calculated; sites in **bold** indicate long-term evaluation sites]

USGS site number	State-ID number (figure 1)	Water-quality model (table 3)	Concentration							Annual loads					
				Flow-adjusted trend			Overall trend								
			Reference concentration (mg/L)	Trend (mg/L/yr)	Trend (%/yr)	p-value	Trend (mg/L/yr)	Trend (%/yr)	p-value	Median annual load (kg)	Median yield (kg/km²)	Reference load (kg)	Trend (kg/yr)	Trend (%/yr)	p-value
Central part—continued															
05500000	MO-28	8	0.098	0.004	4.43	0.011	-0.001	-0.56	0.794	243,000	151	12,900	-840	-6.51	0.078
05514500	MO-29	7	.121	-.001	-.11	.972	-.001	-.34	.930	385,000	164	13,500	-110	-.84	.919
05525500	IL-30	8	.057	.003	5.11	.024	.002	3.11	.283	209,000	181	8,160	-75.0	-.91	.885
05531500	IL-31	9	.887	.009	1.00	.448	.012	1.39	.387	193,000	649	108,000	898	.83	.544
05532500	IL-32	9	.458	.012	2.68	.040	.019	4.08	.020	439,000	269	238,000	4,210	1.77	.215
05540275	IL-33	9	.076	-.001	-.21	.891	-.001	-1.07	.488	990	38.6	435	-18.0	-4.09	.139
05552500	IL-35	5	.214	.006	3.00	.641	.007	3.37	.599	555,000	84.3	399,000	-6,400	-1.60	.791
05586100	IL-37	9	.258	.022	8.38	.000	.022	8.66	.000	8,580,000	124	6,410,000	-57,000	-.89	.676
07018100	MO-39	5	.031	-.001	-.43	.805	-.001	-1.52	.340	71,100	37.4	14,600	-780	-5.35	.007
Eastern part															
03085000	PA-40	5	0.018	0.002	9.08	0.134	0.002	9.66	0.113	578,000	30.4	121,000	NC	14.85	0.063
03353637	IN-42	9	.024	.001	1.83	.236	.001	.26	.901	3,420	77.5	207	-5.40	-2.63	.566
04186500	OH-43	8	.132	-.002	-1.71	.383	-.002	-1.78	.398	81,900	95.3	10,100	-210	-2.07	.657
04193500	**OH-44**	**8**	**.135**	**.001**	**.03**	**.985**	**.001**	**.26**	**.903**	**1,930,000**	**118**	**255,000**	**2,160**	**.85**	**.854**
0422026250	NY-45	9	.390	-.011	-2.69	.000	-.013	-3.30	.000	3,240	124	2,470	7.48	.30	.870
04232034	NY-46	7	.070	-.002	-2.35	.046	-.002	-2.71	.033	3,530	34.8	1,850	-62.0	-3.33	.083
0423204920	NY-47	8	.090	-.001	-1.56	.052	-.002	-1.93	.027	1,210	67.1	425	-15.0	-3.54	.093
0423205025	NY-48	8	.068	.004	5.27	.000	.003	4.17	.000	14,200	38.2	6,570	83.3	1.27	.585

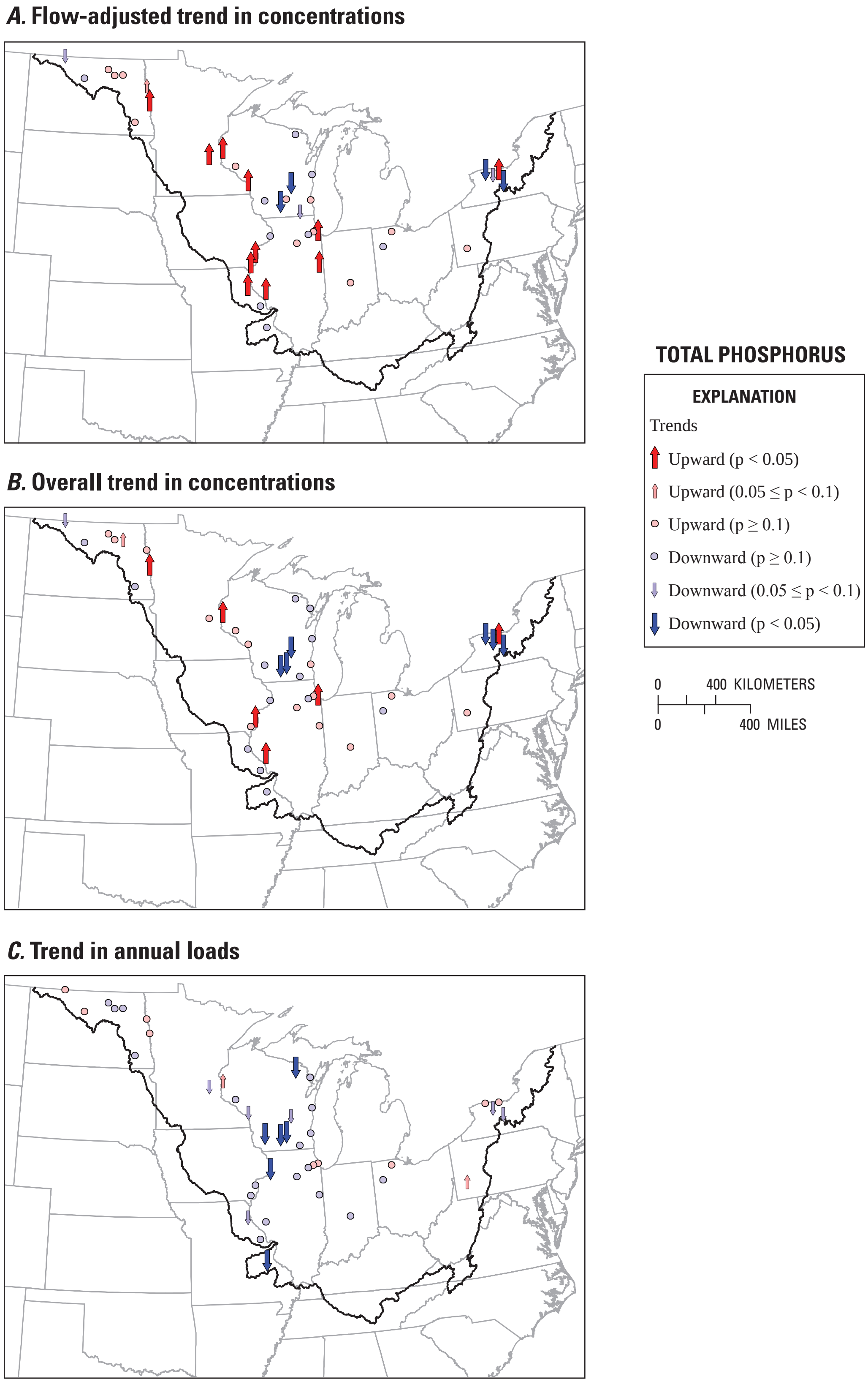


Figure 22. Trends in total phosphorus from 1993 to 2004 for ***A***, flow-adjusted concentrations, ***B***, overall concentrations, and ***C***, annual loads throughout the study area.

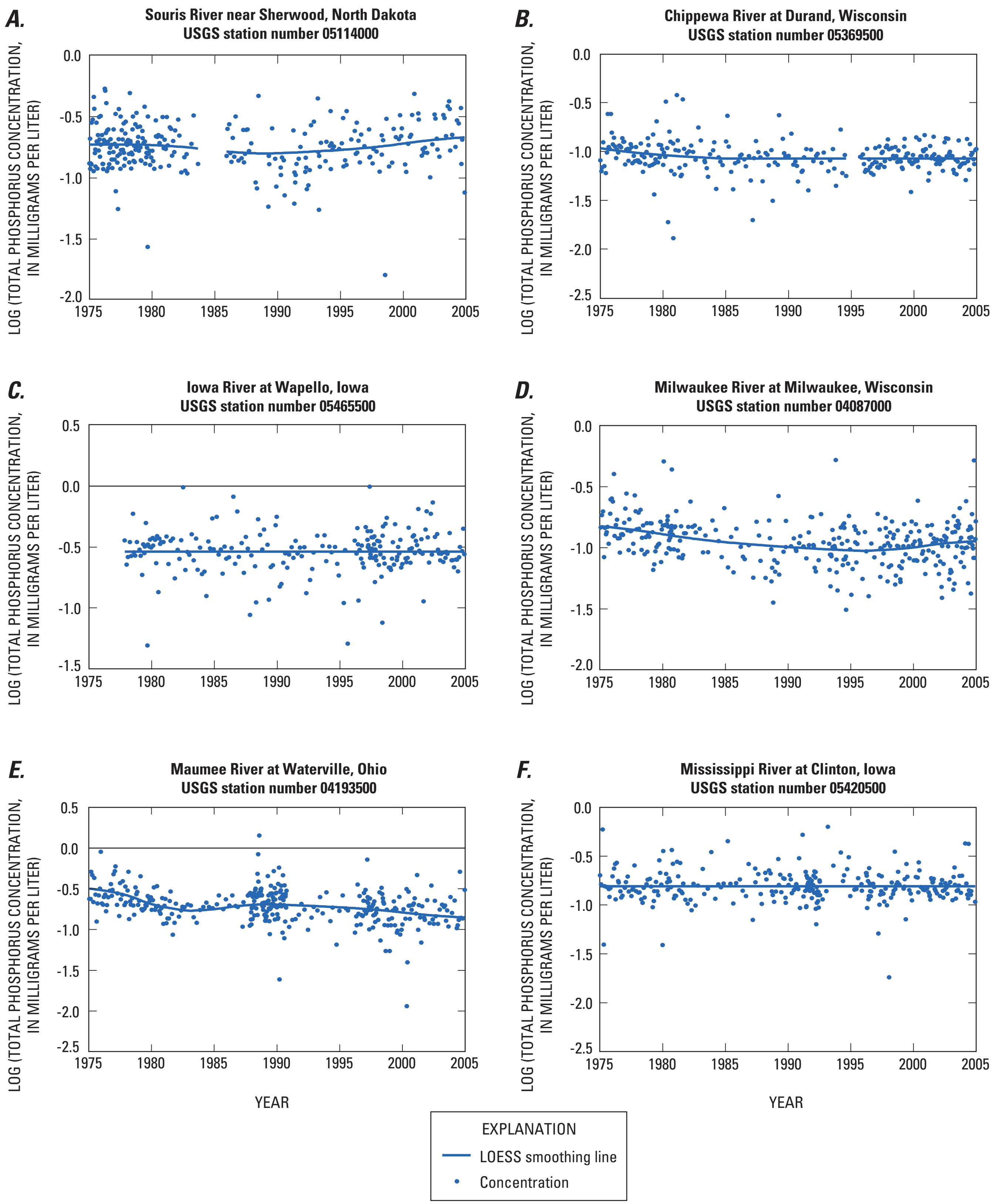


Figure 23. Long-term flow-adjusted trends in total phosphorus concentrations from 1975 to 2004 at six selected sites in the study area.

Table 11. Summary of flow-adjusted trends in total phosphorus concentrations (TP, USGS parameter code 00665) estimated by QWTREND for long-term evaluation sites, 1975 to 2004.

[USGS, U.S. Geological Survey; mg/L, milligram per liter]

USGS site number	Trend period	Description of flow-adjusted concentration
05114000	January 1975–December 1978	Constant value of 0.19 mg/L
	January 1979–December 1988	Downward trend from 0.19 to 0.16 mg/L
	January 1989–December 2004	Upward trend from 0.16 to 0.22 mg/L
05369500	January 1975–December 1984	Downward trend from 0.11 to 0.09 mg/L
	January 1985–December 2004	Constant value of 0.09 mg/L
05465500	November 1977–December 2004	Constant value of 0.29 mg/L
04087000	January 1975–December 1995	Downward trend from 0.15 to 0.09 mg/L
	January 1996–December 2004	Upward trend from 0.09 to 0.11 mg/L
04193500	January 1975–December 1982	Downward trend from 0.32 to 0.17 mg/L
	January 1983–December 1988	Upward trend from 0.17 to 0.21 mg/L
	January 1989–December 2004	Downward trend from 0.21 to 0.14 mg/L
05420500	January 1975–December 2004	Constant value of 0.16 mg/L

Trends in Total Phosphorus Loads

From 1993 to 2004, TP loads at 30 of the 41 sites that had sufficient data for analysis indicated a tendency toward decreasing loads (table 10; fig. 22***C***). Twelve of these sites, all within the central and eastern parts of the study area, had significant ($p < 0.10$) downward trends in loads. Most of these sites had significant downward trends in streamflow. The trends at only 2 of the 11 sites that had upward trends were statistically significant ($p < 0.10$). A comparison of the trends in loads and FA trends in concentrations indicates that most of the trends in loads were driven primarily by streamflow rather than by changes in concentrations, except in southern Wisconsin where trends in both streamflow and concentration were significantly downward.

Annual loads at the six long-term sites from 1975 to 2003 were used to put the short-term trends in TP loads into a longer term context (fig. 24). The long-term changes at all six sites resembled the long-term trend in streamflow (fig. 13), which again indicates the importance of changes in streamflow in driving the trends in loads. Total phosphorus loads at the Souris River, N. Dak., probably were affected by the construction of dams within the basin during the 1990s. Loads at the Souris River, N. Dak., were more variable from 1975 to 1983 and from 1996 to 2004, producing larger average loads during those periods than from 1984 to 1995. Loads at the Milwaukee River, Wis., showed a decrease between 1982 and 1990. Loads at the remaining four sites varied slightly and showed no sustained trends.

Dissolved Phosphorus

General Patterns in Dissolved Phosphorus Concentrations and Yields

Median midmonthly dissolved phosphorus (DP) concentrations at the 35 trend sites that had sufficient data for analysis ranged from 0.020 to 1.29 mg/L, based on data from 1993 to 2004. The mean and median values of the median DP concentrations at these trend sites were 0.149 and 0.075 mg/L, respectively. The highest DP concentrations were in central Illinois and eastern North Dakota; whereas the lowest concentrations (<0.1 mg/L) were in all three parts of the study area. The highest concentrations were in areas with intensive agriculture, and the lowest concentrations were in forested areas.

Median annual DP yields ranged from 2.39 to 496 kg/km^2, based on data from 1993 to 2003 (table 12). The mean and median values of the median DP yields at these trend sites were 48.3 and 19.0 kg/km^2, respectively. Five of the six sites with the highest yields were in Illinois, and the remaining high-yield site was in North Dakota. The lowest yields were from sites throughout the north-central and eastern parts of the study area; especially low yields were from two sites in northern Wisconsin and one site in New York.

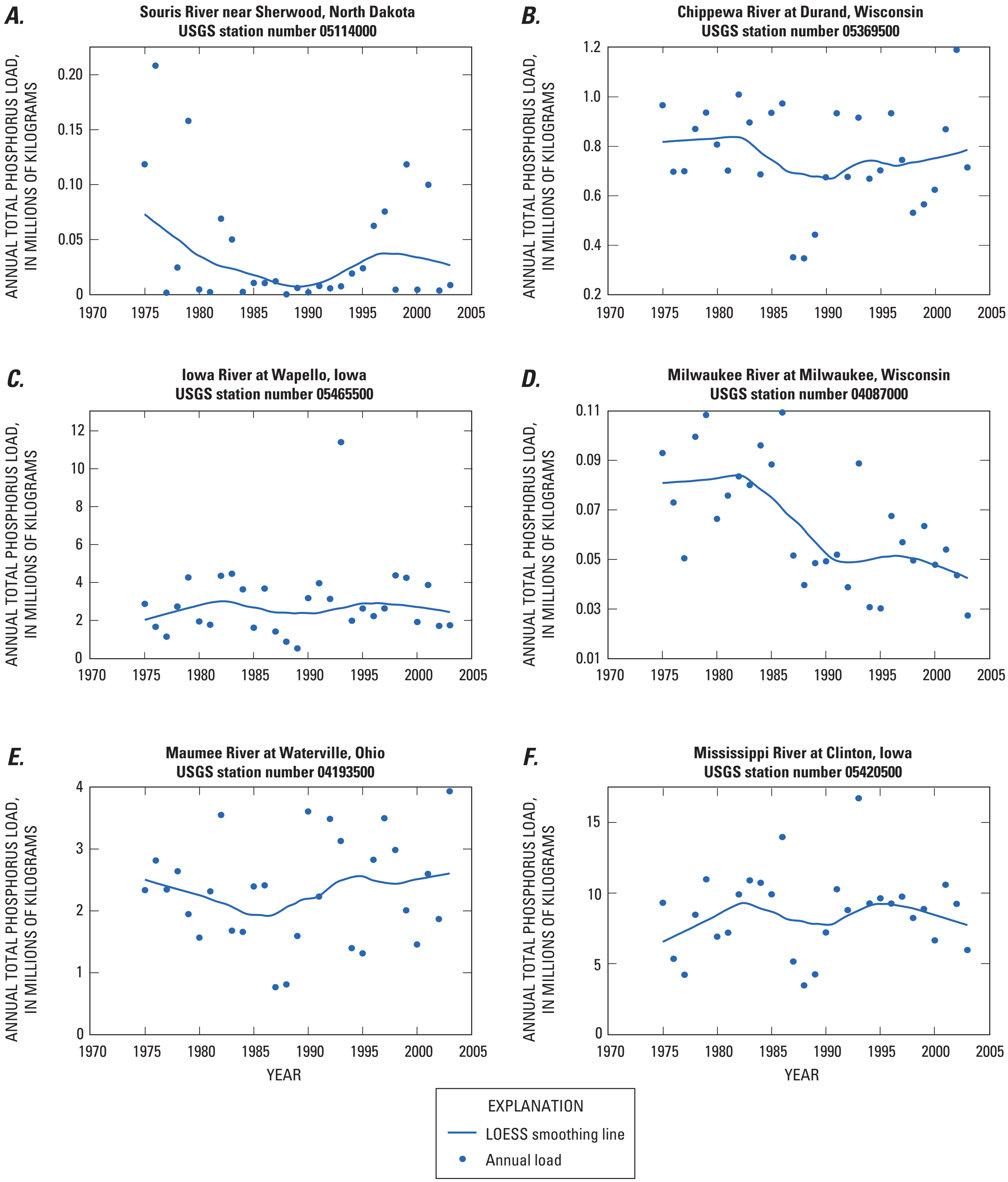


Figure 24. Long-term trends in total phosphorus loads from 1975 to 2003 at six selected sites in the study area.

Table 12. Summary of results for analyses of flow-adjusted trends, overall trends, and trends in loads for dissolved phosphorus (DP, USGS parameter code 00666), 1993 to 2004.

[USGS, U.S. Geological Survey; ID, identification; mg/L, milligram per liter; mg/L/yr, milligram per liter per year; %/yr, percent per year; kg, kilogram; kg/km^2, kilogram per square kilometer; kg/yr, kilogram per year; NC, not calculated; sites in **bold** indicate long-term evaluation sites]

USGS site number	State-ID number (figure 1)	Water-quality model (table 3)	Concentration							Annual loads					
				Flow-adjusted trend			Overall trend								
			Reference concentration (mg/L)	Trend (mg/L/yr)	Trend (%/yr)	p-value	Trend (mg/L/yr)	Trend (%/yr)	p-value	Median annual load (kg)	Median yield (kg/km^2)	Reference load (kg)	Trend (kg/yr)	Trend (%/yr)	p-value
Western part															
05056100	ND-1	7	0.276	-0.007	-2.54	0.345	-0.007	-2.69	0.313	NA	NA	2,580	-157	-6.10	0.299
05056200	ND-2	3	.283	.003	1.09	.743	.004	1.40	.675	NA	NA	4,710	-312	-6.62	.226
05056239	ND-3	3	.146	-.003	-1.75	.699	-.005	-3.63	.394	NA	NA	2,520	-190	-7.54	.160
05058700	ND-4	8	.130	.007	5.33	.375	.004	3.20	.563	60,000	9.30	29,500	-58.1	-.20	.976
05082500	ND-6	3	.097	-.002	-2.14	.453	-.003	-2.90	.291	734,000	74.5	379,000	-17,800	-4.69	.136
Central part															
04063700	WI-9	7	0.026	-0.002	-7.70	0.000	-0.002	-7.73	0.000	860	2.39	2,170	-175	-8.06	0.000
04067500	WI-10	13	.002	.001	.00	.977	.001	.00	.977	30,400	2.99	4,750	-1.18	-.02	.977
04085427	WI-12	3	.077	-.001	-1.77	.605	-.002	-2.19	.498	23,200	17.0	10,100	-428	-4.25	.245
04087000	**WI-13**	**7**	**.043**	**-.001**	**-1.36**	**.402**	**-.001**	**-1.62**	**.317**	**25,300**	**14.0**	**13,900**	**-387**	**-2.79**	**.228**
05287890	MN-14	9	.056	.002	3.52	.025	.001	.96	.568	5,270	23.6	1,280	-65.9	-5.15	.121
05340500	WI-15	5	.016	-.001	-.14	.980	.001	-1.18	.830	116,000	7.15	62,500	-1,340	-2.15	.693
05369500	**WI-16**	**8**	**.039**	**-.001**	**-3.02**	**.142**	**-.001**	**-3.43**	**.091**	**293,000**	**12.6**	**250,000**	**-10,500**	**-4.19**	**.068**
05382000	WI-17	8	.074	-.001	-.77	.419	-.001	-1.99	.035	106,000	19.7	107,000	-4,940	-4.63	.002
05407000	WI-20	3	.038	-.002	-6.35	.036	-.003	-6.78	.008	229,000	8.51	314,000	-22,900	-7.30	.002
05420500	**IA-21**	**7**	**.077**	**-.003**	**-3.29**	**.025**	**-.003**	**-3.76**	**.003**	**4,050,000**	**18.3**	**4,500,000**	**-254,000**	**-5.59**	**.000**
05427718	WI-22	8	.097	-.001	-.28	.836	-.003	-2.81	.048	3,190	16.7	2,220	-104	-4.69	.005
05427948	WI-23	7	.183	-.013	-7.25	.000	-.013	-7.29	.000	1,400	31.6	520	-38.3	-7.38	.000
05465500	**IA-25**	**9**	**.055**	**.001**	**1.04**	**.726**	**-.002**	**-2.97**	**.226**	**944,000**	**29.2**	**498,000**	**-31,000**	**-6.23**	**.027**
05474000	IA-26	9	.100	-.003	-2.51	.248	-.006	-5.85	.012	324,000	29.0	205,000	-16,000	-7.81	.006
05500000	MO-28	7	.053	-.001	-.36	.914	-.001	-2.76	.375	52,200	32.5	4,080	-376	-9.23	.035
05514500	MO-29	5	.071	-.003	-4.80	.095	-.004	-5.00	.195	73,700	31.5	7,870	-418	-5.31	.472
05525500	IL-30	9	.026	.001	.58	.817	-.001	-1.15	.712	75,700	65.5	3,630	-135	-3.71	.512
05531500	IL-31	9	.785	.025	3.19	.006	.029	3.70	.018	148,000	496	95,600	2,870	3.00	.012
05532500	IL-32	3	.490	.016	3.29	.011	.025	5.03	.009	323,000	198	255,000	6,490	2.55	.060

Table 12. Summary of results for analyses of flow-adjusted trends, overall trends, and trends in loads for dissolved phosphorus (DP, USGS parameter code 00666), 1993 to 2004.—Continued

[USGS, U.S. Geological Survey; ID, identification; mg/L, milligram per liter; mg/L/yr, milligram per liter per year; %/yr, percent per year; kg, kilogram; kg/km², kilogram per square kilometer; kg/yr, kilogram per year; NC, not calculated; sites in **bold** indicate long-term evaluation sites]

USGS site number	State-ID number (figure 1)	Water-quality model (table 3)	Concentration							Annual loads					
				Flow-adjusted trend			Overall trend								
			Reference concentration (mg/L)	Trend (mg/L/yr)	Trend (%/yr)	p-value	Trend (mg/L/yr)	Trend (%/yr)	p-value	Median annual load (kg)	Median yield (kg/km²)	Reference load (kg)	Trend (kg/yr)	Trend (%/yr)	p-value
Central part—continued															
05540275	IL-33	9	0.039	0.001	0.98	0.706	0.001	0.69	0.786	411	16.0	227	-6.90	-3.04	0.331
05543500	IL-34	8	.238	.018	7.59	.084	.031	12.98	.009	3,970,000	186	2,450,000	130,000	5.30	.201
05586100	IL-37	7	.158	.005	3.00	.002	.008	5.11	.000	4,110,000	59.3	3,930,000	-97,400	-2.48	.101
Eastern part															
03085000	PA-40	5	0.013	-0.001	-1.46	0.073	-0.001	-1.61	0.059	110,000	5.77	89,500	NC	-10.14	0.091
03353637	IN-42	9	.023	-.002	-6.65	.000	-.002	-6.78	.000	634	14.4	202	-15.0	-7.37	.001
04186500	OH-43	7	.074	-.003	-3.61	.056	-.003	-3.66	.062	26,900	31.3	5,360	-210	-3.87	.338
04193500	**OH-44**	**7**	**.057**	**-.002**	**-4.23**	**.022**	**-.002**	**-4.10**	**.048**	**486,000**	**29.6**	**108,000**	**-4,100**	**-3.79**	**.302**
0422026250	NY-45	9	.268	-.012	-4.34	.000	-.014	-5.18	.000	1,090	41.6	1,700	-49.0	-2.86	.021
04232034	NY-46	7	.008	-.001	.00	.464	-.001	.00	.464	345	3.40	208	-1.90	-.92	.464
0423204920	NY-47	8	.022	-.001	-4.62	.002	-.001	-4.82	.001	228	12.6	106	-6.20	-5.86	.004
0423205025	NY-48	9	.018	-.001	-2.21	.027	-.001	-2.41	.015	2,440	6.59	1,740	-69.0	-3.96	.017

Trends in Dissolved Phosphorus Concentrations

The FA trends in DP concentrations from 1993 to 2004 showed mixed regional tendencies; however, DP concentrations consistently decreased at all of the sites in the eastern part of the study area (table 12; fig. 25***A***). Of the 35 sites that had sufficient streamflow and DP concentration data for analysis, 23 sites had downward trends in concentrations. Twelve of these sites had significant downward trends ($p < 0.10$). Seven sites in the eastern part of the study area had downward trends. Five sites, all in the central part of the study area, had significant upward trends ($p < 0.10$). Two sites had almost no FA trends.

When the OA trends in DP concentrations from 1993 to 2004 were examined, spatial trends followed the same general decreasing concentration pattern in the eastern areas with mixed tendencies in the central and western parts of the study area. Several differences between the FA and OA trend analyses occurred at specific sites in the central area, where many more sites had significantly downward OA trends than downward FA trends. These differences in the results of the FA and OA trend analyses are attributable to the significant trends in streamflow. Of the 35 sites that had sufficient streamflow and DP data for analysis (table 12; fig. 25***B***), 25 sites had downward trends, 8 sites had upward trends, and 2 sites had almost no trend in concentrations. Of these sites, 15 sites had significant downward trends, and 4 sites (all in Illinois) had significant upward trends ($p < 0.10$).

Trends in Dissolved Phosphorus Loads

From 1993 to 2004, dissolved phosphorus loads at 32 of the 35 sites that had sufficient data for analysis indicated a tendency toward decreasing loads (table 12; fig. 25***C***); trends at 15 of these sites were significant ($p < 0.10$). Two of the three sites, all located in northern Illinois, with upward trends in loads had significant upward trends.

Total Suspended Materials

General Patterns in Concentrations and Yields of Total Suspended Materials

Median midmonthly total SMAT concentrations in the study area ranged from 0.7 to 5,240 mg/L, based on data for all of the sites from 1970 to 2004 (fig. 26***A***). The overall median and mean SMAT concentrations were 24.0 and 112 mg/L, respectively. The highest concentrations were in the western, southwestern, and south-central parts of the study area. The lowest concentrations were mostly in northern areas of Wisconsin and Michigan and in the northeastern part of the study area. The highest concentrations were in areas with intensive row-crop agriculture, and the lowest concentrations were in forested areas. Median midmonthly SMAT concentrations ranged from 4.0 to 424 mg/L at the 42 trend sites that had sufficient data for analysis, based on data from 1993 to 2004. The mean and median values of the median SMAT concentrations at these sites were 59.1 and 37.5 mg/L, respectively. The distribution in concentrations of these trend sites was consistent with that shown in fig. 26***A***.

Median annual SMAT yields from 1970 to 2004 ranged from 22.0 to 3,370,000 kg/km^2. The overall mean and median yields based on all of the data in the study area from 1970 to 2004 were 85,100 and 35,400 and kg/km^2, respectively. The highest yields were from basins throughout the southern one-half of the study area, and the lowest yields were from basins throughout the northern one-half of the study area (fig. 26***B***). Similar to TP and TN, the major difference in the distributions in SMAT between concentrations and yields was in the northwestern part of the study area, which had high concentrations but low yields, primarily because of low mean annual runoff (fig. 3). Median annual SMAT yields at trend sites, based on data from 1993 to 2004, ranged from 188 to 939,000 kg/km^2 (table 13). The distribution in yields was consistent with that in fig. 26***B***. The mean and median yields based only on data from the trend sites were 60,100 and 25,300 kg/km^2, respectively.

Trends in Total Suspended-Material Concentrations

The FA trends in SMAT concentrations from 1993 to 2004 showed distinct spatial patterns of tendencies toward increasing concentrations in Illinois, and tendencies toward decreasing concentrations throughout most of Wisconsin, Iowa, and the eastern part of the study area (fig. 27***A***). Of the 42 sites that had sufficient streamflow and SMAT concentration data for analysis (table 13), 25 sites had downward trends, and 17 sites had upward trends in concentrations. Of these sites, 12 sites had significant downward trends, and 5 sites had significant upward trends ($p < 0.10$).

The spatial pattern in the OA trends in SMAT concentrations generally was the same as that for the FA trends (fig. 27***B***); however, several of the sites with nonsignificant downward trends, especially in Iowa, had significant downward OA trends ($p < 0.10$). Of the 42 sites that had sufficient streamflow and SMAT concentration data for analysis (table 13), 30 sites had downward trends, and 12 sites had upward trends in concentrations. Of these sites, 18 sites had significant downward trends, and 5 sites had significant upward trends. The sites with significant increases in concentrations were in Illinois and North Dakota.

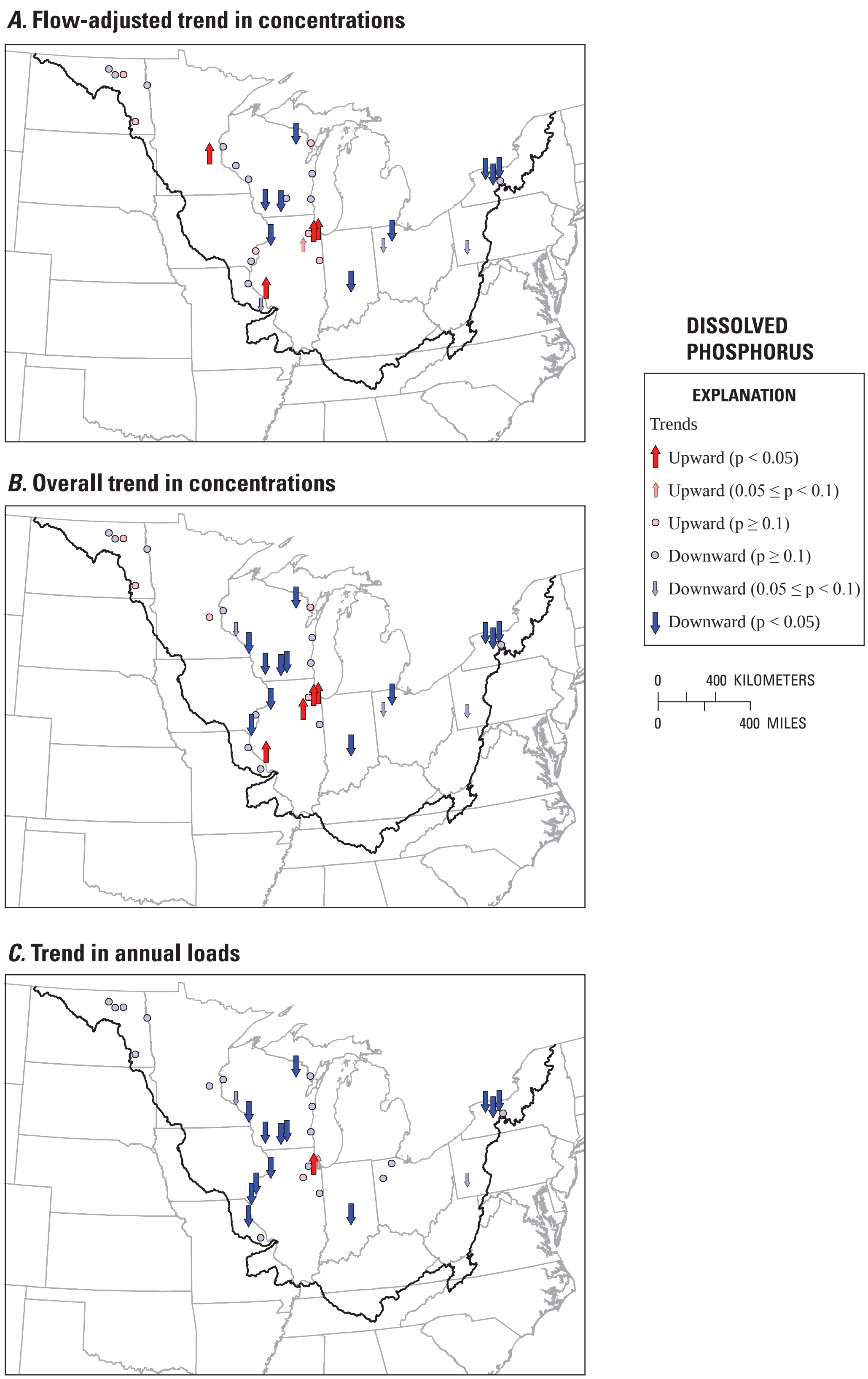


Figure 25. Trends in dissolved phosphorus from 1993 to 2004 for ***A***, flow-adjusted concentrations, ***B***, overall concentrations, and ***C***, annual loads throughout the study area.

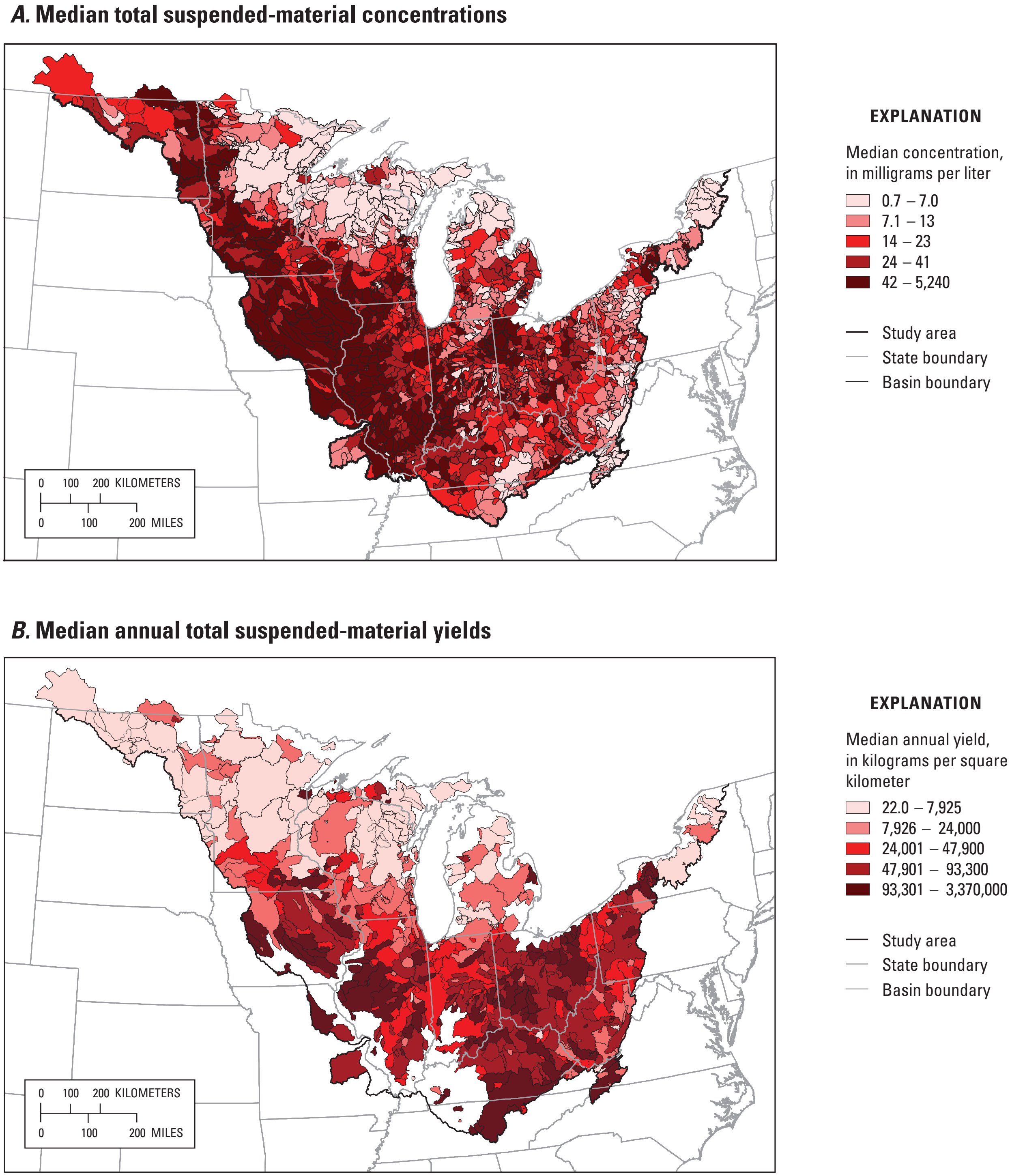


Figure 26. Spatial patterns from 1970 to 2004 in ***A***, median total suspended-material concentrations and ***B***, median annual total suspended-material yields throughout the study area.

Table 13. Summary of results for analyses of flow-adjusted trends, overall trends, and trends in loads for total suspended material (SMAT, USGS parameter codes 80154 and 00530 combined), 1993 to 2004.

[USGS, U.S. Geological Survey; ID, identification; mg/L, milligram per liter; mg/L/yr, milligram per liter per year; %/yr, percent per year; kg, kilogram; kg/km², kilogram per square kilometer; kg/yr, kilogram per year; NA, not available; NC, not calculated; sites in **bold** indicate long-term evaluation sites. Total suspended material, or SMAT, is total suspended sediment combined with total suspended solids, USGS parameter codes 80154 and 00530, respectively.]

USGS site number	State-ID number (figure 1)	Water-quality model (table 3)	Concentration							Annual loads					
			Reference concentration (mg/L)	Flow-adjusted trend			Overall trend								
				Trend (mg/L/yr)	Trend (%/yr)	p-value	Trend (mg/L/yr)	Trend (%/yr)	p-value	Median annual load (kg)	Median yield (kg/km²)	Reference load (kg)	Trend (kg/yr)	Trend (%/yr)	p-value
Western part															
05082500	ND-6	5	70.3	7.148	10.17	0.675	2.750	3.91	0.857	NA	NA	267,000,000	1,280,000	0.48	0.982
05114000	**ND-7**	**9**	**5.0**	**.187**	**3.75**	**.223**	**.427**	**8.56**	**.077**	**1,480,000**	**188**	**26,200**	**NC**	**94.14**	**.052**
05120000	ND-8	9	5.6	.092	1.65	.593	.172	3.07	.420	2,770,000	243	207,000	NC	10.25	.410
Central part															
04063700	WI-9	7	3.8	0.208	5.48	0.132	0.156	4.12	0.221	956,000	2,660	316,000	-1,100	-0.33	0.915
04067500	WI-10	8	5.0	-.149	-2.97	.164	-.150	-2.98	.182	15,800,000	1,560	12,000,000	-360,000	-3.01	.266
04073468	WI-11	9	21.9	.414	1.89	.081	.369	1.68	.128	897,000	6,470	586,000	-6,200	-1.07	.708
04085427	WI-12	3	33.7	-2.134	-6.32	.080	-2.085	-6.18	.078	5,100,000	3,740	4,460,000	-310,000	-7.00	.032
04087000	**WI-13**	**7**	**30.3**	**.205**	**.68**	**.667**	**.057**	**.19**	**.907**	**22,400,000**	**12,400**	**9,790,000**	**-130,000**	**-1.28**	**.619**
05287890	MN-14	7	6.9	.035	.50	.812	-.172	-2.49	.240	1,010,000	4,540	156,000	-9,900	-6.39	.046
05340500	WI-15	8	5.3	.028	.53	.901	-.020	-.38	.927	56,400,000	3,490	19,400,000	-280,000	-1.46	.731
05369500	**WI-16**	**7**	**8.8**	**-.256**	**-2.91**	**.072**	**-.285**	**-3.24**	**.042**	**90,700,000**	**3,880**	**57,000,000**	**-2,300,000**	**-4.03**	**.038**
05382000	WI-17	9	26.3	-1.221	-4.64	.001	-1.396	-5.31	.000	41,000,000	7,600	38,100,000	-2,500,000	-6.63	.000
05389400	IA-18	9	83.9	-.074	-.09	.977	-1.581	-1.88	.410	1,100,000	12,500	2,080,000	-110,000	-5.35	.007
05389500	IA-19	3	36.2	-1.051	-2.90	.179	-1.311	-3.62	.063	2,140,000,000	12,200	1,510,000,000	-82,000,000	-5.42	.004
05407000	WI-20	7	24.5	-1.067	-4.35	.005	-1.110	-4.53	.001	175,000,000	6,490	204,000,000	-12,000,000	-5.63	.000
05420500	**IA-21**	**8**	**54.3**	**-1.534**	**-2.83**	**.109**	**-2.478**	**-4.57**	**.003**	**3,950,000,000**	**17,800**	**3,190,000,000**	**-190,000,000**	**-6.10**	**.000**
05427718	WI-22	9	40.4	-1.257	-3.11	.000	-1.942	-4.80	.000	2,210,000	11,600	929,000	-56,000	-6.00	.000
05427948	WI-23	8	33.7	-1.006	-2.98	.000	-1.046	-3.10	.000	1,410,000	31,900	95,800	-3,300	-3.43	.024
054310157	WI-24	8	48.1	-1.422	-2.95	.007	-1.410	-2.93	.010	447,000	39,800	59,400	-1,600	-2.76	.280
05465500	**IA-25**	**8**	**181.0**	**-2.893**	**-1.60**	**.401**	**-9.416**	**-5.20**	**.008**	**1,860,000,000**	**57,300**	**1,640,000,000**	**-120,000,000**	**-7.21**	**.004**
05474000	IA-26	7	207.0	-4.453	-2.15	.267	-12.022	-5.81	.007	1,420,000,000	127,000	417,000,000	-33,000,000	-7.80	.006
05481650	IA-27	3	52.9	-1.361	-2.57	.232	-2.444	-4.62	.007	213,000,000	14,100	166,000,000	-13,000,000	-7.83	.000
05500000	MO-28	8	28.8	-.893	-3.10	.368	-1.794	-6.22	.030	388,000,000	242,000	3,680,000	-290,000	-7.96	.012

Table 13. Summary of results for analyses of flow-adjusted trends, overall trends, and trends in loads for total suspended material (SMAT, USGS parameter codes 80154 and 00530 combined), 1993 to 2004.—Continued

[USGS, U.S. Geological Survey; ID, identification; mg/L, milligram per liter; mg/L/yr, milligram per liter per year; %/yr, percent per year; kg, kilogram; kg/km^2, kilogram per square kilometer; kg/yr, kilogram per year; NA, not available; NC, not calculated; sites in **bold** indicate long-term evaluation sites. Total suspended material, or SMAT, is total suspended sediment combined with total suspended solids, USGS parameter codes 80154 and 00530, respectively.]

			Concentration							Annual loads					
				Flow-adjusted trend			Overall trend								
USGS site number	State-ID number (figure 1)	Water-quality model (table 3)	Reference concentration (mg/L)	Trend (mg/L/yr)	Trend (%/yr)	p-value	Trend (mg/L/yr)	Trend (%/yr)	p-value	Median annual load (kg)	Median yield (kg/km^2)	Reference load (kg)	Trend (kg/yr)	Trend (%/yr)	p-value
Central part—continued															
05525500	IL-30	9	17.4	6.025	34.60	0.000	5.417	31.11	0.000	100,000,000	87,000	2,510,000	438,000	17.49	0.111
05531500	IL-31	9	16.6	6.202	37.39	.000	6.071	36.60	.000	22,300,000	74,800	2,010,000	NC	34.07	.000
05532500	IL-32	9	16.4	5.681	34.62	.000	4.932	30.06	.000	83,500,000	51,100	8,450,000	NC	23.02	.001
05540275	IL-33	7	10.5	.328	3.13	.260	.063	.61	.827	328,000	12,800	59,700	-1,800	-3.09	.422
05543500	IL-34	8	31.0	.783	2.52	.320	-.199	-.64	.785	542,000,000	25,300	296,000,000	-13,000,000	-4.34	.081
05552500	IL-35	7	37.9	.214	.56	.872	-.960	-2.53	.406	164,000,000	24,900	71,800,000	-3,800,000	-5.29	.087
05570000	IL-36	7	85.0	10.015	11.78	.250	2.602	3.06	.696	489,000,000	115,000	80,600,000	-4,000,000	-4.91	.419
05586100	IL-37	9	81.5	12.415	15.23	.000	8.016	9.83	.000	3,490,000,000	50,300	2,020,000,000	-7,300,000	-.36	.908
07014500	MO-38	7	6.5	.016	.25	.959	-.307	-4.71	.159	46,900,000	12,300	7,280,000	-510,000	-6.95	.015
07018100	MO-39	5	4.1	-.120	-2.96	.558	-.190	-4.68	.248	52,200,000	27,400	1,890,000	-130,000	-6.79	.041
Eastern part															
03085000	PA-40	9	10.7	-0.655	-6.14	0.005	-0.601	-5.63	0.022	609,000,000	32,000	70,700,000	-3,400,000	-4.75	0.173
03353637	IN-42	8	21.2	-.371	-1.75	.361	-.581	-2.74	.206	3,780,000	85,800	178,000	-8,300	-4.67	.236
04186500	OH-43	7	47.4	-3.272	-6.90	.001	-3.301	-6.96	.002	32,500,000	37,700	3,550,000	-250,000	-7.10	.081
04193500	**OH-44**	**9**	**32.9**	**-1.322**	**-4.02**	**.008**	**-1.268**	**-3.86**	**.043**	**1,040,000,000**	**63,600**	**61,900,000**	**-2,200,000**	**-3.54**	**.347**
0422026250	NY-45	7	91.4	-5.705	-6.24	.000	-5.109	-5.59	.000	1,600,000	61,100	572,000	-20,000	-3.56	.212
04232034	NY-46	7	132.1	-7.685	-5.82	.000	-7.818	-5.92	.000	6,020,000	59,300	3,470,000	-220,000	-6.20	.000
0423204920	NY-47	7	72.4	-.850	-1.17	.603	-1.178	-1.63	.472	978,000	54,200	341,000	-11,000	-3.30	.252
0423205025	NY-48	9	45.6	-.585	-1.28	.429	-.791	-1.73	.285	11,900,000	32,000	4,380,000	-150,000	-3.43	.111
04237946	NY-49	9	984.6	-13.718	-1.39	.288	-14.079	-1.43	.280	778,000	939,000	608,000	-7,100	-1.17	.518

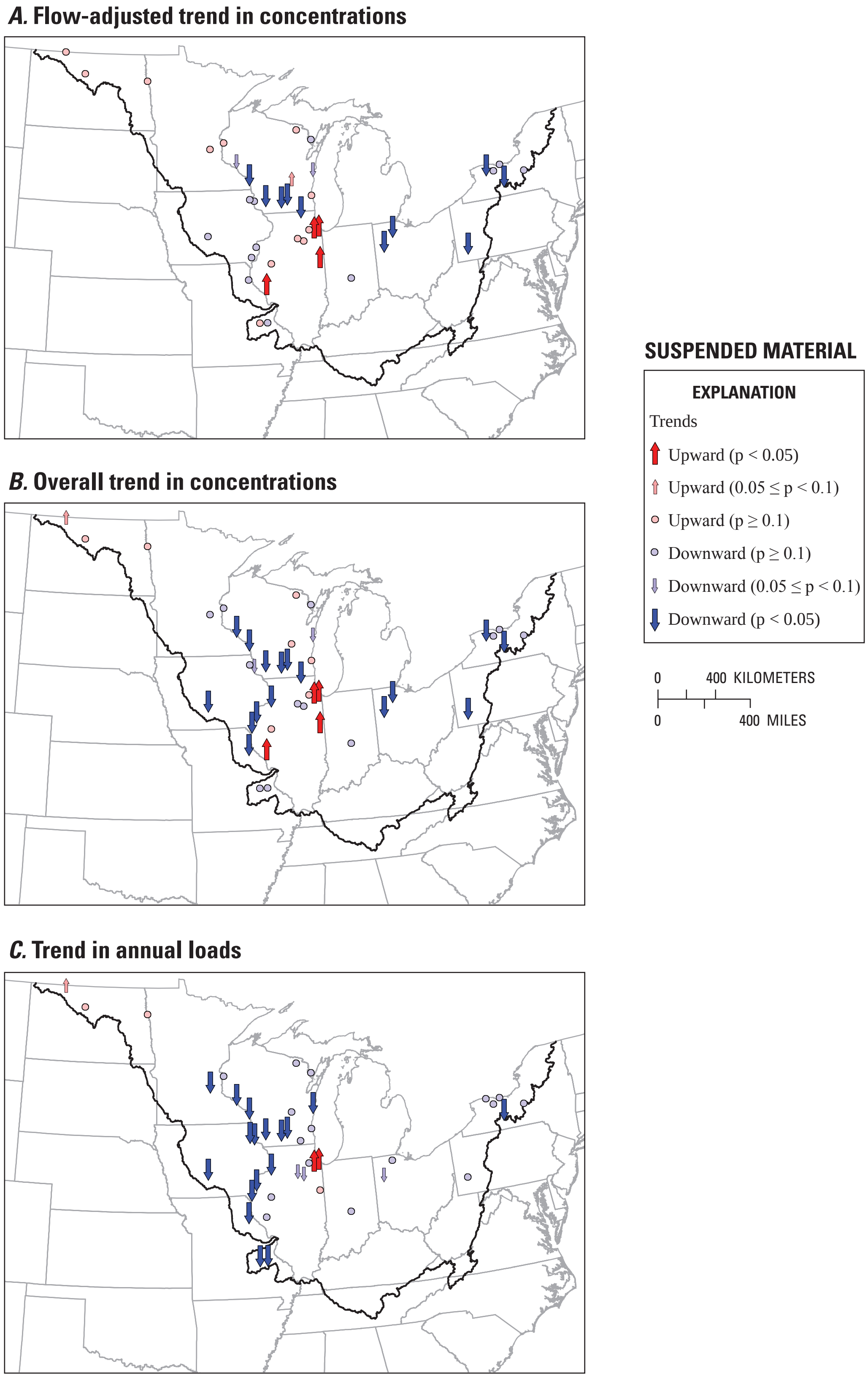


Figure 27. Trends in total suspended materials from 1993 to 2004 for ***A***, flow-adjusted concentrations, ***B***, overall concentrations, and ***C***, annual loads throughout the study area.

To place the 1993–2004 trends in SMAT concentrations into a longer term context, FACs were computed from 1975 to 2004 for the six sites that had sufficient data and examined for trends with QWTREND (table 14; fig. 28). Only one site, the Milwaukee River (04087000), had an upward trend over the entire period from 1975 through 2004. The Chippewa River had an upward trend from 1975 through 1981, a downward trend from 1982 through 2001, and no trend from 2002 through 2004. The Souris (05114000) and Maumee (05420500) Rivers had no trend through the early part of the period and downward trends in the later part, beginning in 1996. Myers and others (2000) reported a relatively small, but significant, downward trend in FACs from 1970 to 1998. The Iowa River (05465500) had a downward trend from 1982 through 2004, and the Mississippi River (05420500) showed downward trends throughout the period from 1975 though 2004. Three sites (05465500, 04193500, and 05420500) had FACs in 2004 that were less than one-half of the FACs in 1975 (table 14).

Trends in Total Suspended-Material Loads

From 1993 to 2004, SMAT loads indicated a tendency toward decreasing loads at 36 sites and a tendency toward increasing loads at 6 sites (table 13; fig. 27*C*). Downward trends in loads at 20 of the sites, throughout most of the study area, were significant, and upward trends at 3 sites in northern Illinois and North Dakota were significant ($p < 0.10$). Once again the pattern in the trends in loads resembles that for streamflow (fig. 12).

Annual SMAT loads at the six long-term sites from 1975 to 2003 can be used to put the short-term trends in loads into a longer term context (fig. 29). The long-term changes at all six sites resembled the long-term trend in streamflow (fig. 12), which again indicates the importance of changes in streamflow in driving the trends in loads. A combination of total suspended sediment (SS) and total suspended solid data was used to estimate changes in the long-term loads at all of the sites except the Iowa and Maumee Rivers, which were estimated on the basis of only SS data. This combination of data from two different types of analysis could in itself induce long-term trends in the loads; however, for these sites, the major changes appear to be associated with changes in streamflow.

The SMAT loads at the Souris River, N.Dak., decreased after a very high streamflow period (fig. 13) in the mid-1970s until approximately 1987, when loads remained relatively constant until 2003; however, very few SMAT samples were collected during the early years of the record, so the apparent large downward trend in loads may be partially attributable to sampling bias. For example, only two suspended-material samples were collected in 1975, and only five samples were collected in 1976—the two highest loading years (fig. 29).

Loads at the Chippewa River at Durand, Wisc., increased from 1975 to 1982 and then decreased at a fairly constant rate until 2003. Loads at the Milwaukee, Iowa, and Mississippi Rivers generally increased from 1975 to about 1982, then generally decreased until about 1990, and then remained approximately constant during a period of irregular wet or dry years until 2003. It is probable that both streamflow and the associated transport of suspended materials in these rivers were affected by the presence of the many dams upstream from the sampling sites. The SMAT loads at the Maumee River at Waterville, Ohio, decreased from 1975 to about 1987, and then generally increased through 1994, then decreased again during a period of wet and dry years until 2003.

Table 14. Summary of flow-adjusted trends in total suspended-material concentrations estimated by QWTREND for long-term evaluation sites, 1975 to 2004.

[USGS, U.S. Geological Survey; mg/L, milligram per liter. Total suspended material, or SMAT, is total suspended sediment combined with total suspended solids, USGS parameter codes 80154 and 00530, respectively]

USGS site number	Trend period	Description of flow-adjusted concentration
05114000	January 1975–December 1995	Constant value of 13.3 mg/L
	January 1996–December 2004	Downward trend from 13.3 to 7.0 mg/L
05369500	January 1975–December 1981	Upward trend from 8.3 to 22.5 mg/L
	January 1982–December 2001	Downward trend from 22.5 to 7.1 mg/L
	January 2002–December 2004	Constant value of 7.1 mg/L
05465500	February 1978–December 1981	Constant value of 149 mg/L
	January 1982–December 2004	Downward trend from 149 to 67 mg/L
04087000	January 1975–December 2004	Upward trend from 19.8 to 28.9 mg/L
04193500	January 1975–December 1995	Constant value of 57.0 mg/L
	January 1996–December 2004	Downward trend from 57.0 to 21.7 mg/L
05420500	January 1975–December 2004	Downward trend from 56.1 to 24.3 mg/L

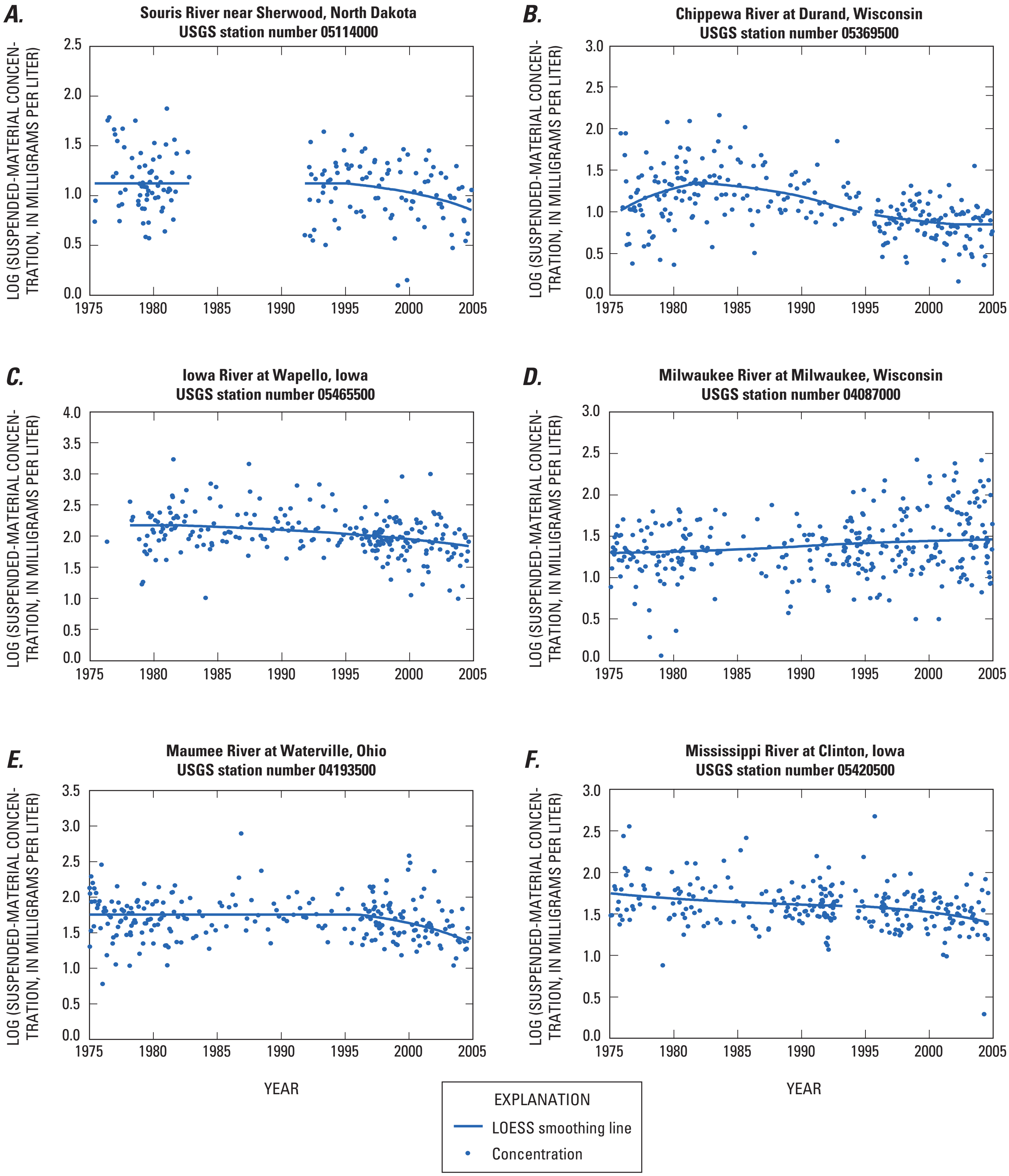


Figure 28. Long-term flow-adjusted trends in total suspended-material concentrations from 1975 to 2004 at six selected long-term sites.

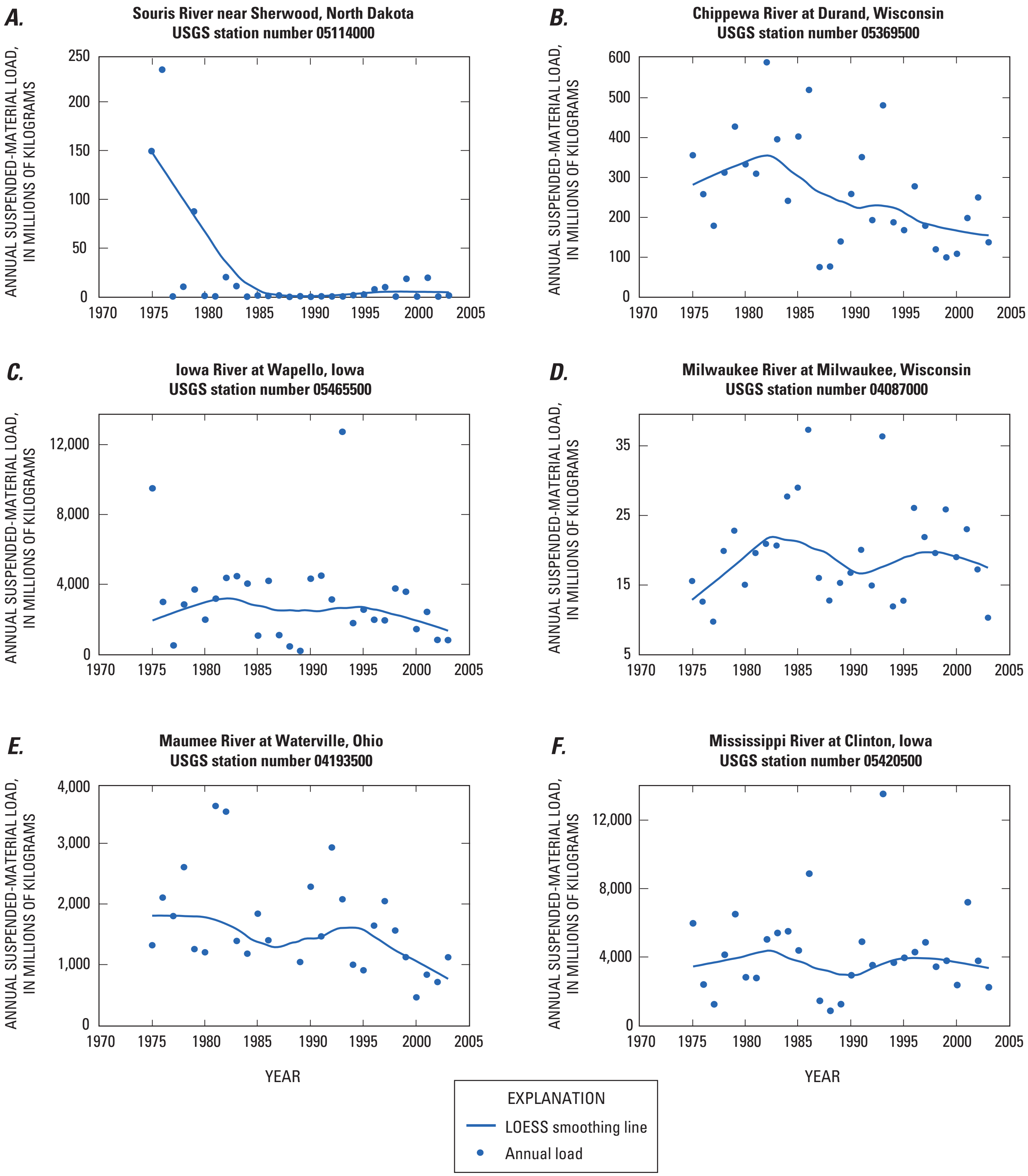


Figure 29. Long-term trends in total suspended-material loads from 1975 to 2003 at six sites in the study area.

Total Suspended Sediment

The USGS generally analyzes water samples for total suspended sediment (SS, USGS parameter code 80154), whereas most State agencies analyze their samples for total suspended solids (USGS parameter code 00530). Gray and others (2000) found that total suspended solid concentrations generally were less than SS concentrations by about 25 to 34 percent because the analytical procedure for total suspended solids inadequately accounts for the amount of sand-sized particles in the sample. This difference often is considered to be small compared to spatial differences that exist throughout large areas (Robertson and others, 2006); however, this bias may be significant when examining long-term trends. Therefore, SS data were analyzed separately to note potential bias in SMAT trends.

Median midmonthly SS concentrations from 22 trend sites ranged from 5.50 to 424 mg/L, based on data from 1993 to 2004. The mean and median values of the median SS concentrations at these trend sites were 65.2 and 40.5 mg/L, respectively. The highest concentrations again were in the western, southwestern, and south-central parts of the study area. The lowest concentrations were mostly in northern parts of Wisconsin and Michigan and in the northeastern part of the study area. The mean and median values based only on the SS data were slightly higher than those for SMAT when all of the sites that had only total suspended solids data were removed. This again indicates that SMAT concentrations may be biased slightly low compared to SS concentrations.

Median annual SS yields ranged from 1,620 to 939,000 kg/km^2 (table 15). The mean and median values of the median SS yields at these sites were 75,500 and 25,200 kg/km^2, respectively. Similar to SMAT yields, the highest SS yields were throughout the southern and eastern parts of the study area. The lowest SS yields were throughout the northern part of the study area.

Trends in Total Suspended-Sediment Concentrations

The spatial tendencies in the FA trends in SS concentrations from 1993 to 2004 resembled those for SMAT, but with several differences (table 15; fig. 30). The main difference was that some of the downward trends observed for the SMAT data in Wisconsin and New York and upward trends in Illinois were not observed for SS data because most of these sites only had total suspended-solids data. In addition, when the total suspended-solids data were removed from the Popple River data in northern Wisconsin, the upward trend in SS became significant. Of the 22 sites that had sufficient streamflow and SS concentration data for analysis (table 15), 15 sites had downward trends, of which 7 trends were statistically significant at $p < 0.10$, and 7 sites had upward trends, of which 3 trends were statistically significant.

When the OA trends in SS concentrations from 1993 to 2004 were examined, a few of the downward FA trends in Iowa and Illinois became statistically significant ($p < 0.10$; fig. 30). The differences between FA and OA trends in SS concentrations are again attributed to the trends in streamflow (fig. 12). Of the 22 trend sites, 16 sites had downward trends, of which 10 trends were significant, and 6 sites had upward trends, of which 1 trend was significant.

Trends in Total Suspended-Sediment Loads

From 1993 to 2004, most of the SS data showed a tendency toward decreasing loads. Nineteen of the 22 sites had a downward trend in SS loads (table 15; fig. 30***C***). Trends at 11 of the 19 sites with downward trends in loads were significant, but the trend at only one site with an upward trend was significant ($p < 0.10$). The only site with a significant increase in loads was in northern Wisconsin. The slight upward trend in SMAT loads in the western part of the study area could not be confirmed because the western part of the study area could not be analyzed for SS trends.

Understanding Water-Quality Trends in the Context of Human Activity

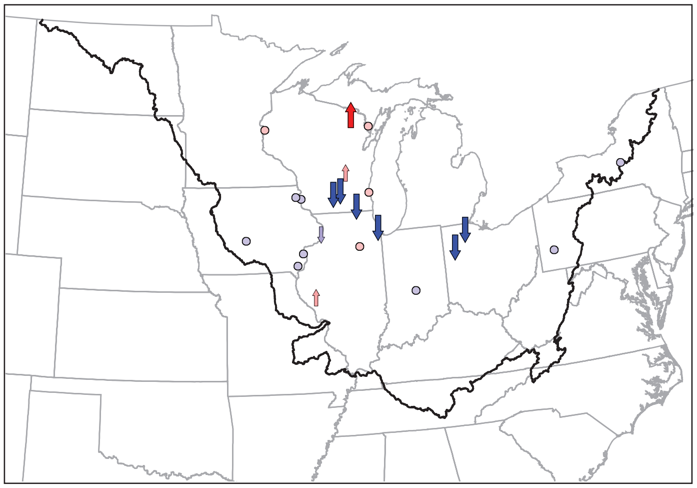
Water-quality studies that summarize and analyze concentration or load data from many monitoring sites provide valuable insights into statewide, regional, or in some cases national trends. A monitoring site may have a downward trend in, for example TN concentrations, that is not statistically significant; however, when that trend is analyzed in a regional context where proximate sites have similar environmental characteristics, especially for the same type of land use, and significantly decreasing TN concentrations, then the downward trend at the original site becomes a candidate for inclusion as part of the broad regional downward trend. In this report, no statistical test was applied to the regional trend. Additional information is thus obtained by analyzing data in a regional context.

Table 15. Summary of results for analyses of flow-adjusted trends, overall trends, and trends in loads for total suspended sediment (SS, USGS parameter code 80154), 1993 to 2004.

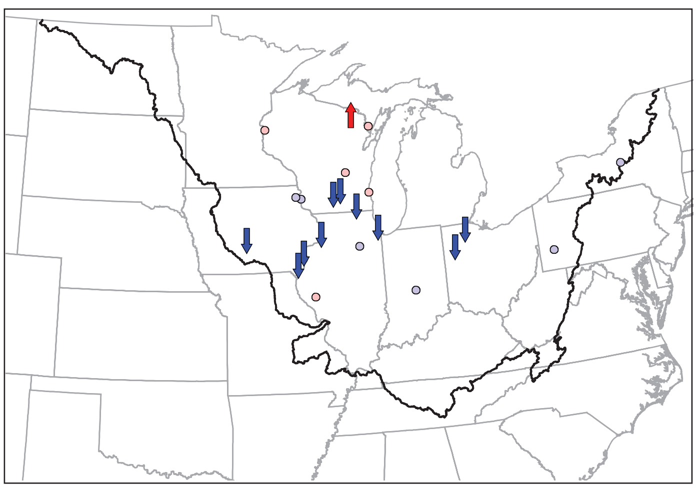
[USGS, U.S. Geological Survey; ID, identification; mg/L, milligram per liter; mg/L/yr, milligram per liter per year; %/yr, percent per year; kg, kilogram; kg/km², kilogram per square kilometer; kg/yr, kilogram per year; NA, not available; NC, not calculated; sites in **bold** indicate long-term evaluation sites]

USGS site number	State-ID number (figure 1)	Water-quality model (table 3)	Concentration: Reference concentration (mg/L)	Concentration: Flow-adjusted trend: Trend (mg/L/yr)	Concentration: Flow-adjusted trend: Trend (%/yr)	Concentration: Flow-adjusted trend: p-value	Concentration: Overall trend: Trend (mg/L/yr)	Concentration: Overall trend: Trend (%/yr)	Concentration: Overall trend: p-value	Annual loads: Median annual load (kg)	Annual loads: Median yield (kg/km²)	Annual loads: Reference load (kg)	Annual loads: Trend (kg/yr)	Annual loads: Trend (%/yr)	Annual loads: p-value
Central part															
04063700	WI-9	3	4.0	0.823	2.78	0.000	0.727	18.35	0.000	1,130,000	3,130	330,000	29,500	8.95	0.026
04067500	WI-10	5	4.6	.033	.72	.721	.031	.67	.789	16,500,000	1,620	10,900,000	69,000	.64	.849
04073468	WI-11	9	21.9	.414	1.89	.081	.369	1.68	.128	897,000	6,470	586,000	-6,200	-1.07	.708
04087000	**WI-13**	**7**	**32.5**	**.549**	**1.69**	**.308**	**.403**	**1.24**	**.464**	**23,800,000**	**13,200**	**10,500,000**	**-43,000**	**-.41**	**.879**
05340500	WI-15	3	5.6	.159	2.83	.369	.086	1.54	.631	59,400,000	3,680	20,400,000	57,800	.28	.937
05389400	IA-18	9	83.9	-.074	-.09	.977	-1.581	-1.88	.410	1,100,000	12,500	2,080,000	-110,000	-5.35	.007
05389500	IA-19	3	30.9	-.445	-1.44	.602	-.706	-2.28	.364	2,140,000,000	12,200	1,290,000,000	-58,000,000	-4.53	.049
05420500	**IA-21**	**8**	**51.8**	**-2.075**	**-4.00**	**.067**	**-2.829**	**-5.46**	**.005**	**4,280,000,000**	**19,300**	**2,790,000,000**	**-200,000,000**	**-7.03**	**.000**
05427718	WI-22	9	40.4	-1.257	-3.11	.000	-1.942	-4.80	.000	2,210,000	11,600	929,000	-56,000	-6.00	.000
05427948	WI-23	8	33.7	-1.006	-2.98	.000	-1.046	-3.10	.000	1,410,000	31,900	95,800	-3,300	-3.43	.024
054310157	WI-24	8	48.1	-1.422	-2.95	.007	-1.410	-2.93	.010	447,000	39,800	59,400	-1,600	-2.76	.280
05465500	**IA-25**	**8**	**181.0**	**-2.893**	**-1.60**	**.401**	**-9.416**	**-5.20**	**.008**	**1,860,000,000**	**57,300**	**1,640,000,000**	**-120,000,000**	**-7.21**	**.004**
05474000	IA-26	7	207.0	-4.453	-2.15	.267	-12.022	-5.81	.007	1,420,000,000	127,000	417,000,000	-33,000,000	-7.80	.006
05481650	IA-27	3	52.9	-1.361	-2.57	.232	-2.444	-4.62	.007	213,000,000	14,100	166,000,000	-13,000,000	-7.83	.000
05532500	IL-32	3	129.0	-11.148	-8.64	.038	-11.653	-9.03	.025	178,000,000	109,000	61,900,000	-6,100,000	-9.87	.017
05543500	IL-34	8	30.1	.124	.41	.876	-.501	-1.66	.505	540,000,000	25,200	283,000,000	-14,000,000	-4.95	.045
05586100	IL-37	9	140.9	7.302	5.18	.068	2.130	1.51	.552	3,260,000,000	47,000	2,830,000,000	-130,000,000	-4.43	.131
Eastern part															
03085000	PA-40	8	12.6	-0.485	-3.86	0.331	-0.351	-2.79	0.517	630,000,000	33,100	83,000,000	-770,000	-0.93	0.865
03353637	IN-42	8	20.1	-.267	-1.32	.531	-.471	-2.34	.309	3,780,000	85,800	170,000	-7,500	-4.39	.278
04186500	OH-43	7	47.4	-3.272	-6.90	.001	-3.301	-6.96	.002	32,500,000	37,700	3,550,000	-250,000	-7.10	.081
04193500	**OH-44**	**9**	**32.9**	**-1.322**	**-4.02**	**.008**	**-1.268**	**-3.86**	**.043**	**1,040,000,000**	**63,600**	**61,900,000**	**-2,200,000**	**-3.54**	**.347**
04237946	NY-49	9	984.6	-13.718	-1.39	.288	-14.079	-1.43	.280	778,000	939,000	608,000	-7,100	-1.17	.518

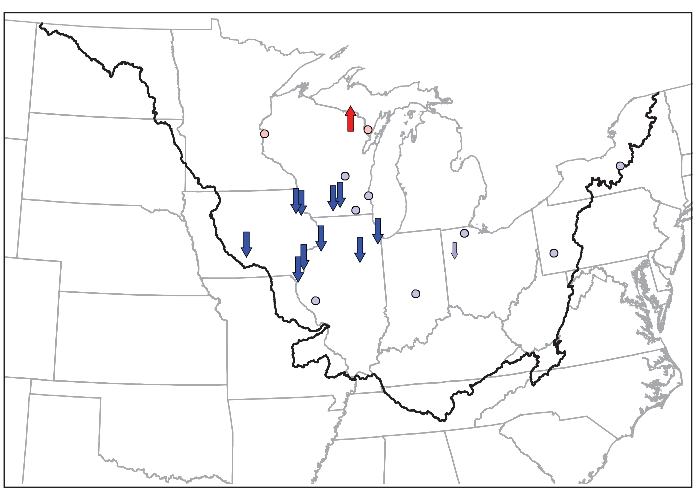
***A.* Flow-adjusted trend in concentrations**



***B.* Overall trend in concentrations**



***C.* Trend in annual loads**



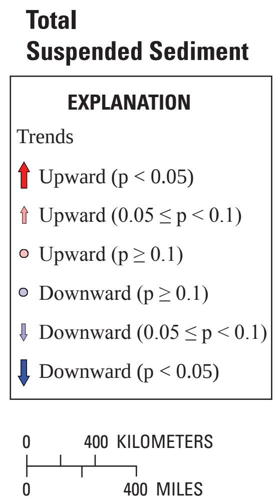


Figure 30. Trends in total suspended sediment from 1993 to 2004 for ***A***, flow-adjusted concentrations, ***B***, overall concentrations, and ***C***, annual loads throughout the study area.

Ideally, any regional trends in concentrations or loads would be linked to changes in sources in a cause-and-effect relation. Mosteller and Tukey (1977) proposed that four conditions should be satisfied to define a cause-and-effect relation: (1) an association, such as a statistically significant correlation between the source variable and the response variable; (2) responsiveness, such as both proportional increases and decreases in response to the correlated changes in the source variable; (3) a mechanism to explain the relation; and (4) consistency, which requires that the relation should be repeatable at more than one location. Defining or proving cause-and-effect relations can be time-consuming, data-intensive, and costly. Few programs or studies have sufficient resources to establish such relations at statewide, regional, or national levels. One type of analysis used in this report to indicate cause-and-effect relations is the consistency in the patterns in the trends in nutrient inputs and the patterns in the trends of the water-quality data. These spatial associations are qualitative, but they are strengthened by the consistency of a large number of sites. A second type of analysis used to derive possible cause-and-effect relations is a regression approach between the changes in concentrations and yields at selected sites and the changes in land use and the amount of a specific source of nutrients or sediment. Flow-adjusted trends in concentrations, being independent of streamflow conditions, are best used to evaluate changes in water quality arising from changes in management activities within a basin. Flow-adjusted trends are therefore examined in this section to try to describe potential cause-and-effect relations in the context of human activities.

Total Nitrogen

Of the 24 sites examined for FA trends in TN concentrations, 19 sites had upward trends, and 5 sites had downward trends (fig. 15***A***). The FA trend patterns for TN are similar to the trends in fertilizer nitrogen applications (fig. 7***B***). The similarity in the patterns of these trends indicates a broad regional association between FA trends in TN and fertilizer applications.

The FA trends in TN concentrations at 10 of these sites had significant ($p < 0.10$) trends, most of which were in the central part of the study area (fig. 15***A***). Six of these sites (05082500, 04087000, 05287890, 05420500, 05465500, and 05586100) were classed as agricultural, and FA trends were upward at all these sites except 05082500. Three sites with significant upward trends (04063700, 05369500, and 05382000) were classed as forested. One site with a significant downward trend (05540275) was classed as mixed. The forested sites in northern Wisconsin and the Upper Peninsula of Michigan also may be affected by the increase in atmospheric deposition of nitrogen (fig. 9***B***). Because nitrogen input from fertilizers is the dominant source of nitrogen in agricultural areas, the annual value in each basin was analyzed in a regression analysis to put the trend into the context of human activity. Atmospheric deposition was used as an ancillary variable in the analysis of trends at the forested sites because that is the major source of nitrogen in those basins. Nitrogen from fertilizer applications was not significant at $p < 0.10$ for any agricultural site. The lack of significance indicates that many factors affect TN in streams, such as agricultural practices including artificial drainage, land-retirement plans, tillage practices, and animal husbandry. The use of nitrogenous fertilizers has remained relatively stable over the 12-year period, increasing in some areas and decreasing in others (fig. 7***B***). The significant upward trends in TN concentrations at the five agricultural sites and upward trends at several of the other sites indicate that efforts to reduce runoff from agricultural lands have not been universally successful. At least two reasons account for that lack of success—the first is that local benefits do not necessarily translate to basinwide benefits, and the second is that unregulated activities such as tile drainage may counteract any benefit from efforts to reduce agricultural runoff. Nitrogen as nitrate moves easily through the soils to tile drains and into streams. In tile-drained areas, nitrate makes up most of the TN in the streams (Baker and others, 2006). Because much of the row cropland in the study area is heavily tiled (Strock and others, 2007), much of the nitrate enters streams by bypassing best management practices such as filter strips (Baker and others, 2007; Strock and others, 2007). Another confounding factor is the growth of confined animal feeding operations (CAFOs) in the study area (Horne, 2000). These CAFOs are highly regulated—the Clean Water Act (U.S. Environmental Protection Agency, 2006a) requires operators to carry a national pollutant discharge elimination system permit; the concentration of animal waste requires proper disposal.

Annual atmospheric deposition inputs were significant at $p < 0.10$ for two of the three forested sites, and fertilizer input was significant at one site (04063700). The results for the atmospheric deposition analysis indicate that the atmospheric deposition of nitrogen may have some effect on TN concentrations in streams.

The mixed site had a downward trend, which indicates that urban controls on nitrogen could have been effective in reducing total nitrogen in that stream. Unfortunately, no ancillary data pertaining to urban controls are available to test that conclusion.

Dissolved Ammonia

One of the goals of the Clean Water Act (U.S. Environmental Protection Agency, 2006a) is to reduce ammonia concentrations in streams through management of both animal and human wastes. The widespread, downward trends in NH4 concentrations for the 36 trend sites indicate that those goals are being met; 24 of the 36 trend sites had significant downward trends ($p < 0.10$). Regionally, good agreement exists between FA trends in NH4 concentrations (fig. 18***A***) and change in manure nitrogen application rates (fig. 8***B***), except in the farthest western part of the study area.

The strength of the relation between FA trends in NH4 concentrations and the percent change in manure nitrogen was evaluated for agricultural sites. The p-value of the Spearman rank correlation estimate was 0.13, which indicates that the relation is not statistically significant at $p < 0.10$. This finding implies that although reduction in manure input may partially account for the reduction in FACs for NH4 , other efforts, such as improved wastewater treatment, have contributed to the observed downward trends.

The median values of the reference concentrations were about 0.05 mg/L for sites with no significant trends and about 0.06 mg/L for sites with significant trends. Those two medians are not substantially different. However, the range in concentrations was smaller for those sites that did not have significant trends than those that did, which indicates for sites without significant trends that either NH4 was never an issue or that controls on waste had been effective earlier. The site that had a significant upward trend, Northrup Creek, N.Y. (0422026250), had a reference concentration of 0.032 mg/L. Land use at that site was classified as mixed, and the site has a relatively small drainage area of 26 km^2, which implies that a very small, possibly local, source may be responsible for the trend observed at that site.

Total Kjeldahl Nitrogen

The FA trends in KJN concentrations from 1993 to 2004 showed no distinct spatial pattern. Of the 36 sites that had sufficient streamflow and KJN concentration data for analysis (table 8; fig. 19***A***), 17 sites had downward trends, and 19 sites had upward trends in concentrations. Seven sites had significant downward trends, and six sites had significant upward FA trends ($p < 0.10$). In general, human activities that affect KJN are the processing of human and animal waste, which can affect both NH4 and organic nitrogen in streams. A comparison between the change in TN from manure from 1992 to 2002 (fig. 8***B***) and the FA trends in KJN concentrations (fig. 19) indicates that less manure was produced in all of the basins in 2002 than in 1992, except for the Maumee River, Ohio (04193500), which had a slight increase in manure production and a significant upward trend in FA concentrations. The broad decrease in manure production in most basins does not explain the significant upward trends in KJN at the remaining five sites. The p-value of the Spearman rank correlation test was 0.49. Human population was constant or increased in all but three basins from 1990 to 2000. The p-value of the Spearman rank correlation between KJN trends and changes in human population was 0.56. The increase in human population and apparent decrease in animal manure production are competing factors that confound the interpretation of trend in KJN concentrations. Other confounding factors are the changes in treatments of those human and animal wastes.

Dissolved Nitrite Plus Nitrate

The FA trends in NO2NO3 concentrations from 1993 to 2004 showed no distinct spatial pattern (fig. 20***A***). Of the 29 sites that had sufficient data for analysis, 12 sites had downward trends, and 17 sites had upward trends in concentrations. Some general agreement exists between the FA trends in NO2NO3 concentrations (fig. 20***A***) and the change in nitrogen fertilizer application (fig. 7***B***). The small changes in the forested sites in northern Wisconsin are consistent with the changes in atmospheric deposition of nitrogen (fig. 9).

Only two sites—Milwaukee River, Wis. (04087000) and Chippewa River, Wis. (05369500)—had significant ($p < 0.10$) positive relations between FA trends in NO2NO3 concentrations and the logarithm of nitrogen fertilizer application as an ancillary variable for trend. Another site, Little Buck Creek, Ind. (03353637), had a significant negative relation between FA trends in NO2NO3 concentrations and the logarithm of nitrogen fertilizer application; that is, increasing fertilizer application resulted in a decrease in concentrations, which contradicts a reasonable mechanism, although the substantial decrease in manure applications, about 80 percent decrease, could explain the downward trend.

An issue related to downward trends in NH4 concentrations often is an associated upward trend in NO2NO3 concentrations, which could be a product of the microbiologically mediated oxidation of NH4. Three sites—Milwaukee River, Wis. (04087000), Iowa River, Iowa (05465500), and Illinois River, Ill. (05586100)—had significant upward FA trends in NO2NO3 concentrations; however, none of these sites had significant downward FA trends in NH4 concentrations. Two sites—Little Buck Creek, Ind. (03353637) and Spring Brook, Ill. (05540275)—had significant downward FA trends in NO2NO3 concentrations. Both of these sites had significant downward FA trends in NH4 concentrations, and both sites are very urbanized and have 45 and 27 percent urban land cover, respectively.

Total Phosphorus

From 1993 to 2004, FA trends in TP concentrations were upward at 24 of the 40 sites (fig. 22***A***). The 24 sites with upward trends were distributed throughout the entire study area; however, most of these sites were in the central part and the eastern side of the western part of the study area. Twelve of these sites, mostly in the central part of the study area, had significant upward trends ($p < 0.10$). Downward trends were significant at 7 of the 16 sites, mostly in Wisconsin and the eastern part of the study area. A weak agreement exists between these trends and the trends in phosphorus fertilizer application (fig. 7***C***). Both trends indicate increases west of the Mississippi River and decreases in Wisconsin and eastern part of the study area. However, several sites in central and northern Illinois and in western Wisconsin are not consistent with this relation and had significant upward trends in TP concentrations ($p < 0.05$). These concentration increases may be associated with increases from local point sources.

Changes in the amount of phosphorus fertilizer use are consistent with the trend in FA trend in TP concentrations at four sites—Green Lake Inlet, Wis. (04073468), Pheasant Branch, Wis. (05427948), Jackson Creek Tributary, Wis. (054310157), and Northrup Creek, N.Y. (0422026250)—all of which had significant downward trends. However, phosphorus fertilizer use was negatively associated with the FA trend in TP concentrations at six sites ($p < 0.05$), which contradicts its use in explaining the observed trend at these sites. The lack of a strong association between phosphorus fertilizer use and FA trends in TP concentrations indicates that other factors are important in determining TP concentrations in streams.

Dissolved Phosphorus

The FA trends in DP concentrations from 1993 to 2004 showed mixed regional tendencies; however, DP concentrations consistently decreased at all of the sites in the eastern part of the study area (fig. 25***A***). Many of the sites with significantly decreasing TP concentrations had little change in DP concentrations. Of the 35 sites analyzed, 23 sites had downward trends in DP concentrations; most of these sites were east of the Mississippi River, and they included 10 of the 12 sites with significant ($p < 0.10$) downward trends. A comparison of these patterns with those of nonpoint phosphorus inputs (fig. 10***C***) and those for fertilizers and manure indicate fairly good regional agreement indicating that reduction in fertilizers and manure east of the Mississippi River could explain most of the observed trends in DP concentrations. Some discrepancies occur in Illinois, but those could be caused by nonagricultural influences.

Excluding the sites classified as urban in part of the study area east of the Mississippi River, 7 of the remaining 11 sites have a significant positive relation between the FA trends in DP concentrations and the logarithm of phosphorus fertilizer application rates as an ancillary variable in a regression analysis. No sites west of the Mississippi River had a significant relation between the FA trends in DP concentrations and phosphorus fertilizer application rates. This again indicates that the reductions in phosphorus fertilizer application rates possibly have had some effect on the concentration of phosphorus in the streams.

With the exception of one site, Spring Brook, Ill. (05540275), the correlation between trends in phosphorus fertilizer use and FA trends in DP concentrations was significant (fig. 31; Spearman correlation coefficient p-value was less than 0.05). Although most sites had downward trends in farm phosphorus use and FACs for DP—two sites Salt Creek, Ill. (05531500), and Des Plaines River, Ill. (05532500)—had the opposite relation. Land use at both of these sites was classified as predominantly urban. The urban and mixed sites had no significant relation between trends in phosphorus fertilizer use and FA trends in DP concentrations (fig. 31). All of the urban and mixed sites were well above the 1:1 reference line in figure 31, indicating that sources other than farm fertilizers affect the FA trends at those sites. In addition, all urban and mixed sites, except 05531500, had upward trends in nonfarm phosphorus fertilizer use, but FA trends in DP concentrations were mixed—four sites had downward trends and four sites had upward trends.

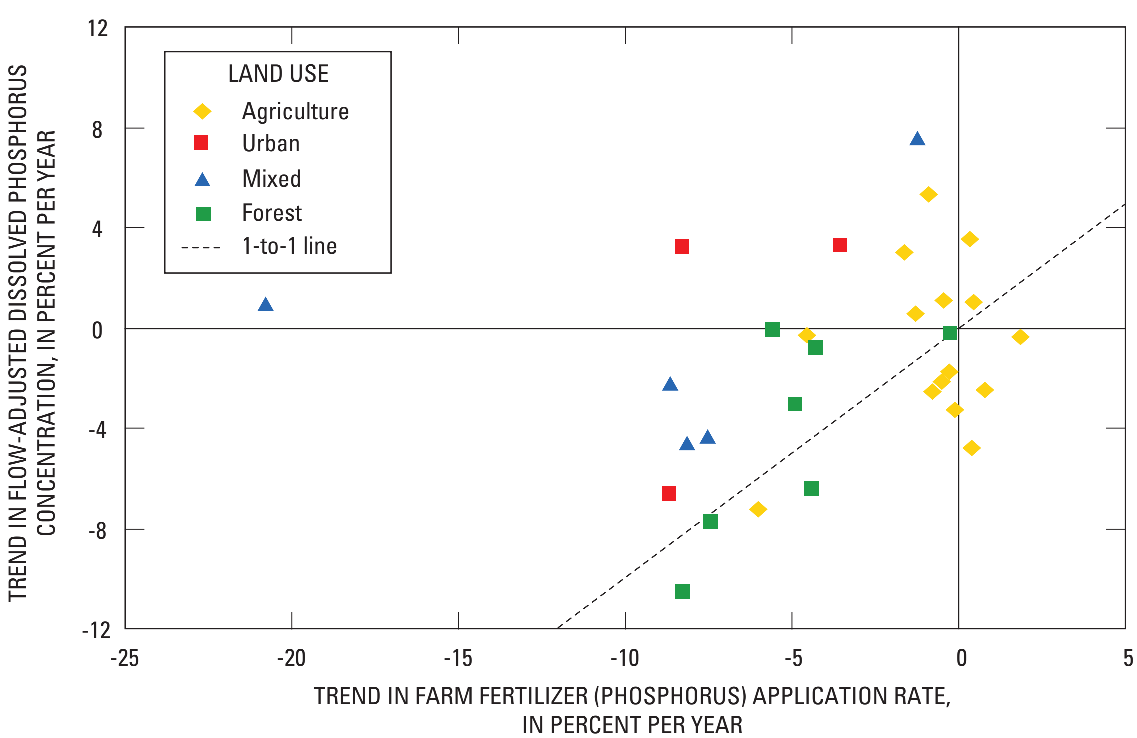


Figure 31. Relation between trends in flow-adjusted dissolved phosphorus concentrations and trends in phosphorus inputs from farm fertilizers for sites with varying types of land use in the basin, 1993 to 2004.

Total Suspended Materials

The FA trends in SMAT concentrations from 1993 to 2004 showed distinct spatial patterns (fig. 27), with significant upward trends in Illinois, and downward trends throughout most of Wisconsin, Iowa, and in the eastern part of the study area (fig. 27***A***). Of the 42 sites, 12 sites, mostly east of the Mississippi but not in Illinois, had significant ($p < 0.10$) downward trends. All but one of the five sites with statistically significant increases in SMAT were in Illinois. A comparison of these patterns with those of FA trends in TP concentrations (fig. 22***A***) indicates fairly good regional agreement.

To better determine if the FA trends in SMAT concentrations (fig. 27***A***) were related to the FA trends in TP concentrations (fig. 22***A***), the log-transform of FACs for SMAT was included as an ancillary variable in the regression models that showed significant FA trends in TP concentrations. The log-transform of SMAT was significant ($p < 0.10$) at 16 of the 19 sites. The log-transform of SMAT concentrations was not significant at one site (05114000), and data for two Red River of the North sites—05064500 in Minnesota and 05082500 in North Dakota—were insufficient to evaluate the relation. This indicates that the FA trends in TP concentrations are strongly related to the FA trends in SMAT concentrations, which is very reasonable because TP includes phosphorus bound to suspended material.

FA trends in SMAT can be related to agricultural practices, urbanization, and forest management. The widespread downward FA trends in SMAT concentrations indicate that best management practices may have had some effect on reducing suspended material in many of the streams. The upward trends in Illinois are in urban and heavily agricultural areas and indicate that there may be confounding factors in the Illinois River Basin. Many types of human activity in streams, such as construction or removal of dams or bridges, can affect suspended material in streams. Natural changes, such as degradation or aggradation of the stream bed, also can affect suspended material. Ancillary data about these activities and changes are not broadly available and complicate the interpretation of trends in suspended material.

Changes in Loads from 1993 to 2004 in a Longer Term Context

Changes in TN, TP, and SMAT loads were examined from 1975 to 2003 at six sites (figs. 17, 24, and 29). These changes were examined to provide a longer term perspective on the data examined for the 1993 to 2004 period. The long-term, nonmonotonic trends, which were shown as curves on the 1975–2003 graphs, were generated by LOESS smoothing, a type of moving average function. The statistical significance of those trends over the 1975–2003 period cannot be determined; however, the long-term changes for annual streamflow and load data indicate that the monotonic trends from 1993 to 2004 should not be extrapolated backward in time.

Examination of the longer term (1975–2003) mean annual streamflow data (fig. 13) and associated loads of TN (fig. 17), TP (fig. 24), and SMAT (fig. 29) at the six long-term sites indicates that the trends in streamflow and associated loads from 1993 to 2004 are not, in most cases, simple continuations of longer term linear trends; rather, the changes consist of a series of increases and decreases in streamflow and the associated loads. Most of the changes in long-term loads are directly caused by changes in streamflow rather than changes in nutrient or sediment concentrations resulting in long-term changes in loads resembling the long-term changes in streamflow. The most prominent example of this is in data for streamflow, and TN, TP, and SMAT loads at the Souris River, N. Dak., from 1975 to 2003. During this time, both streamflow and the associated loads, although trending downward, were all substantially larger during the 1970s and early 1980s than during 1993–2004 (figs. 13***A***, 17***A***, 24***A***, and 29***A***). Streamflow and associated TN, TP, and SMAT loads in the Souris River are to some extent artificially controlled by the large Canadian dams located upstream, especially in the 1993–2004 period when two of the dams were constructed.

In addition, a few of the sites are in the same basin and, therefore, trends should resemble one another. For example, the Chippewa River (site 16 in fig. 11) is in the UM Basin upstream from the Mississippi River at Clinton, Iowa (site 21 in fig. 11), which partially accounts for the similarities in trends in the streamflow and loads at these two sites (figs. 13***B***,***F***; 17***B***,***F***; 24***B***,***F***; and 29***B***,***F***). The largest exception to the general bimodal similarity, which was caused by similarities in climate over the UM region, is a local wet year recorded at the Chippewa Basin during 2002 that was not representative of conditions in the larger UM Basin upstream from Clinton. This difference resulted in the LOESS curves for streamflow, TN, and TP at the Chippewa River site trending upward during 2000–2003, whereas those for the Mississippi River at Clinton trended downward.

Annual precipitation across the study area (fig. 4) and associated streamflow and loading at the Iowa River at Wapello, Iowa, and at the Maumee River, Ohio, varied substantially during 1975–2003 (figs. 13***C***,***E***; 17***C***,***E***; 24***C***,***E***; and 29***C***,***E***). The wet years generally averaged out with the dry years because the LOESS procedure at these two sites do not show any substantial or sustained upward or downward trends during this period. With the exceptions of the unusually dry year in 1989 and the unusually wet year in 1993 for the Iowa River at Wapello, Iowa, the short-term trends in streamflow, TP, TN, and SMAT during 1993–2004 generally were representative of conditions during the longer term period 1974–2003.

Unlike streamflow and TP loads, SMAT loads in the Maumee River decreased during 1996–2003 relative to 1975–1995 (fig. 29). Such a decrease in SMAT loads relative to streamflow is of interest in the Maumee River Basin, because the Maumee River has been identified as being the single greatest contributor of sediment to Lake Erie and is an official USEPA "Area of Concern" for this and other water-quality problems (U.S. Environmental Protection Agency, 2007). Because the FA trend in SMAT concentrations during 1993–2004 is significantly downward (fig. 27, table 13) and although the trend in streamflow is slightly upward (fig. 12, table 4), the decrease in SMAT loads could be attributed to implementation of sediment control practices in the Maumee River Basin, which began in 1987 (U.S. Environmental Protection Agency, 2007).

The Milwaukee River at Milwaukee, Wis., drains to Lake Michigan and has the smallest drainage area of the six sites selected for the long-term analysis. Annual TN and TP loads for the Milwaukee River during 1975–1989 (figs. 17***D***; 24***D***) were larger than those measured during 1993–2004 because of trends in annual precipitation (fig. 4) and associated streamflow (fig. 13***D***). The SMAT loads during 1975–1984 showed an upward trend, but not substantially different in magnitude than during 1985–2004 (fig. 29***D***).

Understanding Trends in Loads in the Context of Climatic Variability

The insights provided by data in figures 13, 17, 24, and 29 from sites that have a long history of streamflow and water-quality monitoring demonstrate the value of establishing and maintaining a long-term streamflow and water-quality monitoring network for planning purposes in the United States, especially as current thought (Solomon and others, 2007) indicates that global warming could lead to large-scale changes in the climate of North America during the 21st century. In general, such large-scale regional changes in climate and associated hydrology could necessitate modifications to urban infrastructure if changes occur in lake levels, flood crests, or flood frequency, or if lesser rainfall in some areas places unanticipated strains on water use and sewerages.

The nutrient and sediment loads described in this report primarily are controlled by changes in streamflow rather than changes in concentration, as governed by the substantially larger magnitude of the streamflow coefficients in all load computations relative to the much smaller concentration coefficients. In almost all cases for both the short-term loads (1993–2004) and the long-term loads (1975–2003), changes in loading closely mirror changes in streamflow because of dominance of streamflow coefficients. Sources of nitrogen, phosphorus, and sediment were relatively steady over the study period. Therefore, changes and trends in annual loading primarily were attributed to long-term trends and the regional distribution of precipitation (fig. 4) or water use, such as dam management or irrigation, and associated streamflow rather than the magnitude of trends in TP, TN, and SMAT concentrations. The implication of streamflow dominance in the load computations is that the 12-year loads presented in this report could be at least crudely estimated backward to the mid-1970s if a streamflow record exists for the period of interest and an assumption is made that nutrient and sediment sources did not change substantially in the basin during the period of interest.

The strong influence of streamflow on nutrient and sediment loads implies that long-term trends in loads can be more sensitive to changes in climate rather than anthropogenic changes. For example, SMAT data for the Souris River, N. Dak., shown on figure 29***A***, might indicate that the substantial reduction in loads during the period 1975–1983 is attributed to some widespread implementation of an effective agricultural best-management practice for sediment control or improvements in industrial or sewage treatment plant processes; however, further consideration is required to understand the loads. As previously described, the streamflow in the Souris River decreased dramatically through time and, in addition, the Souris River is downstream from major dams in Canada, so streamflow in the Souris River is artificially controlled. Therefore, examination of FA trends in concentrations provides much more information regarding the effects of anthropogenic practices.

Comparison between Nutrient Loads from This Study and Loads Contributing to the Gulf of Mexico Hypoxic Zone

In addition to degrading the local quality of the streams, the nutrients and sediments transported in the streams can cause problems in downstream reservoirs, lakes, and coastal ecosystems. An increasing supply of nutrients, primarily nitrogen, from the Mississippi River Basin has been implicated as one of the main causes for the expanding hypoxic zone in the Gulf of Mexico (Rabalais and others, 2002; Goolsby and Battaglin, 2000). The Mississippi River and its distributary, the Atchafalaya River, drain an area of about 3,200,000 km^2, or about 41 percent of the conterminous United States. In an average year, the Mississippi-Atchafalaya River (MAR) Basin contributes about 1,568,000,000 kg of TN to the Gulf of Mexico (Goolsby and others, 1999).

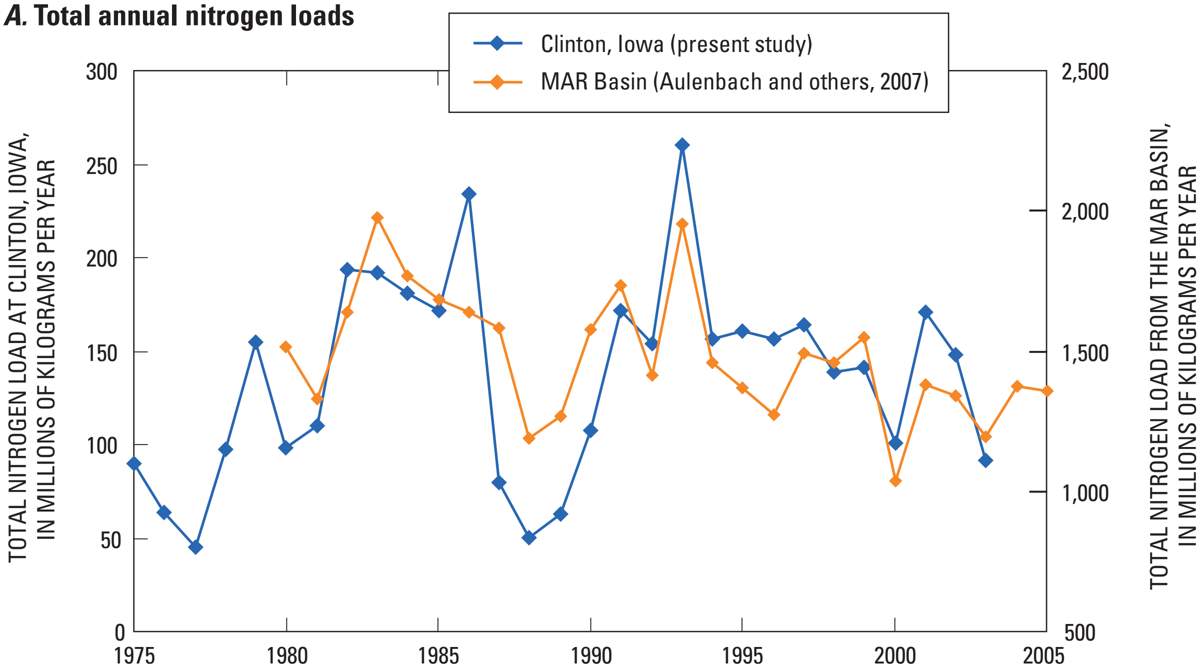
Relative loading (yields) of nitrogen from the individual tributaries of the Mississippi River are not uniform. To better define where these nutrients originate, Goolsby and Battaglin (2000) quantified the TN loading from 42 large drainages within the Mississippi River Basin, including the Mississippi River at Clinton, Iowa, by use of a multiple regression approach, which is a method similar to that used in this study to compute loads. They then related these data to various nitrogen sources to estimate yields from similar sized drainages throughout the entire basin. They found that the highest TN yields were from areas with the highest streamflow, precipitation, and nitrogen inputs, primarily fertilizers; the highest yields were from drainages in the Ohio and middle part of the Mississippi River Basins. For the period from 1980 to 1996, they found that the UM Basin contributed 9.6 percent or 149,800,000 kg of the TN load of 1,568,000,000 kg to the Gulf of Mexico from the MAR Basin, whereas the UM Basin represented only about 6.9 percent of the entire area. The load computed at the Mississippi River by Goolsby and Battaglin (2000) at Clinton, Iowa, is similar to that computed in this study of 149,300,000 kg for the same years. Only the Ohio River Basin and middle part of the Mississippi River Basins contributed a relatively higher percentage of the total load than the UM Basin. Goolsby and Battaglin (2000) found that streams draining Iowa and Illinois contribute as much as 35 percent of the TN flux during years with average and high rainfall, although they represent only 9 percent of the total area. The UM Basin contributed about 6 percent or 8,000,000 kg of the TP load of 136,500,000 kg from the MAR Basin, which is similar to the percentage of the area it represents. The TP load computed by Goolsby and Battaglin (2000) at the Mississippi River at Clinton, Iowa, is slightly less than that computed in this study of 9,030,000 kg for the same years.

The size of the hypoxic zone in the Gulf has been shown to be directly related to the TN flux from the MAR Basin (Scavia and others, 2003). Rabalais and others (2002) concluded that almost a three-fold increase in TN load to the Gulf has driven the long-term increase in hypoxia since the middle of the last century. To determine how the changes in the annual loads of TN and TP computed from the study area are related to changes in annual loads entering the Gulf, the long-term annual loadings at the Mississippi River at Clinton, Iowa, from this study were compared to total load estimated from the MAR Basin by the Aulenbach and others (2007).

The long-term annual TN and TP loads at the Mississippi River at Clinton, Iowa, are compared with those estimated from the MAR Basin in figure 32. Data from Aulenbach and others (2007) demonstrate a slight decrease in TN loads from the MAR Basin from 1980 to 2004. Although the TN load at Clinton, Iowa, represents only about 10 percent of the total load from the MAR Basin, the loads at Clinton, Iowa, are strongly correlated to those estimated from the MAR Basin. Annual variability in the TN loads at Clinton, Iowa, describe about 57 percent of the variance in the loads estimated by simple correlation analysis (Aulenbach and others, 2007). The loads estimated at Clinton, Iowa, indicate that the decrease in TN loads from 1980 to 2004 in the data from Aulenbach and others (2007) is not a long-term trend but rather a short-term decline after increasing from 1975 to about 1985.

Data from Aulenbach and others (2007) demonstrate no trends in TP loads from the MAR Basin from 1980 to 2004. The TP loads from Clinton, Iowa, represent a smaller proportion of the loads from the MAR Basin than for TN and result in poorer correlation between TP loads than between TN loads. Annual variability in the TP loads at Clinton, Iowa, describe only about 10 percent of the variance in the loads estimated by Aulenbach and others (2007), estimated by simple correlation analysis. The loads estimated at Clinton, Iowa, also indicate no long-term trend in TP loads.

A. **Total annual nitrogen loads**



B. **Total annual phosphorus loads**

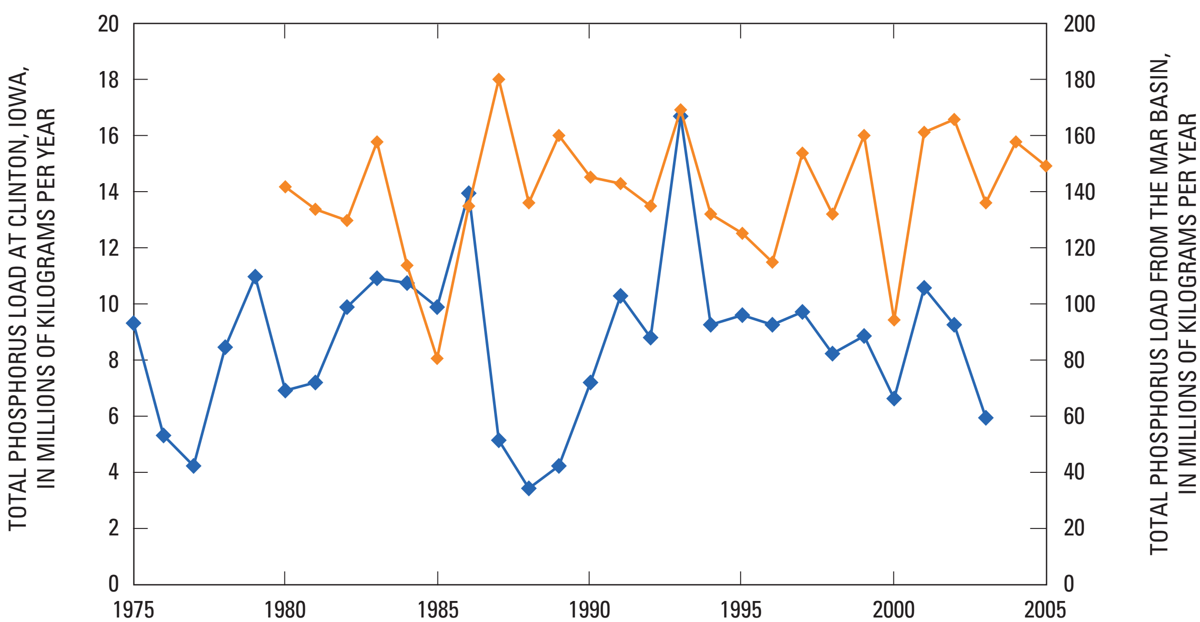


Figure 32. Total annual ***A***, nitrogen and ***B***, phosphorus loads at the Mississippi River at Clinton, Iowa (site with the largest drainage area in this study) and from the entire Mississippi/Atchafalaya River (MAR) Basin from 1975 to 2005 as estimated in this study and by Aulenbach and others (2007).

Summary

Elevated nutrient and suspended-sediment concentrations are two of the most common stressors or contaminants affecting streams throughout the United States. The elevated concentrations in rivers and streams can result in the overgrowth of benthic algae in shallow areas and areas with fast current and in an overabundance of phytoplankton and macrophytes in deep areas with slow current. High algal and macrophyte biomass, in turn, can cause severe diurnal fluctuations in dissolved oxygen and pH because of biotic production and respiration and can generate harmful organic materials when they die. Excessive transport of nutrients also has been linked to eutrophication of downstream lakes and impoundments, outbreaks of *Pfiesteria* in bays and estuaries in various Gulf and Mid-Atlantic States, and hypoxia in the Gulf of Mexico. Suspended sediment reduces clarity in streams and affects sight-feeding fish. Suspended sediment also interferes with water-treatment processes and recreational uses of streams. Excessive siltation can bury and suffocate fish eggs and bottom-dwelling organisms. In addition to instream effects, excessive sediment loading causes sedimentation problems in many downstream lakes and harbors and water-clarity problems in nearshore areas.

As a result of the Clean Water Act and subsequent regulations, many actions have been taken to reduce nutrient and suspended-sediment concentrations and the amount of nutrients and sediment transported in streams. It is important to assess how nutrient and suspended-sediment concentrations and loads in streams have changed in recent years to determine if these actions have been successful. A focus of this study was on flow-adjusted trends in concentrations, which reflect anthropogenic changes in the basin and are not the result of changes in climatic factors.

The study area comprises four major river basins: Upper Mississippi, Ohio, Red, and Great Lakes River Basins. The Upper Mississippi River Basin is defined as the Mississippi River and its tributaries upstream from the confluence with the Ohio River, excluding the Missouri River. The Ohio River Basin includes the Allegheny and Monongahela Rivers, and the Ohio River and its tributaries except the Tennessee River. The Red River Basin includes all rivers in the United States that drain northward to Lake Winnipeg and Hudson Bay, including the Red River of the North, the Souris River, and the Rainy River. The Great Lakes Basin includes all streams in the United States draining into the Great Lakes upstream from Montreal, Canada. Lake Michigan was joined with the Illinois River (Upper Mississippi River Basin) through construction of the Chicago Sanitary and Ship Canal in 1900, a project engineered to reverse the streamflow of the Chicago River in order to divert Chicago wastewaters into the Upper Illinois River and away from the historic receiving water of Lake Michigan.

Flow-adjusted (FA) and overall (OA) trends in concentrations and trends in loads were computed for total nitrogen (TN), dissolved ammonia (NH4), total organic plus ammonia (Kjeldahl; KJN), dissolved nitrite plus nitrate (NO2NO3), total phosphorus (TP), dissolved phosphorus (DP), total suspended material (SMAT), and total suspended sediment (SS) at selected sites in the study area. Only sites that had at least 10 years of data for at least one constituent from 1993 to 2004, at least 60 samples, and streamflow data from a gage or a nearby gage with a drainage-area ratio ranging from 0.9 to 1.1 were selected for trend analyses. The base period from 1993 to 2004 was selected because it is relevant to current conditions and represents the period since the beginning of the National Water-Quality Assessment Program of the U.S. Geological Survey. For the final trend analysis in the base period, only 49 sites met the criteria. The sites are divided among the three different climatic parts of the study area, the western, central, and eastern parts. Eight of the sites were in the western part, 31 sites were in the central part, and 10 sites were in the eastern part of the study area.

From 1993 to 2004, FA trends in TN at 19 of the 24 sites that had sufficient data for analysis showed a tendency toward increasing concentrations. The 19 sites with increasing TN concentrations were distributed throughout the entire study area; however, only 8 of these sites had significant upward trends, and they were in the central part of the study area. These upward FA trends indicate that the effects of anthropogenic activities are continuing to increase in these areas and are contributing additional TN to the streams. Only two of the five sites with a downward tendency in concentrations had significant trends. The patterns in these FA trends in TN concentrations are similar to the changes in fertilizer nitrogen applications. The similarity in the patterns of these changes indicates a broad regional association between FA trends in TN concentrations and fertilizer applications. However, the significant upward FA trends in TN concentrations at five agricultural sites and upward trends at several of the others indicate that efforts to reduce runoff from agricultural lands have not been universally successful. At least two reasons account for that lack of success—the first is that local benefits do not necessarily translate to basinwide benefits, and the second is that unregulated activities such as tile drainage may counteract any benefit from efforts to reduce agricultural runoff.

The OA trends in TN concentrations from 1993 to 2004 indicated a general tendency toward increasing concentrations at 16 of the 24 sites; however, only 3 of these sites had significant upward trends; 2 sites were in southern Wisconsin in the central part of the study area, and 1 site was in the western part of the study area. Of the eight sites with a tendency toward decreasing concentrations, only two sites had significant downward trends and they are the same two sites that had downward FA trends. The sites with downward trends in concentrations were scattered throughout the study area.

From 1993 to 2004, TN loads at 17 of the 24 sites that had sufficient data for analysis indicated a tendency toward decreasing loads. Most of the sites with a downward trend were in the central or western parts of the study area. Eight of these sites had significant downward trends in loads; most of those sites had significant downward trends in streamflow.

The only exception was the site in the eastern North Dakota, which had a significant downward trend in loads and no trend in streamflow. Only one of the seven sites with an upward trend in TN loads had a significant trend.

The FA trends in NH4 concentrations had a widespread downward tendency from 1993 to 2004. Of the 36 sites that had sufficient streamflow and NH4 concentration data for trend analysis, 32 sites had a downward trend in concentrations. Of these 32 sites, 24 sites had significant downward trends (at $p < 0.10$). Only four sites had an upward FA trend in NH4 concentrations, of which only one site, Northrup Creek, N.Y. (0422026250), had a significant upward FA trend in concentrations; however, the magnitude of this upward trend was only 0.002 mg/L/yr. The widespread, downward trends in NH4 concentrations for the trend sites indicate that some of the goals of the Clean Water Act are being met. There is, regionally, good agreement between FA trends in NH4 concentrations and changes in manure nitrogen, except in the farthest western part of the study area.

The OA trends in NH4 concentrations from 1993 to 2004 were similar to those of the FA trends. The OA trends in NH4 concentrations at 32 of the 36 sites that had sufficient data showed a general tendency toward decreasing concentrations, and the downward trends were statistically significant ($p < 0.10$) at 23 of these sites. Of the remaining four sites with upward OA trends in concentrations, the only site with a significant trend was again Northrup Creek, N.Y.

From 1993 to 2004, NH4 loads at 32 of the 36 sites that had sufficient data for analysis indicated a tendency toward decreasing loads. These sites were distributed across all three parts of study area. Twenty-six of these sites had significant downward trends in loads ($p < 0.10$), and most of these sites had significant downward trends in both FA concentrations and streamflow.

The FA trends in KJN concentrations from 1993 to 2004 showed no distinct spatial pattern. Of the 36 sites that had sufficient streamflow and KJN concentration data for analysis, 17 sites had downward trends, and 19 sites had upward trends in FA concentrations. Seven sites had significant downward trends, and six sites had significant upward FA trends. The increase in human population and apparent decrease in animal manure production are competing factors that confound the interpretation of trend in KJN. Other confounding factors are the changes in treatments of those human and animal wastes.

When the OA trends in KJN concentrations from 1993 to 2004 were examined, no distinct spatial patterns were evident; however, the tendencies and statistical results were somewhat different than those of the FA trends. The differences in the results of the FA and OA trend analyses were attributable to differences in the trends in streamflow. Of the 36 sites that had sufficient streamflow and KJN concentration data for analysis, 18 sites had downward trends, and 18 sites had upward trends in KJN concentrations. Of these, nine sites had significant downward trends and six sites had significant upward trends ($p < 0.10$).

The KJN loads from 1993 to 2004 had a downward trend at 30 sites and an upward trend at 6 sites, of the 36 sites that had sufficient data for analysis. The sites with downward trends in KJN loads were distributed across all three parts of study area; three of the six sites with upward trends were in the eastern part of the study area. Thirteen of the sites with downward trends in loads had significant trends, whereas only one of the six sites with upward trends in loads had a significant trend ($p < 0.10$).

The FA trends in NO2NO3 concentrations from 1993 to 2004 showed no distinct spatial pattern. Of the 29 sites that had sufficient streamflow and NO2NO3 concentration data for analysis, 12 sites had downward trends, and 17 sites had upward trends in concentrations. Of these sites, three sites—one in North Dakota, one in Illinois, and one in Indiana—had significant downward trends, and four sites—one in Iowa, one in Illinois, and two in Wisconsin—had significant upward trends ($p < 0.10$). Some general agreement exists between the FA trends in NO2NO3 concentrations and the change in nitrogen fertilizer application. The small changes in the forested sites in northern Wisconsin are consistent with the changes in atmospheric deposition of nitrogen.

When the OA trends in NO2NO3 concentrations from 1993 to 2004 were examined, no distinct spatial patterns were evident; however, tendencies and trend results were somewhat different than those of the FA trends. These differences in the results of the FA and OA trend analyses are attributable to differences in streamflow trends. Of the 29 sites that had sufficient streamflow and NO2NO3 concentration data for analysis, 13 sites had downward trends, and 16 sites had upward trends in concentrations. Of these sites, four sites had significant downward trends, and three sites had significant upward trends ($p < 0.10$).

Of the 29 sites that had sufficient data from 1993 to 2004 for analysis, 16 sites had downward trends, and 13 sites had upward trends in NO2NO3 loads. Seven of the sites with downward trends in loads had significant trends, whereas only one of the 12 sites with upward trends in loads had a significant trend ($p < 0.10$).

From 1993 to 2004, FA trends in TP concentrations at 24 of the 40 sites that had sufficient data for analysis were upward. The 24 sites that had increasing TP concentrations were distributed throughout the entire study area; however, most of these sites were in the central part of the study area and the eastern side of the western part. Twelve of these 24 sites had significant upward trends ($p < 0.10$). These upward FA trends indicate that the effects of anthropogenic activities are continuing to increase in these areas and are contributing additional TP to the streams. Downward FA trends in concentrations at 7 of the 16 sites were significant ($p < 0.10$) and indicate that the effects of anthropogenic activities are decreasing. Changes in the amount of phosphorus fertilizer use are consistent with FA trends in TP concentrations at four sites (04073468, 05427948, 054310157, and 0422026250), all of which had downward trends. However, phosphorus fertilizer use was negatively associated with the FA trends in

TP concentrations at six sites ($p < 0.05$), which contradicts its use in explaining the observed trend at these sites. The lack of a strong association between phosphorus fertilizer use and FA trends in TP concentrations indicates that other factors are important in determining TP concentrations in streams.

The OA trends in TP concentrations from 1993 to 2004 showed a general tendency toward increasing concentrations at 21 of the 41 sites; however, the upward trends were significant at only 7 of these sites ($p < 0.10$). Of the 20 sites with a downward trend, only 7 sites had significant trends. The sites with upward or downward trends were scattered throughout the study area.

From 1993 to 2004, trends in TP loads at 30 of the 41 sites that had sufficient data for analysis were downward. Twelve of these sites, all within the central and eastern parts of the study area, had significant ($p < 0.10$) downward trends in loads. Most of these sites had significant downward trends in streamflow. The upward trends were statistically significant ($p < 0.10$) at only 2 of the 11 sites. A comparison of the trends in loads and FA trends in concentrations indicates that most of the trends in loads were driven primarily by streamflow rather than changes in concentration, except in southern Wisconsin where trends in both streamflow and concentrations were significantly downward.

The FA trends in DP concentrations from 1993 to 2004 showed mixed regional tendencies; however, DP concentrations consistently decreased at all of sites in the eastern part of the study area. Of the 35 sites that had sufficient streamflow and DP concentration data for analysis, 23 sites had downward trends in concentrations. Of these sites, 12 sites had significant downward trends, and the trends at all 7 of the sites in the eastern part of the study area were significant. Five sites, all in the central part of the study area, had significant upward trends ($p < 0.10$). A comparison of the patterns of FA trends in DP concentrations with those of nonpoint phosphorus inputs indicates fairly good regional agreement indicating that the reduction in fertilizers and manure east of the Mississippi River influences observed trends in DP concentrations. Some discrepancies exist in Illinois, but those probably are caused by nonagricultural influences.

When the OA trends in DP concentrations from 1993 to 2004 were examined, spatial trends followed the same general pattern of decreasing concentrations in the eastern areas and increasing and decreasing concentrations in the central and western parts of the study area. Several differences between the FA and OA trend analyses were found for specific sites in the central part of the study area, where many more sites had significant downward OA trends than downward FA trends. These differences in the results of the FA and OA trend analyses are attributable to the significant trends in streamflow. Of the 35 sites that had sufficient streamflow and DP data for analysis, 26 sites had downward trends, and 8 sites had upward trends in concentrations. Of these sites, 15 sites had significantly downward trends, and 4 sites in Illinois had significant upward trends ($p < 0.10$).

From 1993 to 2004, trends in DP loads at 32 of the 35 sites that had sufficient data for analysis were downward; trends at 15 of these sites were significant ($p < 0.10$). Upward trends in DP loads at two of the three sites, all located in northern Illinois, were significant.

The FA trends in SMAT and SS concentrations from 1993 to 2004 showed distinct spatial patterns; concentrations tended to increase in Illinois and to decrease throughout most of Wisconsin, Iowa, and the eastern part of the study area. The main difference was that some of the downward trends observed for the SMAT data in Wisconsin and New York and upward trends in Illinois were not observed for SS data because most of these sites only had total suspended solids data. In addition, when the total suspended solids data were removed from the data from the Popple River in northern Wisconsin, the upward trend in SS became significant. Of the 42 sites that had sufficient streamflow and SMAT concentration data for analysis, 25 sites had downward trends, and 17 sites had upward trends in concentrations. Of these sites, 12 sites had significant downward trends and 5 sites had significant upward trends ($p < 0.10$). Of the 22 sites that had sufficient streamflow and SS concentration data for analysis, 15 sites had downward trends, of which 7 trends were statistically significant, and 7 sites had upward trends in concentrations, of which 3 trends were statistically significant ($p < 0.10$).

The spatial pattern in the OA trends in SMAT and SS concentrations generally was the same as that for the FA trends; however, several of the sites with non-significant trends, especially in Iowa, had significant downward OA trends ($p < 0.10$). Of the 42 sites that had sufficient streamflow and SMAT concentration data for trend analysis, 30 sites had downward trends, and 12 sites had upward trends in concentrations. Of these sites, 18 sites had significant downward trends, and 5 sites had significant upward trends. The sites with significant upward trends in concentrations were in Illinois and North Dakota.

From 1993 to 2004, SMAT and SS loads indicated a tendency toward decreasing loads. SMAT loads had downward trends at 36 sites and upward trends at 6 sites. Nineteen of the 22 sites had a downward trend in SS loads. Twenty of the sites, throughout most of the study area, with a downward trend in SMAT loads had significant trends, and three sites in northern Illinois and North Dakota had a significant upward trend in loads. Once again the pattern in the load trends resembles that for streamflow.

Changes in TN, TP, and SMAT loads were examined from 1975 to 2003 at six sites. These changes were examined to provide a longer term perspective on the data examined from 1993 to 2004. The long-term, nonmonotonic trends were described by LOESS smoothing. The statistical significance of those trends cannot be determined; however, the long-term changes for annual streamflow and load data indicate that the monotonic trends from 1993 to 2004 should not be extrapolated backward in time.

Most of the changes in long-term loads are directly caused by the change in streamflow rather than a change in nutrient or sediment concentrations. The long-term changes in concentrations were much smaller than the long-term changes in streamflow. This results in the long-term changes in loads directly resembling the long-term changes in streamflow. The most prominent example of this is in data for streamflow and TN, TP, and SMAT loads at the Souris River, N. Dak., from 1975 to 2003.

The insights provided by examining trends at sites that have a long history of streamflow and water-quality monitoring demonstrate the value of establishing and maintaining a long-term streamflow and water-quality monitoring network for planning purposes in the United States, especially because current thought (2007) is that global warming could lead to large-scale changes in the climate of North America during the 21st century. The strong influence of streamflow on nutrient and sediment loads can result in long-term trends caused by changes in climate rather than changes caused by anthropogenic changes. In general, such large-scale regional changes in climate and associated hydrology could necessitate modifications to infrastructure if changes occur in lake levels, flood crests, and flood frequency, or if lesser future rainfall in some areas places unanticipated strains on water use and sewerages.

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