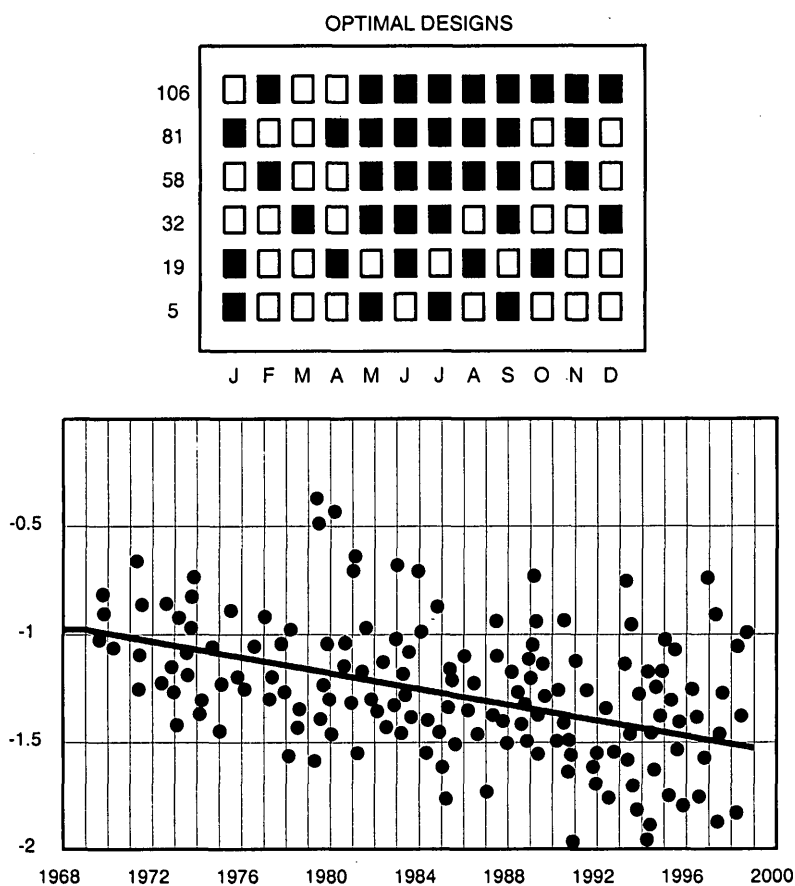


Water-Quality Trend Analysis and Sampling Design for Streams in Connecticut, 1968–98

Water-Resources Investigations Report 02-4011



Prepared in cooperation with the
Connecticut Department of Environmental Protection

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

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By Elaine C. Todd Trench and Aldo V. Vecchia

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**Prepared in cooperation with the
Connecticut Department of Environmental Protection**

**East Hartford, Connecticut
2002**

U.S. DEPARTMENT OF THE INTERIOR

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U.S. GEOLOGICAL SURVEY

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CONVERSION FACTORS, VERTICAL DATUM, AND ABBREVIATIONS

Multiply	By	To obtain
Length		
foot (ft)	0.3048	meter
mile (mi)	1.609	kilometer
Area		
square mile (mi ²)	2.590	square kilometer
Flow rate		
cubic foot per second (ft ³ /s)	0.02832	cubic meter per second
cubic foot per second per square mile (ft ³ /s/mi ²)	0.01093	cubic meter per second per square kilometer
Volume		
cubic foot (ft ³)	0.02832	cubic meter
million gallons (Mgal)	3,785	cubic meter
Mass		
ton	0.9072	megagram

Temperature in degrees Celsius (°C) may be converted to degrees Fahrenheit (°F) as follows:

$$^{\circ}\text{F} = (1.8 \times ^{\circ}\text{C}) + 32$$

Temperature in degrees Fahrenheit (°F) may be converted to degrees Celsius (°C) as follows:

$$^{\circ}\text{C} = (^{\circ}\text{F} - 32) / 1.8$$

Altitude, as used in this report, refers to distance above or below sea level.

Concentration of chemical constituents is given in milligrams per liter (mg/L).

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By Elaine C. Todd Trench and Aldo V. Vecchia

ABSTRACT

Data from five water-quality stations, part of a larger network of 34 stations throughout Connecticut, were analyzed for trends in dissolved chloride, dissolved sulfate, total nitrogen, total phosphorus, total organic carbon, and turbidity using time-series analysis. The five stations, on the Connecticut, Salmon, Saugatuck, Quinnipiac, and Naugatuck Rivers, all have more than 20 years of water-quality record, and their drainage areas represent major land uses and hydrologic conditions in Connecticut. The five drainage basins range in size from about 20 mi² to almost 10,000 mi².

Long-term or short-term trends were detected in the flow-adjusted data for all constituents at all stations evaluated in this study. Chloride concentrations generally increased in all five streams during the period of the study. Significant decreases in dissolved sulfate concentrations were detected in the data for all five streams during one or more time periods within the period of record. Concentrations of total nitrogen in the Connecticut, Naugatuck, Salmon, and Saugatuck Rivers increased during different periods from the mid-1970's to the mid- or late 1980's, and then decreased to a time in the early 1990's. The most pronounced change in total phosphorus concentrations took place on the Connecticut River, where a highly significant downward trend was detected for the period of the study. Concentrations of total organic carbon on the Naugatuck River decreased substantially during the 1970's and early 1980's and fluctuated substantially in the 1980's and 1990's. With the exception of the Naugatuck River, no large overall trends in total organic carbon were detected during the 1980's and

1990's. Graphs of turbidity data for all five stations display a sawtooth pattern of decreases and increases over the period of record. Positive step trends in total organic carbon and turbidity in the 1970's and early 1980's may be related to method changes.

The statistical time-series model was used to evaluate the efficiency of various sampling designs for monitoring trends in water quality. Optimal sampling designs for the Connecticut, Salmon, Saugatuck, Quinnipiac, and Naugatuck Rivers were identified for each of six sampling frequencies ranging from 4 through 9 samples per year, based on a design period of 5 years.

Sampling designs with reasonable power to detect trends in the constituents evaluated can be identified at the sampling frequency of 8 samples per year; however, the timing of samples for the 8-sample designs differs among stations and constituents.

Results from the design analysis indicate that monthly sampling from approximately April to September (with two winter samples) will meet general data requirements for long-term trend analysis at some stations, including the Connecticut and Quinnipiac River stations, but may not be well-suited for the detection of trends in all situations. Monthly sampling in the winter and spring months yields more information for trend analysis on the Naugatuck, Salmon, and Saugatuck Rivers. This difference is largely due to the effects of one constituent, sulfate, on the design requirements for these three stations. Trend detection for total nitrogen also benefits from additional winter sampling on the Connecticut, Salmon, and Saugatuck Rivers.

The greatest difference between the optimal designs and the existing monitoring program is in the current schedule for the Salmon and Saugatuck Rivers. Quarterly sampling designs for these two rivers have very low power for trend detection for all constituents evaluated, and all designs with less than 8 samples per year have low power for trend detection for constituents other than chloride or sulfate.

INTRODUCTION

The U.S. Geological Survey (USGS) and the State of Connecticut have worked together since 1955 to monitor and interpret the water quality of Connecticut's rivers and streams. An expanded cooperative water-quality monitoring program with the Connecticut Department of Environmental Protection (CTDEP) began in 1974, in response to the passage of Connecticut's Clean Water Act in 1967 and the Federal Water Pollution Control Act in 1972.

Protecting and maintaining good water quality in Connecticut streams is important for human health, ecological integrity, and aesthetic and economic benefits. Management decisions of increasing complexity require a monitoring program that provides data for multiple purposes, including trend analysis to detect improvement or deterioration in water quality over time. A USGS study of trends in surface-water quality in Connecticut, covering the period 1969–88 (Trench, 1996), reported a number of significant trends in water-quality constituents and physical properties. Some trends indicated improvement in water quality, whereas other trends indicated deterioration. Periodic assessment of water-quality trends is necessary to determine if constituent concentrations have a continuing trend in the same direction, have changed direction, or remain constant at some level. Additionally, as statistical methods for trend analysis and interpretation become more sophisticated and flexible, different approaches to trend analysis can provide new insights into recent and long-term water-quality conditions. As part of the continuing effort to study water quality in Connecticut, the USGS and CTDEP began a cooperative project in 1998 to study long-term trends in water quality and evaluate sampling designs.

Purpose and Scope

This report presents results of an analysis of long-term trends in the water quality of selected

streams in Connecticut and an evaluation of sampling designs for monitoring future trends. Trends were analyzed and sampling frequencies were evaluated using a statistical time-series model developed and described by Vecchia (2000), elements of which are summarized in this report.

A thorough interpretation of how the detected trends relate to hydrogeology, land use, population distribution, hydrologic modifications, and pollution sources is beyond the scope of this report; however, some supporting information is presented to provide perspective on the detected trends and to point toward possibilities for further analysis.

This report presents an analysis of long-term data for five water-quality stations that are part of a network of 34 monitoring stations throughout Connecticut. The five stations selected for trend analysis all have more than 20 years of water-quality record, and their drainage areas represent a variety of land uses and hydrogeologic conditions. Water-quality records for these stations were retrieved from the USGS National Water Information System (NWIS) and were analyzed for trends in selected chemical constituents and physical properties for selected periods of record during water years 1968–98.

Previous Investigations

Major issues in choosing appropriate trend-detection procedures for water-quality data have been discussed and summarized by Hirsch and others (1991). Nonparametric tests for water-quality trend analysis have been developed and applied by Hirsch and others (1982), Smith and others (1982), and Schertz and others (1991).

A USGS summary of national streamwater-quality conditions and trends in all 50 states included trend analysis for selected constituents and stations in Connecticut for the period 1970–89 (Paulson and others, 1993). Water-quality trends in Connecticut streams for the period 1969–88 were analyzed by Trench (1996). Trends in nutrients in Connecticut streams were analyzed by Zimmerman and others (1996), Zimmerman (1997), and Trench (2000) for varying periods of record from the 1970's to 1995.

Hipel (1985) reviewed time-series analysis, a parametric procedure, in water-resources applications. Application of time-series models with periodic autoregressive and moving average components (PARMA models) to water-resources time series has been inves-

tigated by Vecchia (1985). A national team of hydrologists and statisticians in the USGS National Water-Quality Assessment (NAWQA) program developed time-series models for analysis of trends and applied these models to data sets for 34 water-quality monitoring stations. The selected stations were located throughout the continental United States and had at least 20 years of water-quality data during the period 1966–95. The primary goal of the study was to evaluate the effectiveness of various sampling designs for detection of long-term, non-monotonic trends in stream quality. Two Connecticut streams, the Salmon River and the Connecticut River at Thompsonville, were included in the national study. The national study determined that no single sampling design is optimal for all stations and constituents, even in a small state such as Connecticut. Time-series analysis of trends and analysis of sampling designs have been applied to water-quality data for the Souris River in North Dakota (Vecchia, 2000).

Acknowledgments

The CTDEP, a long-time cooperator of the USGS on water-quality projects, took an interest in the potential for time-series analysis to evaluate the efficiency of water-quality sampling designs. Roy Fredricksen of the CTDEP provided information on changes to municipal wastewater-treatment facilities in the project area during the study period. Laura Medalie and Brent Troutman of the USGS provided colleague reviews for the report. Barbara Korzendorfer of the USGS provided editorial review and prepared the report for publication.

DESCRIPTION OF THE STUDY AREA

Connecticut is a coastal state in southern New England with a land area of 5,009 mi² (fig. 1). Major streams generally flow from north to south into Long Island Sound. The Connecticut River is the largest stream in New England, with a drainage area that extends from Quebec to Long Island Sound.

The State consists primarily of eastern and western upland areas that are separated by a central lowland and bordered by a coastal lowland. Areas north of Connecticut are generally hilly or mountainous. Unconsolidated glacial deposits of varying thickness blanket the bedrock surface throughout most of Connecticut. The climate in Connecticut is generally temperate and humid, with precipitation distributed evenly throughout the year.

Streamflow in Connecticut varies considerably throughout the year in response to precipitation, evapotranspiration, and snowmelt conditions. Although high-flow conditions generally take place in the spring, and low-flow conditions generally take place in late summer or early fall, major flooding can occur at any time of year. Streamflow also varies from year to year in response to varying climatic conditions.

Connecticut ranked fourth highest of the 50 states in population density in 1985. The most highly urbanized areas are in the central lowland and the southwestern coastal lowland. The period covered by this trend study coincides with a period of population growth, changes in the geographic distribution of population, major land-use changes, and intensive management activity related to water-pollution and air-pollution control. Human activity affects the quality of surface water throughout Connecticut, even in relatively undeveloped areas.

DATA SELECTION

Monitoring stations and constituents for time-series trend analysis were selected in consultation with the CTDEP. Stations were selected to represent major hydrogeologic settings, land uses, and pollutant sources in the State. Chemical constituents and physical properties were reviewed to select a small number that would represent major constituent groups as well as critical water-quality issues in the State.

Selection of Monitoring Stations

Five water-quality stations, part of a larger network of 34 stations throughout Connecticut, were selected for trend analysis—Connecticut River at Thompsonville, Salmon River near East Hampton, Quinnipiac River at Wallingford, Naugatuck River at Beacon Falls, and Saugatuck River near Redding (fig. 1, table 1). The five stations selected for trend analysis all have more than 20 years of water-quality record, and their drainage areas represent the major land uses and hydrogeologic conditions in Connecticut. The five drainage basins range in size from about 20 mi² to almost 10,000 mi². The largest drainage basin (Connecticut River) has a multi-state drainage area, and the other four basins are entirely within Connecticut. Water-quality monitoring for various properties and constituents was initiated in these drainage basins at various times during 1966–74.

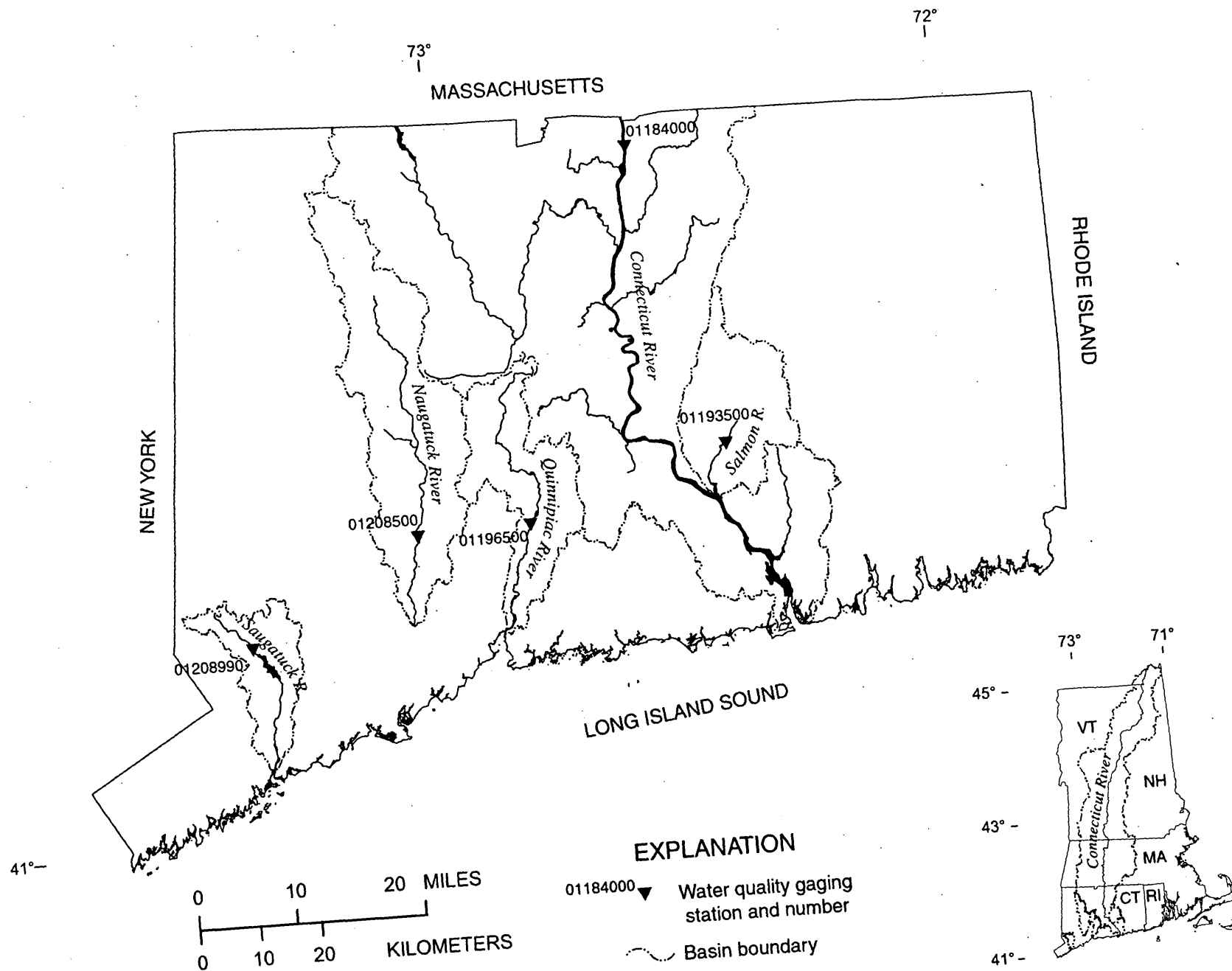


Figure 1. Drainage basins of streams in Connecticut selected for trend analysis.

Table 1. Water-quality sampling and discharge information for drainage basins evaluated using time-series analysis

[Sampling frequency changed from monthly to 8 times per year starting in 1993. M, monthly; 8, 8 times per year (monthly in summer and bimonthly in winter); Q, quarterly; /, indicates change in sampling frequency]

Map reference number	Water-quality and streamflow-gaging station number	Water-quality and streamflow-gaging station name	Drainage area at monitoring station (square miles)	Latitude	Longitude	Period of water-quality record (water years)	Sampling frequency
1	01184000	Connecticut River at Thompsonville, Conn.	9,660	41°59'14"	72°36'21"	1966–98	M/8
2	01193500	Salmon River near East Hampton, Conn.	100	41°33'08"	72°26'59"	1968–98	M/8/Q
3	01196500	Quinnipiac River at Wallingford, Conn.	115	41°26'58"	72°50'29"	1968–98	M/8
4	01208500	Naugatuck River at Beacon Falls, Conn.	260	41°26'32"	73°03'47"	1974–98	M/8
5	01208990	Saugatuck River near Redding, Conn.	21.0	41°17'40"	73°23'44"	1968–98	M/8/Q

The Connecticut River, with a drainage area of 9,660 mi² at the Thompsonville monitoring station, was selected for analysis because of the importance of this freshwater resource to the State of Connecticut and Long Island Sound. The drainage basin incorporates parts of four states and includes large undeveloped forested areas, agricultural land in some upland areas and along the Connecticut valley, and major urban areas in central Massachusetts (table 2).

The Salmon River and Saugatuck River were selected to represent relatively undeveloped, small drainage basins with no point sources. Both drainage basins, although primarily forested, are undergoing suburban development. The Salmon River, with a drainage area of 100 mi², is in the eastern uplands of Connecticut and is a tributary to the tidal reach of the Connecticut River. The Saugatuck River, with a drainage area of 21 mi², is in the western uplands of Connecticut, near the highly developed southwestern coastal area of the State.

The Quinnipiac River and the Naugatuck River were selected to represent urbanized drainage basins

with major point discharges. The Quinnipiac River, with a drainage area of 115 mi², is in the central lowland of Connecticut, in an area of generally low relief and extensive floodplains underlain by thick layers of stratified glacial deposits. The Naugatuck River, with a drainage area of 260 mi², is in the western uplands of Connecticut, in an area of steep bedrock hills with thin glacial deposits. Both rivers have historically received point discharges from major urban areas.

Selection of Properties and Constituents

Water-quality records for the five stations selected for trend analysis were retrieved from the USGS Water-Quality Data Base (QWDATA) of the National Water Information System (NWIS). Data were analyzed for trends in selected chemical constituents and physical properties for periods of record during water years 1968–98. Five chemical constituents and one physical property were selected for analysis in consultation with CTDEP (table 3).

Table 2. Land-use characteristics of drainage basins selected for trend analysis

[Data compiled by J.R. Mullaney, U.S. Geological Survey, written commun., 2001]

Map reference number	Water-quality station number	Water-quality station name	Drainage area (square miles)	Land-use and land-cover category, as a percentage of basin area					
				Urban	Agricultural	Forested			
						Forest	Water	Wetlands	Other
1	01184000	Connecticut River at Thompsonville, Conn.	9,660	4.3	8.4	79.4	2.2	4.6	0.9
2	01193500	Salmon River near East Hampton, Conn.	100	8.4	12.3	68.2	1.5	9.1	0.6
3	01196500	Quinnipiac River at Wallingford, Conn.	115	50.4	0.6	40.3	1.7	5.9	1.0
4	01208500	Naugatuck River at Beacon Falls, Conn.	260	21.1	10.2	59.3	2.3	6.6	0.5
5	01208990	Saugatuck River near Redding, Conn.	21.0	8.1	4.5	79.9	2.6	5.0	0.1

Table 3. Water-quality constituents and properties selected for trend analysis in Connecticut

[Parameter code is a five-digit number used to uniquely identify a specific constituent. NWIS, National Water Information System; mg/L, milligram per liter; NTU, nephelometric turbidity units]

Water-quality constituent or property	Unit	NWIS parameter code
Chemical Constituents		
Chloride, dissolved	mg/L	00940
Sulfate, dissolved	mg/L	00945
Nitrogen, total	mg/L	00600
Phosphorus, total	mg/L	00665
Carbon, organic, total	mg/L	00680
Physical Properties		
Turbidity	NTU	00076

Data Selection and Evaluation

The characteristics of all data sets were reviewed prior to trend analysis. Summary statistics were calculated for each constituent (table 4), and percentages of censored data values (data below detection limits) were calculated.

Minor modifications to the total nitrogen data retrieved from QWDATA were necessary to create representative data sets, because total nitrogen is not an analyzed constituent. Total nitrogen is calculated in QWDATA by adding the analytical value for nitrite-plus-nitrate to the analytical value for Kjeldahl

nitrogen. If either of these two constituents is censored, a total nitrogen value is not calculated by the QWDATA program. For streams with high nutrient concentrations, this does not significantly affect the distribution of values in the data set. For streams with low nutrient concentrations, the absence of calculated total nitrogen values at low concentrations may bias the data set toward high concentrations that are not fully representative of the water quality. To overcome this potential bias, a total nitrogen value was calculated and added to the data set where one or both component constituents were censored, using an approach outlined by D.K. Mueller (U.S. Geological Survey, written commun., 1998) and described by Trench (2000, p. 16).

Effects of Method Changes on Historical Data

The period of record for the Connecticut trend study encompasses three decades of improvement and increasing sophistication in water-quality sampling and analytical methods. During this period, understanding of the ways in which sampling location, equipment, field methods, and laboratory analytical methods affect water-quality data has increased markedly. Although the need for documenting any changes in these factors is clearly understood now, the importance of such changes was not always apparent in the earlier years of water-quality data collection, and changes have not always been adequately documented.

Table 4. Summary statistics for constituents used in trend analysis

[All stations are in Connecticut. mg/L, milligrams per liter; NTU, nephelometric turbidity units; (e), estimated]

Map reference number	Water-quality station name	Period of record	Number of samples	Concentration percentiles (mg/L)		
				25th	50th (median)	75th
Chloride						
1	Connecticut River at Thompsonville	10/66-9/98	358	8.6	11.0	13.0
2	Salmon River near East Hampton	6/68-7/98	317	11.0	13.0	15.0
3	Quinnipiac River at Wallingford	6/68-9/98	329	24.0	29.0	34.0
4	Naugatuck River at Beacon Falls	7/74-9/98	274	29.0	41.0	57.0
5	Saugatuck River near Redding	6/68-7/98	318	11.0	14.0	18.0
Sulfate						
1	Connecticut River at Thompsonville	10/66-9/98	356	9.0	11.0	12.0
2	Salmon River near East Hampton	6/68-7/98	286	9.4	11.0	12.0
3	Quinnipiac River at Wallingford	6/68-9/98	297	19.0	22.0	27.0
4	Naugatuck River at Beacon Falls	7/74-9/98	244	21.0	28.0	40.0
5	Saugatuck River near Redding	6/68-7/98	286	10.0	12.0	15.0
Total Nitrogen						
1	Connecticut River at Thompsonville	10/66-9/98	396	.50	.73	.96
2	Salmon River near East Hampton	6/68-7/98	322	0.0	.45	.70
3	Quinnipiac River at Wallingford	6/68-9/98	360	2.2	3.0	3.9
4	Naugatuck River at Beacon Falls	7/74-9/98	306	1.9	3.0	5.0
5	Saugatuck River near Redding	6/68-7/98	319	0.0	.33	.51
Total Phosphorus						
1	Connecticut River at Thompsonville	10/66-9/98	362	.04	.06	.09
2	Salmon River near East Hampton	6/68-7/98	282	(e) .007	(e) .010	(e) .020
3	Quinnipiac River at Wallingford	6/68-9/98	320	.27	.42	.61
4	Naugatuck River at Beacon Falls	7/74-9/98	287	.21	.39	.67
5	Saugatuck River near Redding	6/68-7/98	280	(e) .01	(e) .02	(e) .03
Total Organic Carbon						
1	Connecticut River at Thompsonville	10/66-9/98	255	3.4	4.0	5.0
2	Salmon River near East Hampton	6/68-7/98	250	3.0	3.9	5.0
3	Quinnipiac River at Wallingford	6/68-9/98	276	3.9	4.8	6.3
4	Naugatuck River at Beacon Falls	7/74-9/98	262	4.3	6.0	7.6
5	Saugatuck River near Redding	6/68-7/98	251	3.5	4.5	5.8
Turbidity Data percentiles (NTU)						
1	Connecticut River at Thompsonville	10/66-9/98	239	1.2	2.0	3.0
2	Salmon River near East Hampton	6/68-7/98	196	.80	1.0	1.5
3	Quinnipiac River at Wallingford	6/68-9/98	230	2.2	3.6	6.0
4	Naugatuck River at Beacon Falls	7/74-9/98	225	1.8	2.6	4.0
5	Saugatuck River near Redding	6/68-7/98	198	.90	1.1	2.0

The effects of changes in field or laboratory methods need to be taken into account in evaluating the results of trend analysis. Where information on method changes is available, the dates of such changes have been considered in selecting beginning and ending dates for trend periods and in interpreting trend results. Constituents for which method changes are known to have taken place include chloride, sulfate, total nitrogen, and total phosphorus.

Chloride

Laboratory analytical methods for chloride were changed in April 1990, December 1992, and January 1994; no perceptible change in data quality was expected from the 1994 method changes (P.F. Rogerson, U.S. Geological Survey, written commun., 1993). A slight positive bias was detected in results for chloride during the period from April 1990 to October 1992 (C.J. Patton and others, U.S. Geological Survey, written commun., 1992).

Sulfate

Reported concentrations of dissolved sulfate may be biased high by an unknown amount because of an analytical method interference problem during 1983–90 (Fishman and others, 1994, p. 49; D.A. Rickert, U.S. Geological Survey, written commun., 1989; W.E. Webb, U.S. Geological Survey, written commun., 1989). The magnitude of the bias is likely to be greater in samples with low sulfate concentrations and high natural color or turbidity. Method correction procedures were undertaken during 1989, and a new method was implemented mid-year in 1990 (W.E. Webb, U.S. Geological Survey, written commun., 1989; P.F. Rogerson, U.S. Geological Survey, written commun., 1993). Consequently, the period 1983–89 has been selected to represent the period of method bias in the current study. A positive bias for sulfate determinations has also been reported for the period from April 1990 to March 1992 (Alexander and others, 1996).

Total Nitrogen

Reported concentrations of total nitrogen may have been affected by the laboratory method used to analyze Kjeldahl nitrogen (ammonia-plus-organic nitrogen) from 1986 to October 1, 1991. A change in the digestion step that is part of the method used to analyze Kjeldahl nitrogen was implemented on October 1, 1991, at the USGS National Water Quality

Laboratory (NWQL). Statistical analysis of paired data analyzed using the old and new methods indicates that median concentrations of Kjeldahl nitrogen, and consequently total nitrogen, determined prior to October 1, 1991, are generally (but not uniformly) biased high by about 0.1 mg/L (Patton and Truitt, 2000, p. 1). The concentration of other nitrogen constituents appears to affect the extent of the bias. A full discussion of this method change is presented by Patton and Truitt (2000). For urban streams in Connecticut with high total nitrogen concentrations, the bias resulting from the Kjeldahl nitrogen method change represents a small percentage of the concentration, and may not have a substantial effect on trend results. For cleaner streams with low total nitrogen concentrations, the bias represents a larger percentage of the total nitrogen concentration, and effects on trend results are likely to be more pronounced. Additional information on method changes that have affected the component constituents of total nitrogen has been summarized by Zimmerman and others (1996, p. 29).

Total Phosphorus

Improvements in analytical methods for total phosphorus were implemented by the USGS NWQL in 1990 and 1991 to eliminate bias. Total phosphorus data prior to October 1, 1991 tend to be negatively biased; that is, the reported analytical concentrations probably are lower than actual environmental concentrations under certain conditions (D.A. Rickert, U.S. Geological Survey, written commun., 1992). The negative bias probably is greater in data produced prior to May 1, 1990. National statistical analyses have shown that the negative bias in total phosphorus data increases in samples with high concentrations of particulate phosphorus, suspended sediment, and organic carbon. Additional discussion of the possible implications of this bias for the analysis and interpretation of phosphorus data from Connecticut streams is presented by Zimmerman and others (1996, p. 29–30).

TIME-SERIES METHODS FOR TREND ANALYSIS AND SAMPLING DESIGN

A set of observations of a monitored variable, arranged chronologically, is called a time series. Time-series analysis is a process of fitting a time-series model to a time series of observations, such as water-quality data. Purposes of time-series analysis can include forecasting, generating synthetic sequences of

data for use in simulation studies, or investigating and modeling the underlying characteristics of a system (Hipel, 1985, p. 609). Time-series analysis is used extensively in many fields, and has become an important form of analysis in water resources. A time-series model for water-quality trend analysis has been developed by the USGS NAWQA as an alternative to other trend-analysis methods currently in use.

Comparison of Time-Series Analysis with Non-Parametric Methods

A widely used procedure for detecting trends in hydrologic time series is seasonal Kendall's tau (Hirsch and Slack, 1984). Seasonal Kendall's tau is a nonparametric trend test that is independent of the probability distribution of the water-quality data. The seasonal Kendall test adjusts for seasonality in the mean, variance, and autocorrelations; allows for a moderate degree of missing or censored data; and can be used to screen a large number of stations and constituents for the presence of monotonic (one-directional) trends in water quality.

Although the seasonal Kendall test is a powerful tool for detecting trends, several drawbacks to the procedure limit its usefulness in certain situations. For example, the test is not suited for data with highly irregular sampling frequencies, which is commonly the case with water-quality data sets. When applying the seasonal Kendall test, some data may be discarded because of the necessity to select a constant sampling frequency over the whole period of record. The test is designed to detect monotonic trends (uniform increases or decreases in a water-quality variable), whereas many water-quality time series have nonseasonal cyclic trends or other complex variability.

Time-series analysis can be used to evaluate data for nonmonotonic trends—trends that have one or more changes in slope during the period being evaluated—and to detect cyclic trends. The approach also can handle data sets with missing data and variable sampling frequencies. Time-series analysis uses all water-quality information, even if sampling frequencies have changed one or more times during the period of record.

Time-series analysis of trends is more technically demanding and computationally intensive than

the seasonal Kendall test. Time-series analysis of trends requires discharge data in the form of mean daily discharges for each day in the period analyzed. This is a much more extensive data set than instantaneous discharges at the time of water-quality sampling, which are acceptable for the flow-adjustment procedures associated with application of the seasonal Kendall test. Time-series analysis requires a long-term record of water-quality data. The model used in this report requires at least 60 water-quality measurements during at least 15 years, although the years may be nonconsecutive. The most restrictive data requirement is that fewer than 10 percent of the measurements may be below the detection limit. When carefully applied and interpreted, time-series analysis can be used to detect complex trends in concentration and evaluate the efficiency of various sampling designs for monitoring trends in water quality.

Time-Series Modeling of Water-Quality Trends

The concept of separating a meaningful pattern, such as seasonal variability, from random variations has been described by Davis (1973, p. 222):

Statisticians have borrowed several terms from the jargon of electrical engineering, and sometimes speak of a sequence of data as being "noisy." This implies that the observations consist of two parts; an underlying *signal* or meaningful pattern of variation, and a superimposed *noise* or random variation. These expressions are most apt to be encountered in time series problems, because research on radio-signal analysis has contributed greatly to this branch of statistics. The idea that a meaningful message is submerged in the welter of often confusing data that a geologist accumulates is an appealing one. . .

The methods used in this study to analyze water-quality trends have been described in detail by Vecchia (2000). A brief summary of the methods is included here. A joint time-series model for daily mean discharge and concentration is fitted to historical data for each site and each constituent. The model is used to filter out as much natural, discharge-related variability in concentration as possible before analyzing for trends. The time-series model separates the data into

components of annual variability, seasonal variability, and noise (deviations from the basic conditions).

The time-series model is applied to log-transformed discharge and concentration data rather than to untransformed data, because these types of environmental data are usually closer to a normal distribution after being log-transformed. Also, because discharge and concentration data often vary over several orders of magnitude, log-transformed data are more numerically stable than untransformed data for estimating the parameters of the time-series model (Vecchia, 2000, p. 36). In applying the log-transformation, zero values are replaced by the smallest nonzero value of the corresponding time series. Some of the trend results are exact only if the time-series model residuals (defined later) are normally distributed. Vecchia (2000, appendix A) shows, however, that trend results are not sensitive to deviations from the assumption of normally distributed residuals, as long as the number of concentration measurements is large and there are no extreme outliers that may significantly influence the fitted model. The time-series model residuals for each site and constituent analyzed in this study were determined to be approximately normally distributed. The presence of occasional outliers did not significantly affect the fitted model in most cases, and did not alter the general nature of the trend results.

The form of the time-series model used to analyze variability in discharge data can be expressed as:

$$X = C + A + S + U, \quad (1)$$

where X = base-10 logarithm of discharge,
 C = a constant (the overall mean of the discharge data),
 A = a time series that represents annual variability in discharge,
 S = a time series that represents seasonal variability in discharge, and
 U = a time series of deviations from the discharge values represented by $C + A + S$.

Likewise, the concentration model is expressed as:

$$Y = C + A + S + T + W, \quad (2)$$

where Y = base-10 logarithm of concentration,
 C = a constant,
 A = a time series that represents annual variability in concentration,
 S = a time series that represents seasonal variability in concentration,
 T = a trend in concentration, and
 W = a time series of deviations from concentration values represented by $C + A + S + T$.

A non-technical description of the time-series model for chloride concentration for the Connecticut River at Thompsonville is presented in this section. The model first filters out as much natural, discharge-related variability in concentration as possible, using mean daily discharge data (figs. 2 and 3). The solid line in figure 2 is a 1-year moving average that represents annual variability in discharge (A in eq. 1) for the Connecticut River at Thompsonville. The solid line in figure 3 represents smooth inter-annual and seasonal variability in discharge ($C+A+S$ in eq. 1). This line, which is called the low-frequency component of discharge ($C+A+S$), was generated by a smoothing algorithm that separates low-frequency (annual and seasonal) variability from high-frequency variability, or noise. Both the low-frequency component and the noise (U in eq. 1) are important for explaining variability in concentrations of the various chemical constituents. The noise also is referred to as the high-frequency component of discharge.

Concentrations of dissolved chloride for the Connecticut River are shown in figure 4. The solid line represents the fitted low-frequency component in concentration ($C+A+S$ in eq. 2). The mathematical procedures for determining C , A , and S for concentration are more complex than for discharge, because daily concentration data are unavailable. The low-frequency component is estimated from a nonlinear regression of concentration on the low-frequency component of discharge (Vecchia, 2000, eq. A6, p. 39–41). Considerable inter-annual variability is present in chloride concentration; there also is strong seasonality in chloride concentrations, with peak concentrations tending to take place in winter and a secondary peak in summer. Concentrations during the spring runoff season tend to be the lowest.

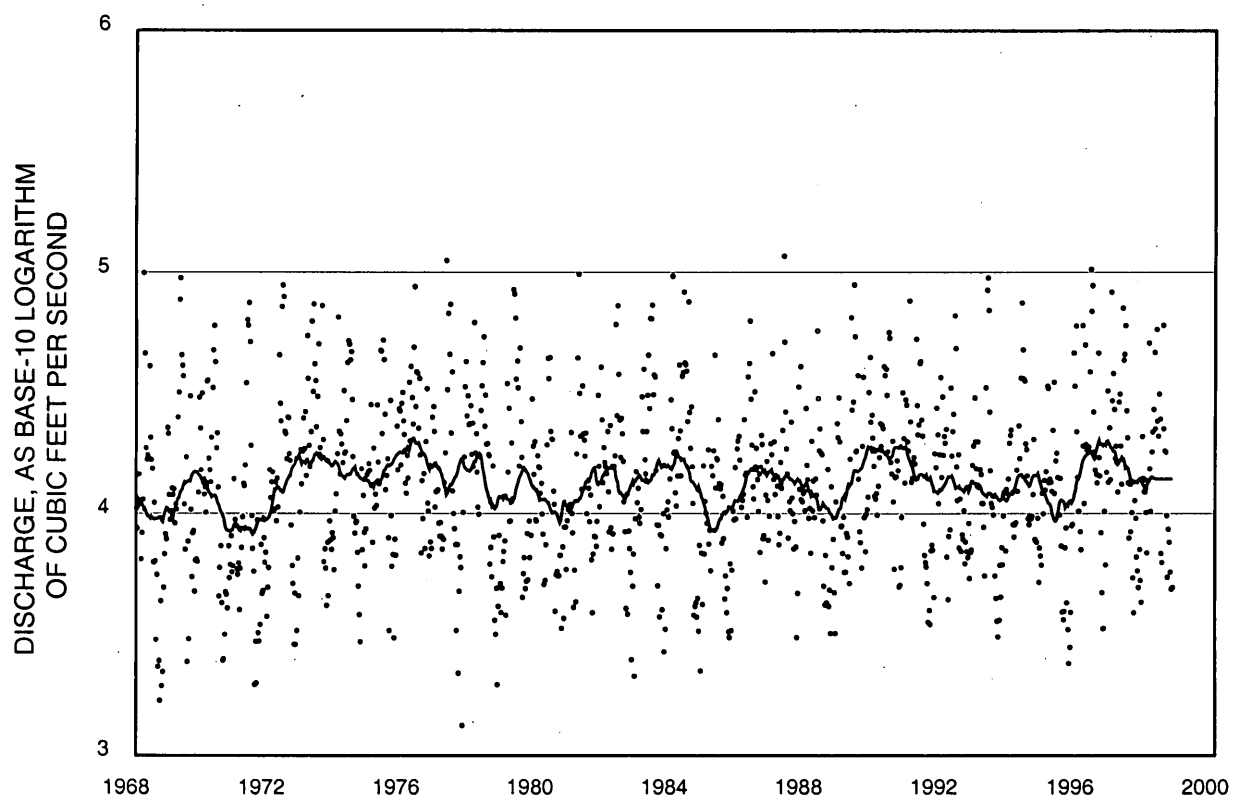


Figure 2. Annual variability in discharge on the Connecticut River at Thompsonville, Conn., 1968–98. Points show recorded daily discharges (36 per year). Line shows 1-year moving average (A in eq. 1).

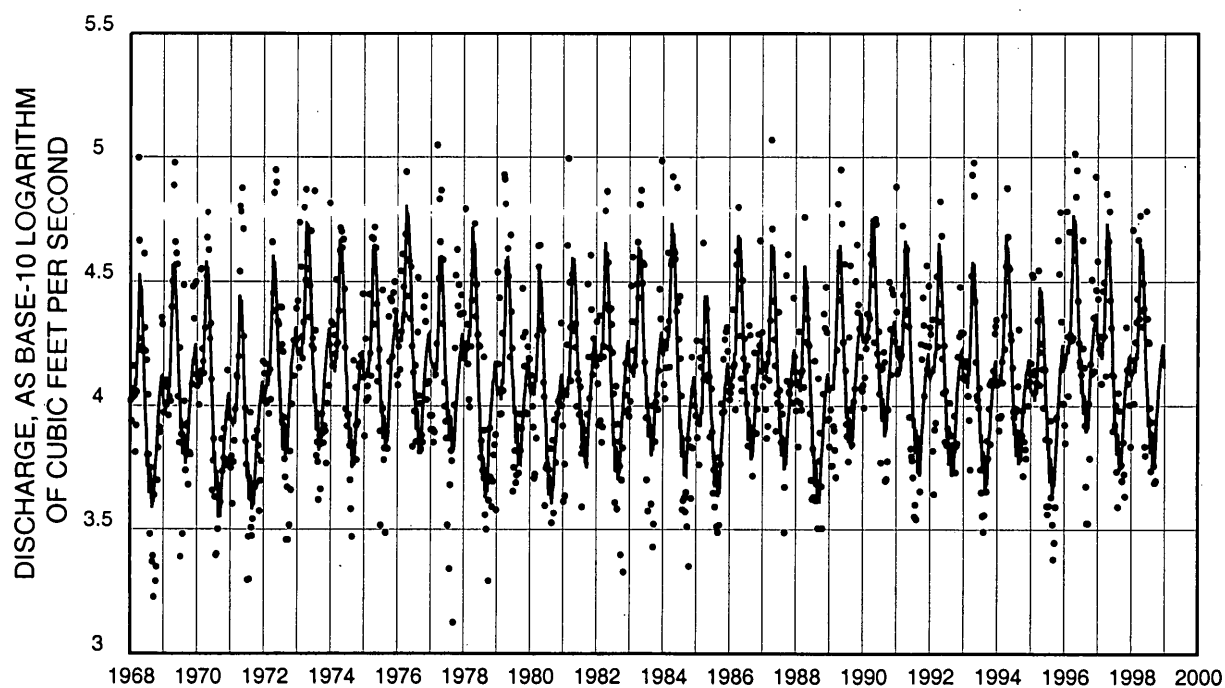


Figure 3. Recorded daily discharges (points, 36 per year) and fitted low-frequency component (line, $C + A + S$ in eq. 1), Connecticut River at Thompsonville, Conn., 1968–98.

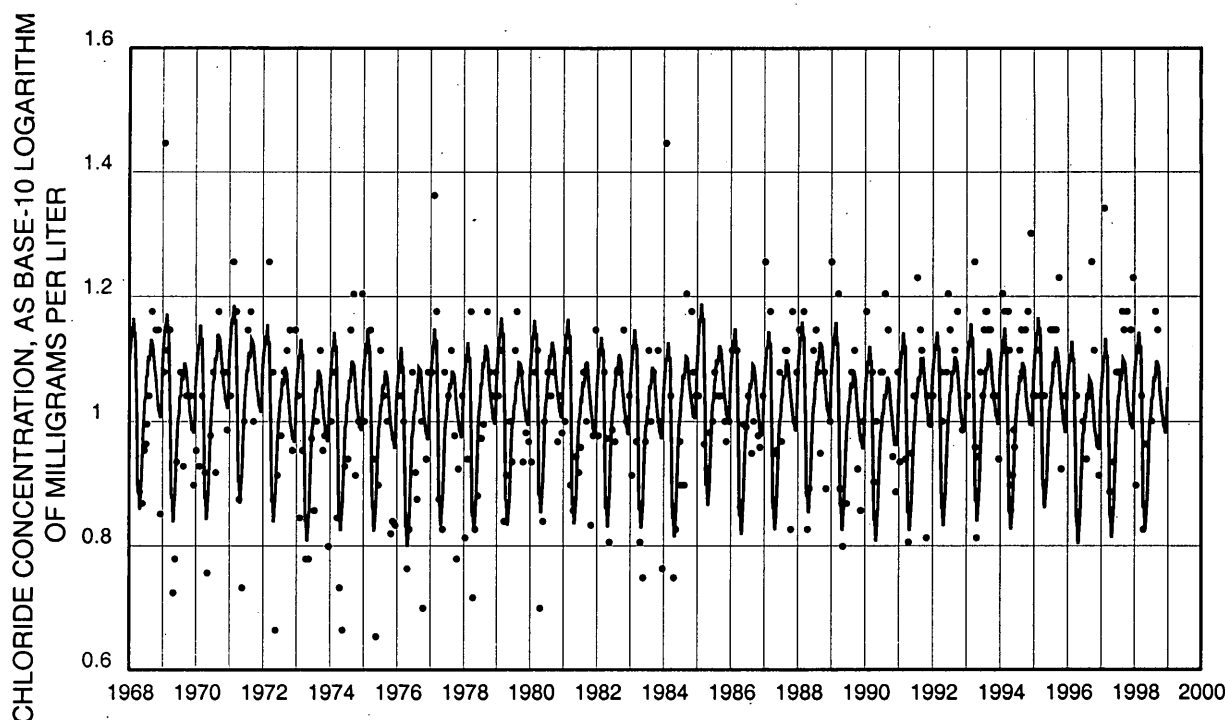


Figure 4. Recorded dissolved chloride concentrations (points) and fitted low-frequency component showing discharge-related variability (line, $C + A + S$ in eq. 2), Connecticut River at Thompsonville, Conn., 1968–98.

Flow-adjusted dissolved chloride concentrations are shown in figure 5. In the flow-adjustment process, most of the flow-related annual and seasonal variability ($A+S$ in eq. 2) is removed from concentration (Y in eq. 2), as described by Vecchia (2000, p. 41). The flow-adjusted concentrations are thus composed of a constant plus any trend present plus noise ($C+T+W$ in eq. 2). Flow-adjusted concentrations can be interpreted loosely as the concentrations that would have been observed if flow conditions had been uniform throughout the entire sampling period.

A significant decrease in flow-adjusted concentrations of chloride took place on the Connecticut River during 1968–73, followed by a steady increase in concentrations during the remaining years (fig. 5). The solid line in figure 5 is the estimated trend in chloride concentration, obtained by methods described later in this report. The statistical significance of the trends depends on the statistical properties of the noise (W in eq. 2). The noise, which is also referred to as the high-frequency component of concentration, may have a complex time-series structure that is not immediately evident from simple inspection of the data. For example, the standard deviation of the noise may vary

depending on the time of year. The concentration noise (W in eq. 2) also may be cross-correlated with the noise in the discharge data (U in eq. 1) and this cross-correlation may vary depending on time of year. There also may be a serial correlation between the concentration noise at a particular time and the noise at neighboring times, and this serial correlation may depend on time of year as well.

A special type of time-series model, called a periodic autoregressive moving average (PARMA) model (Vecchia, 2000, appendix A) is used to detect and filter out the complex statistical properties of the noise in concentration data. Some of the statistical properties of the noise for the data in figure 5, estimated using the PARMA model, are shown in figure 6.

The standard deviation of the noise for dissolved chloride concentrations on the Connecticut River (fig. 6a) is highest during December and January and lowest during June and July, indicating that variability of chloride concentrations from typical seasonal values is highest during winter and lowest during summer. It should be noted that seasonal standard deviations are expressed in logarithmic units, and the variability of

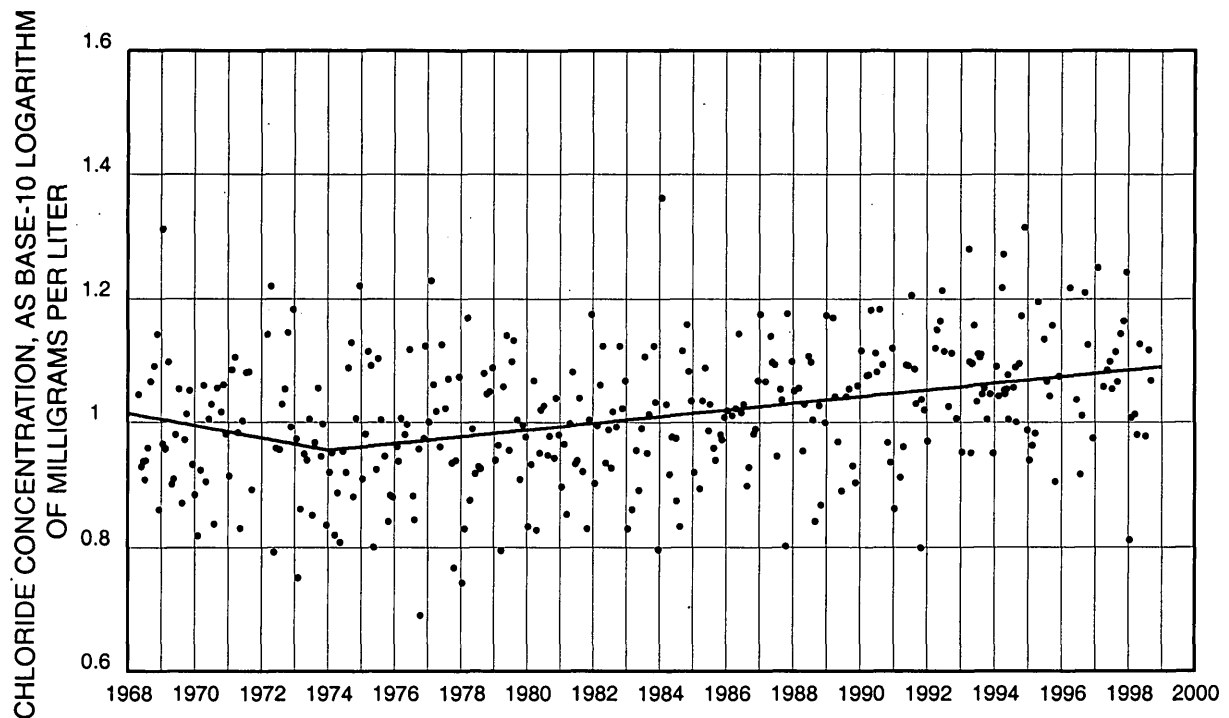


Figure 5. Flow-adjusted dissolved chloride concentrations (points) and fitted trends (lines), Connecticut River at Thompsonville, Conn., 1968–98.

chloride concentrations in arithmetic units does not necessarily follow the same pattern. The cross-correlation between the noise in chloride concentration and the noise in discharge (fig. 6b) is negative throughout the year, indicating that higher than normal discharge tends to coincide with lower than normal chloride concentrations. The magnitude of the negative correlation is not uniform throughout the year. The negative correlation is strongest in April to June and again in October to November. The serial correlation (autocorrelation) in the concentration noise (fig. 6c) indicates that the noise is nearly uncorrelated at a 10-day lag during December to January but significant autocorrelation exists during other months. For example, the autocorrelation between the noise on April 1 and the noise 10 days before (March 22) is about 0.7. Thus, from late

February to early November, above normal (or below normal) chloride concentrations tend to persist for at least 10 days before returning to normal.

Statistical properties such as those shown in figure 6 can bias estimated trends and significance levels if not properly accounted for in the trend analysis. The PARMA model provides a convenient way to filter out structure in the noise and correct the bias in estimated trends and significance levels. The PARMA model also uses information on the statistical properties of the noise to identify months during which concentration sampling yields the most information for trend analysis, enabling sampling schedules to be identified that maximize the sensitivity for detecting trends for a given sampling frequency.

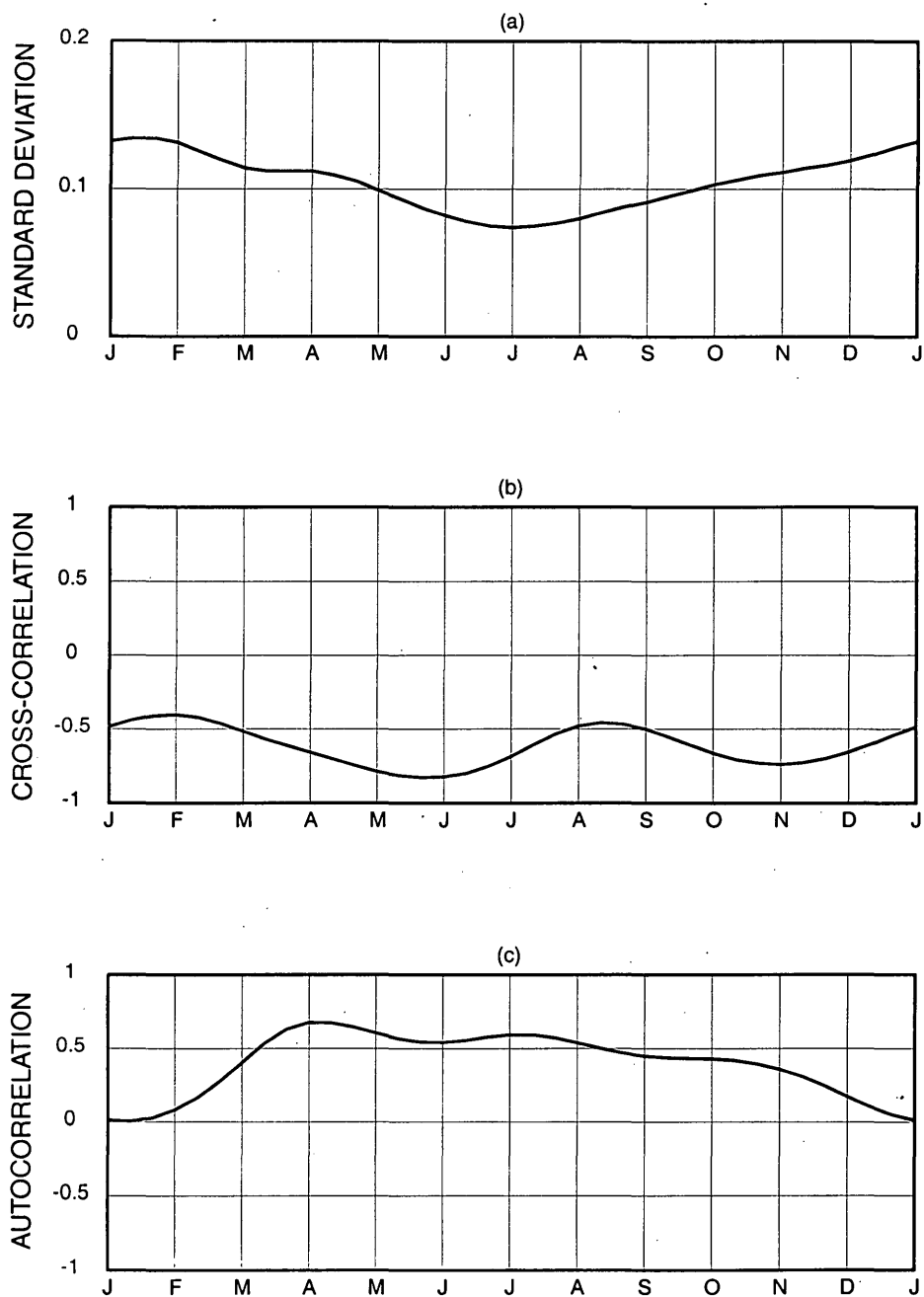


Figure 6. Seasonal characteristics of the noise in dissolved chloride concentrations for the Connecticut River at Thompsonville, Conn., 1968–98: (a) computed seasonal standard deviation for the high-frequency component of dissolved chloride concentration, (b) cross-correlation at a 0-day lag with the high-frequency component of discharge, and (c) autocorrelation at a 10-day lag.

Application of the Time-Series Model to Connecticut Data

The time-series model was first run for each constituent at each station with no trend periods specified. Trends in flow-adjusted concentration are often apparent in the plot of flow-adjusted concentrations (fig. 5) and in the distribution of the PARMA model residuals for the no-trend model (fig. 7). Residuals from the PARMA model are essentially the noise within the noise for concentration data. After the PARMA model is applied to the noise in concentration data (W in eq. 2), and the statistical properties shown in figure 6 have been filtered out, the residuals from the PARMA model (points in fig. 7) are the unexplained remnant of concentration variability, including any trends that may be present.

A smooth line was added to the plot of PARMA model residuals to aid in identifying central patterns in the data (fig. 7). The form of the smoothing procedure

used is LOWESS, or locally weighted scatterplot smoothing (Cleveland, 1979). The smooth line indicates the major changes in slope and trend directions over the period of record. For example, the smooth line through the residuals for chloride for the Connecticut River at Thompsonville shows a distinct upward trend for the 20-year period from 1979–98. The plot of PARMA model residuals from the no-trend model was then used to select appropriate trend periods for the time-series model. Trend periods as short as 3 years and as long as the period of record were used in defining possible models for each station and constituent. Information on dates of important laboratory method changes or environmental changes such as wastewater-treatment plant upgrades also was used to select dates for linear trend periods or step-trend periods and to evaluate trend results. Trend results have been reported as significant in this report if the p -value for the test statistic was less than or equal to 0.05.

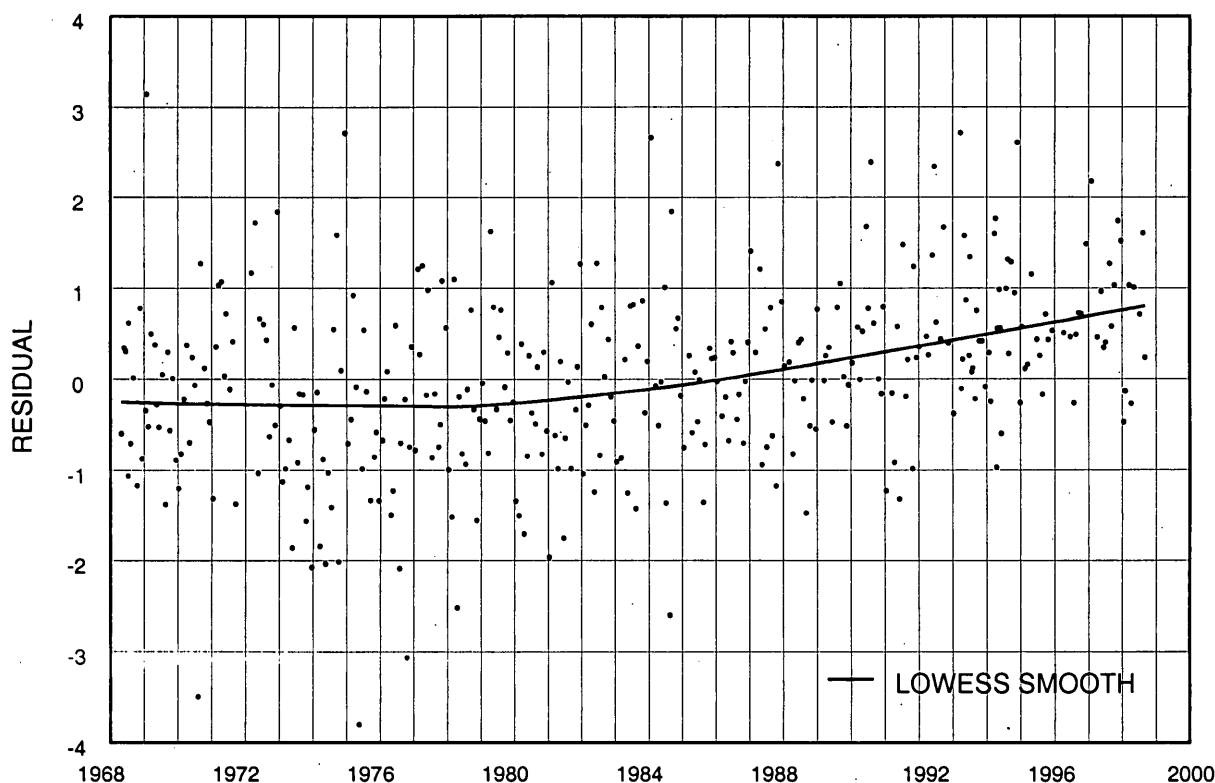


Figure 7. Residuals from periodic autoregressive moving-average (PARMA) model for the high-frequency component of dissolved chloride concentrations, with no trend periods specified, Connecticut River at Thompsonville, Conn., 1968–98.

Where the water-quality data set has a large number of values in each year, as was the case for most stations and constituents analyzed, the time-series model can be broken down into linear trend segments as short as 3 years. This feature makes possible a detailed examination of water-quality variability during the period of record. For example, an overall downward trend over a long period of time may be shown, on closer examination, to consist of several short-term decreases in concentration separated by "plateaus" of no-trend years. Several trend models were tested for each constituent at each station using linear trend periods, and in some cases, step-trend

periods of varying lengths. A numerical evaluation of the model fit and p-values for the various trend periods were examined and compared to select the model that best represented the changes in the constituent during the period of record. Residual plots for the selected model were examined to ensure that residuals met assumptions of random distribution with constant variance and no apparent trends. For example, when appropriate trend periods were added to the model for chloride on the Connecticut River, the trend in residuals for the no-trend model (fig. 7) was eliminated (fig. 8).

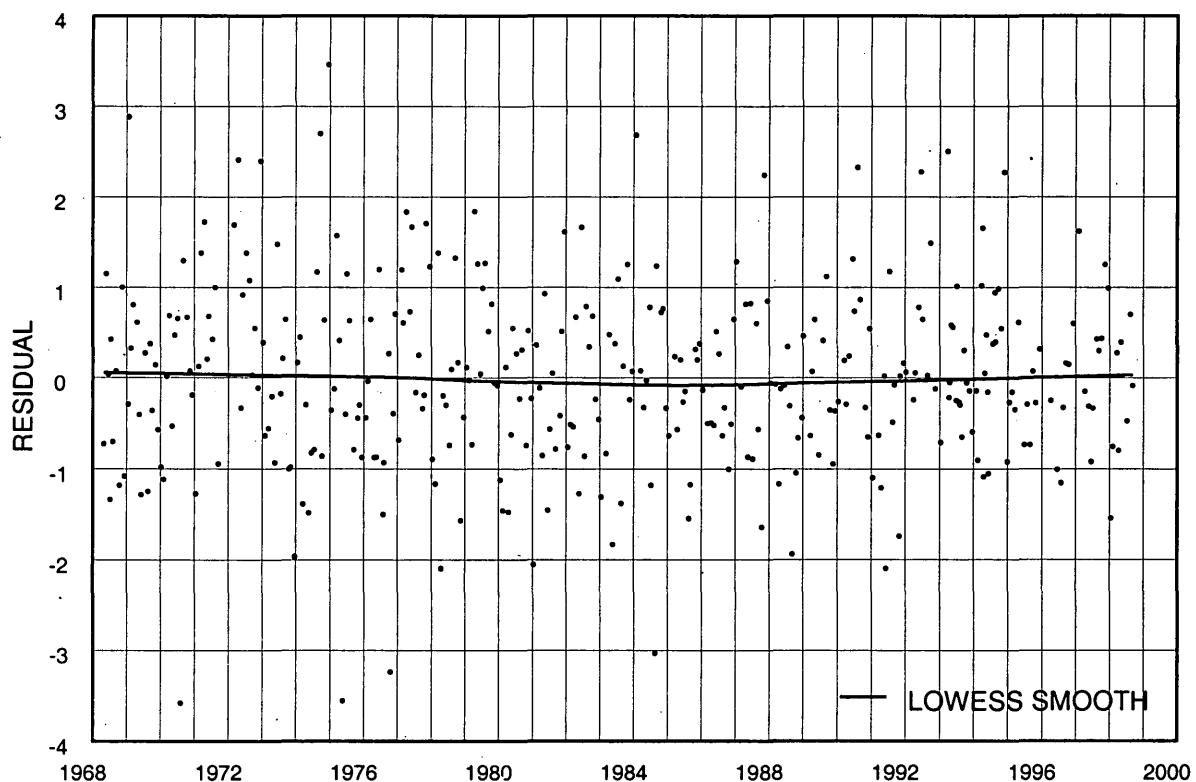


Figure 8. Residuals from periodic autoregressive moving-average (PARMA) model for the high-frequency component of dissolved chloride concentrations, with trend periods specified, Connecticut River at Thompsonville, Conn., 1968–98.

Evaluation of Sampling Schedules with Time-Series Analysis

The time-series model can be used not only to analyze historical data for trends but also to determine sampling designs that are efficient for monitoring future trends (Vecchia, 2000, p. 26). Sampling designs can be evaluated in terms of maintaining a sampling frequency that is sufficient for future trend analysis, reducing sampling costs by eliminating samples that provide redundant information, or shifting the most frequent sampling to seasons that provide the greatest gain in information.

National Evaluation of Sampling Designs

Findings from the NAWQA national trend study showed that trends were generally not monotonic—increases, decreases, and plateaus took place throughout the period of record for many streams. If trends are nonmonotonic or periodic, it becomes important to optimize sampling frequency over time in order to detect these variations. Sampling schedules that alternate between years of high-frequency sampling and periods with one or more full years of missing water-quality record may result in a situation in which nonmonotonic or periodic trends are not detected or are misinterpreted. Maintaining even a low-sampling frequency at a station (two to four samples per year) is preferable to the periodic suspension of sampling and will improve the possibility for trend detection.

One of the clear conclusions of the NAWQA study of national trends and sampling frequencies is that it is not advisable to discontinue sampling at a station for a period of time if data sets will be evaluated for long-term trends. If sampling must be reduced, it is preferable to cut back on sampling frequency at several stations and maintain continuity of sampling, rather than to maintain a higher but discontinuous sampling frequency that is rotated among stations periodically.

Theoretical Basis for the Sampling Design Program and Application to Connecticut Data

Given an historical record of discharge and concentration data with which to calibrate the statistical time-series model, the model can be used to evaluate the efficiency of various sampling designs (that is, sampling schedules) for monitoring trends in water quality. The theory and mathematical basis for applying the time-series model to analysis of sampling designs have been described in detail by Vecchia (2000, appendix A).

An optimal sampling design for a given water-quality constituent is defined as the design that maximizes the capability for detecting a trend in the given constituent for a fixed sampling cost, which is usually measured in terms of the number of samples per year. As the number of samples is increased, the sensitivity for detecting a given trend increases; however, the placement of samples during the year is an important consideration. A design with only 6 samples per year may be more efficient than a design with 8 samples per year if the sampling times of the 8-sample design are poorly selected.

The optimal sampling design needs to be defined on the basis of the objectives of the trend study. For example, if only those trends that take place in summer are of interest, then an optimal design would place more emphasis on sampling during summer than during other times of the year. The designs developed in this report assume that trends during all times of year are equally important, and samples are allocated throughout the year to maximize capability to detect trends whenever they take place. The optimal design also may depend on the type of trend being monitored. For example, to detect a consistent monotonic trend over the next 10 years, an optimal design would place more samples near the beginning and end of the 10-year period, because the differences in concentrations are largest when contrasting the early and late periods. The designs developed here assume that trends may be non-monotonic and that the times in which the direction of the trends may change are not known in advance. Therefore, it is assumed that the same sampling frequency and sampling times are repeated year after year. A constant sampling schedule balances the capability to detect trends that take place at random times and persist for random durations.

Given the assumptions that trends are equally likely to take place during any time of year and that the beginning year of the trend and duration of the trend are not known in advance, efficient designs can be developed for a given constituent. In this study, monthly sampling is the maximum allowable sampling frequency based on cost considerations, with one sample per month collected at approximately the same time of month. The sampling design is not particularly sensitive to whether samples are taken near the beginning, middle, or end of each month. The assumption in this study is that samples are collected near the 15th of each month. Various designs with sampling frequencies ranging from 4 to 11 samples per year are evaluated by omitting selected months from the design. Lower-cost (lower-frequency) designs are compared to

the efficiency and power (defined below) of the monthly design to determine the best configuration of months to sample for given cost, and to evaluate whether the efficiency of the design improves as costs increase.

For trend detection purposes, many of the thousands of mathematically possible sampling designs are virtually equivalent. In this study, the entire set of potential designs has been reduced to a set of 150 designs that represent the range of reasonable possibilities for designs with 4 to 12 samples per year.

Sampling designs are evaluated in two ways: (1) in terms of their power to detect trends, and (2) in terms of the efficiency of the sampling design. Power refers to the probability of detecting a trend of given size, and design efficiency is measured in terms of the size of the detectable trend. Together, these two measures are used to determine optimal sampling designs.

The efficiency of a design is measured in terms of its characteristic trend. For purposes of this study, the characteristic trend is defined as the trend, in percent, that can be detected over the entire design period (5 years) with probability 0.8, using a significance level of 0.05. The smaller the characteristic trend is for a given sample size, the more efficient the design.

The magnitude of the trend that can be detected depends on the length of the design period (the period over which a trend is to be evaluated). For example, a trend that persists for only 2 years would need to be much larger than a trend that persists for 10 years to have the same probability of being detected. However, the relative efficiency of various designs in relation to monthly sampling does not depend on the length of the design period. So, for example, if a particular design with 8 samples per year is efficient for detecting a trend after 10 years, the same design will be efficient for detecting a trend after 2 years. That is, the months identified in the 8-sample design yield the most information for trend analysis for that constituent at that station. Although the trend may need to be large to be detected after 2 years, increasing sampling frequency to more than 8 samples per year for the 2-year period will not necessarily decrease the size of trend that can be detected.

The power of the various designs can all be plotted on the same scale, making possible a dimensionless comparison among constituents. The overall power of a design for multiple constituents is defined as the average power of the design over all the constituents.

The optimal design for a given sample size is defined as the design with the highest average power over all constituents in the design. The average power (indicated by 'C' for composite) and the power for each of the individual constituents are shown in design plots in this report.

The design program uses results from the time-series analysis of trends to generate information on sampling designs. Design results have been generated for each station for design periods ranging in length from 5 to 10 years. For each design period, the program identifies optimal designs for sampling frequencies ranging from 4 to 11 samples, or observations, per year. The discussion of results in this report is based primarily on results from the 5-year design period. Decreasing (or increasing) the number of years in the design period will result in an increase (or decrease) in the magnitudes of trends that can be detected compared to the magnitudes reported here.

LONG-TERM TRENDS IN SURFACE-WATER QUALITY

Long-term and short-term trends in six constituents were identified for variable periods of record during the study period of 1968–98. Trends with an attained significance level of 0.05 or less have been presented as significant trends in this report. Test statistics for trend results and trend magnitudes in percent change per year are presented in the appendix.

Chloride

Concentrations of dissolved chloride increased significantly in all five streams during the period of record (fig. 9a–e). The smallest increases were on the Salmon River (fig. 9b). Concentrations increased on the Connecticut River from 1974 to 1998. On the Salmon and Saugatuck Rivers, there was considerable variability in chloride concentration during the 1960's and 1970's, followed by increasing concentrations during the 1980's, and stabilization at higher levels during the 1990's (figs. 9b and 9c). Chloride concentrations increased steeply on the Quinnipiac River from the mid-1980's to the mid-1990's, and then declined slightly (fig. 9d). On the Naugatuck River, concentrations increased during the 1980's and then declined gradually during the 1990's (fig. 9e). Upward chloride trends in the Salmon and Saugatuck River Basins, the two forested basins with no point sources, indicate that nonpoint sources are probably a factor affecting chloride trends.

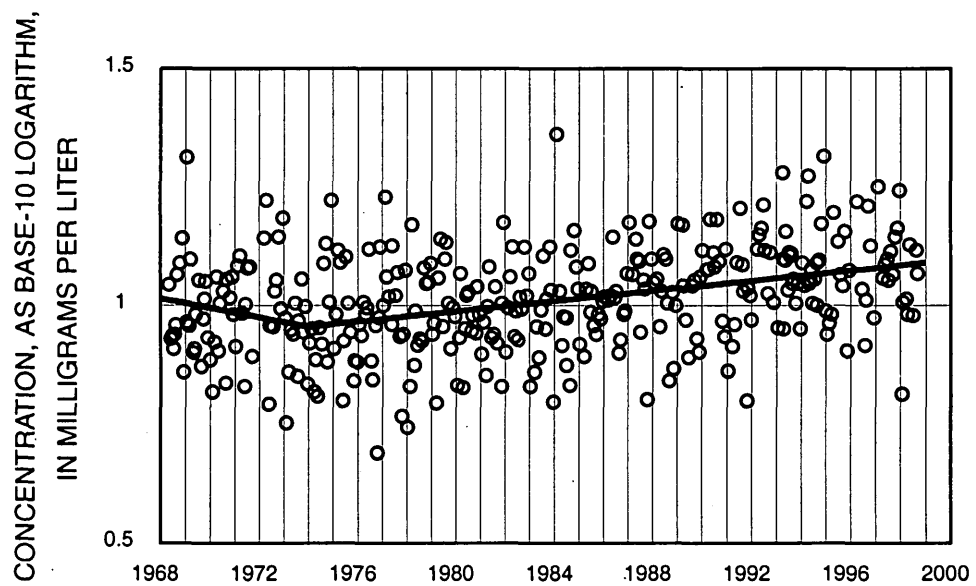


Figure 9a. Trends (lines) in flow-adjusted concentrations of dissolved chloride (points), Connecticut River at Thompsonville, Conn., 1968–98.

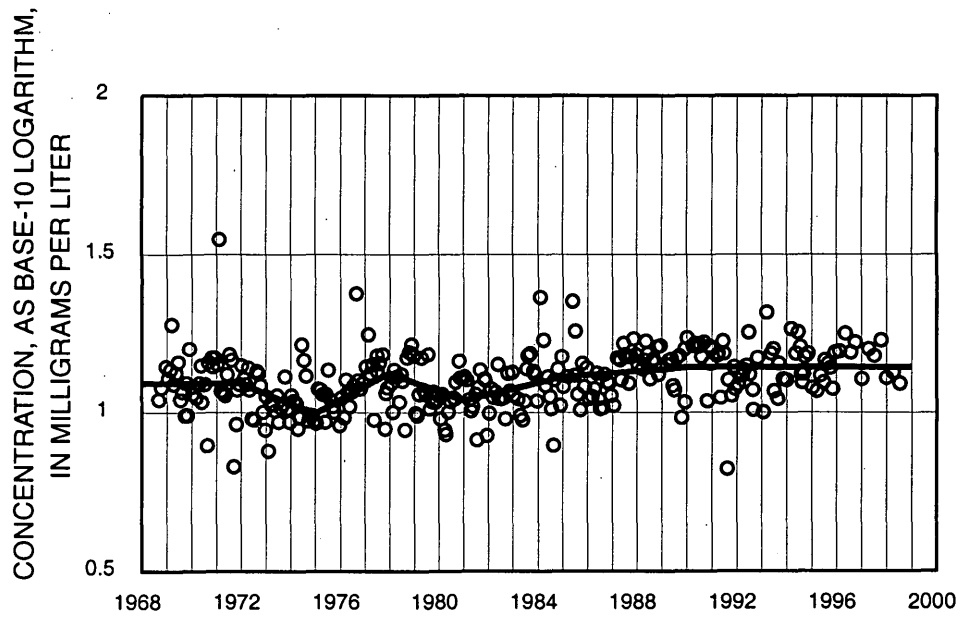


Figure 9b. Trends (lines) in flow-adjusted concentrations of dissolved chloride (points), Salmon River near East Hampton, Conn., 1968–98.

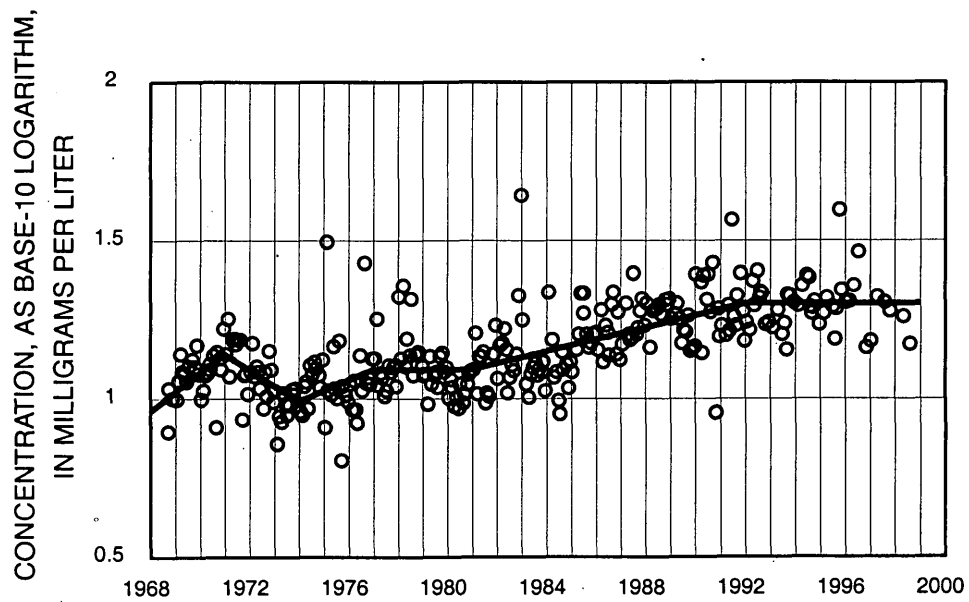


Figure 9c. Trends (lines) in flow-adjusted concentrations of dissolved chloride (points), Saugatuck River near Redding, Conn., 1968–98.

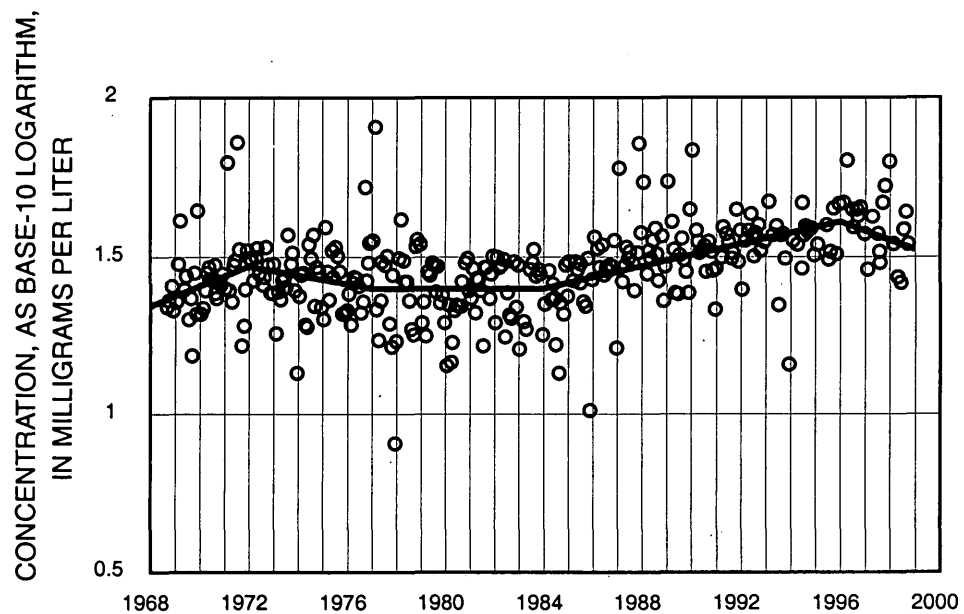


Figure 9d. Trends (lines) in flow-adjusted concentrations of dissolved chloride (points), Quinnipiac River at Wallingford, Conn., 1968–98.

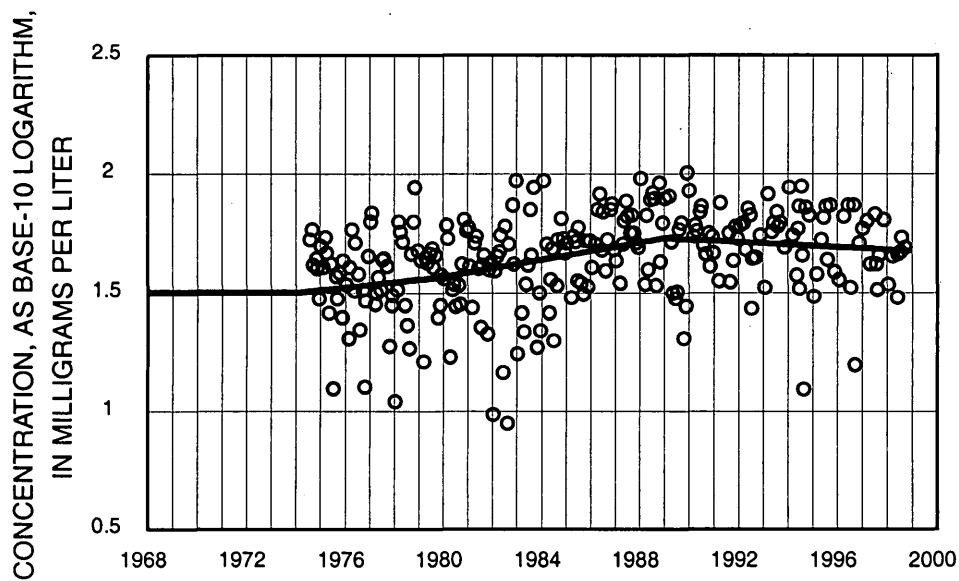


Figure 9e. Trends (lines) in flow-adjusted concentrations of dissolved chloride (points), Naugatuck River at Beacon Falls, Conn., 1974–98.

Sulfate

Significant increases and decreases in dissolved sulfate concentrations were detected in the data for all five streams during one or more time periods within the period of record (fig. 10). Typical flow-adjusted concentrations of dissolved sulfate were lower at the end of the period of record than at the beginning in all five streams. Data for all stations were modeled to evaluate possible effects of a period of positive method bias (1983–89) described previously in this report. A positive step trend for all or part of the period of method bias was detected in the data for all streams except the Naugatuck River (figs. 10a–e); step trends are shown as part of the trend lines. Concentrations of sulfate in the Naugatuck River are higher than in the other four streams, and the magnitude of the method bias may not have been significant relative to typical stream concentrations at this station.

A highly significant downward trend in dissolved sulfate concentrations was detected for the Quinnipiac River for the period of record, 1968–98. A more detailed look at the trend period shows that the significant decreases in concentration took place during three time intervals, 1968–75, 1981–84, and 1992–98, with periods of no trend during 1976–80 and 1985–91 (fig. 10d). The sharpest downward linear trend in concentration took place during the early 1980's (the downward trend line broken by the step trend in 1983).

Declines in sulfate at all five stations, in some cases spanning three decades, suggest a regional cause. Nationally, air emissions of sulfur dioxide have declined substantially between the early 1970's and the late 1990's (U.S. Environmental Protection Agency, 2000, table 3-13, p. 3-19 to 3-20; fig. 3-5, p. 3-25). The relation of trends in stream quality in Connecticut to trends in regional air quality is a potential area for further investigation.

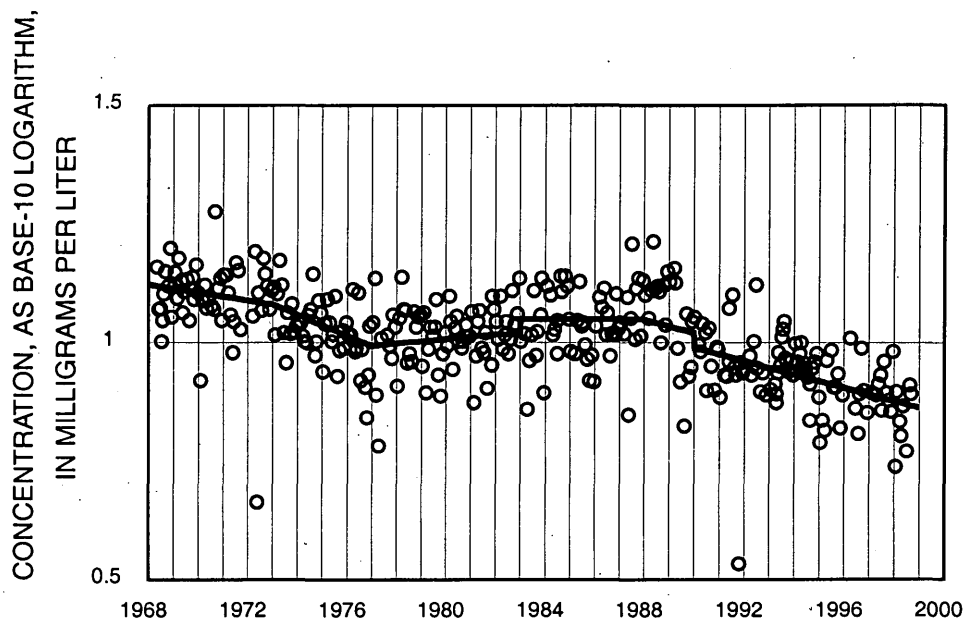


Figure 10a. Trends (lines) in flow-adjusted concentrations of dissolved sulfate (points), Connecticut River at Thompsonville, Conn., 1968–98.

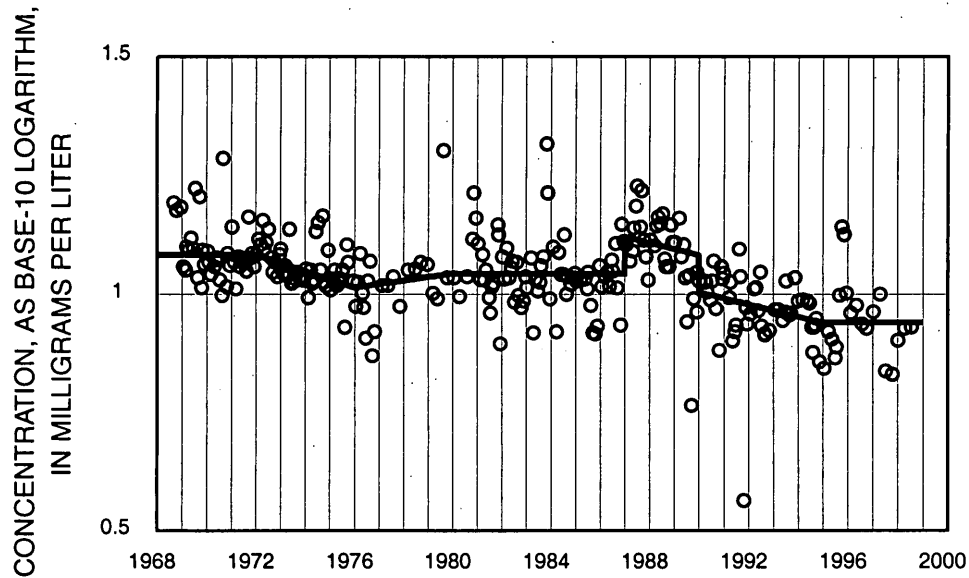


Figure 10b. Trends (lines) in flow-adjusted concentrations of dissolved sulfate (points), Salmon River near East Hampton, Conn., 1968–98.

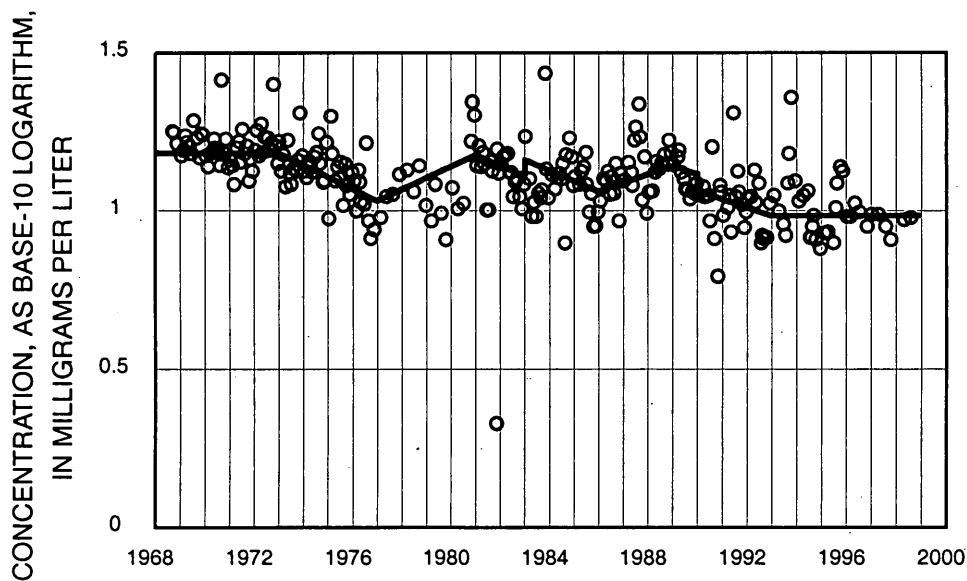


Figure 10c. Trends (lines) in flow-adjusted concentrations of dissolved sulfate (points), Saugatuck River near Redding, Conn., 1968–98.

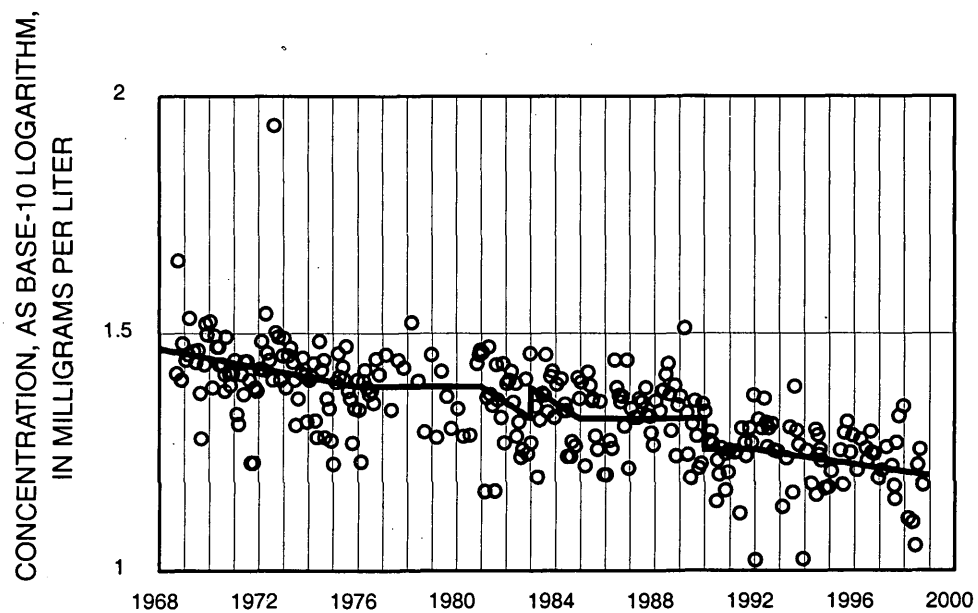


Figure 10d. Trends (lines) in flow-adjusted concentrations of dissolved sulfate (points), Quinnipiac River at Wallingford, Conn., 1968–98.

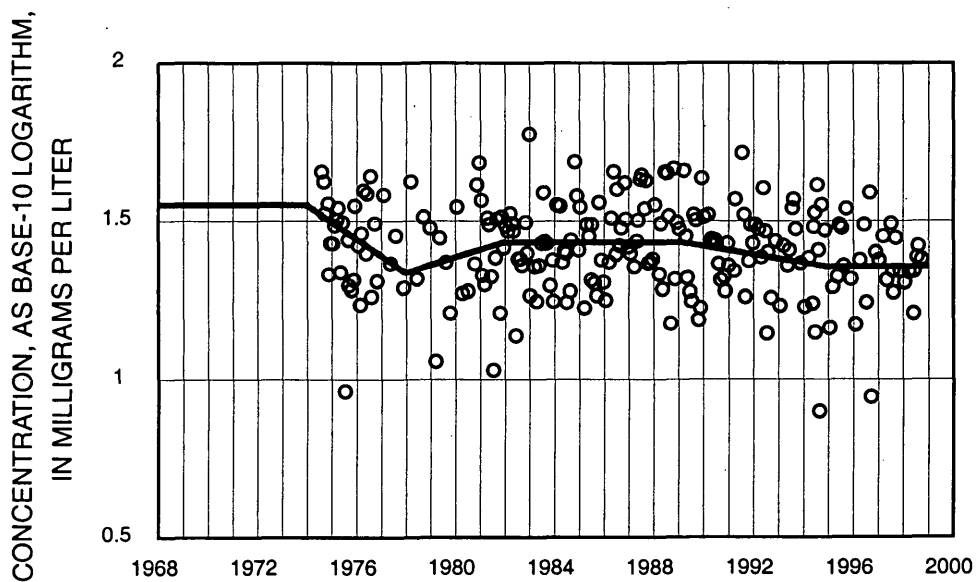


Figure 10e. Trends (lines) in flow-adjusted concentrations of dissolved sulfate (points), Naugatuck River at Beacon Falls, Conn., 1974–98.

Total Nitrogen

Flow-adjusted concentrations of total nitrogen in the Connecticut, Salmon, Saugatuck, and Naugatuck Rivers increased during variable periods from the mid-1970's to the mid- or late 1980's, and then decreased to a time in the early 1990's (figs. 11a–c, e). On the Naugatuck, Salmon, and Saugatuck Rivers, concentrations remained relatively stationary during the mid- to late 1990's, although data for the Salmon River show a slight downward trend at the 0.06 significance level. Total nitrogen in the Connecticut River declined slightly during the 1990's (fig. 11a). Several significant short-term increases and decreases in total nitrogen concentration were detected for the Quinnipiac River (fig. 11d); however, no overall trend was detected for the period of record, 1972–98.

As described in the methods section of this report, a positive bias in Kjeldahl nitrogen concentrations (and consequently total nitrogen concentrations), was caused by a laboratory analytical problem from 1986 to 1991. This bias caused a positive step trend in data for all streams except the Naugatuck River (figs.

11a–e). For the Connecticut, Salmon, and Saugatuck Rivers, there was a statistically significant positive step trend for the period of method bias (1986–91); a significant downward linear trend also was detected at these stations during a similar time period (figs 11a–c). The step trend for the Quinnipiac River data intersects two short-term linear trends, one downward and one upward (fig. 11d). Linear trend results for these four streams indicate that significant increases and decreases in total nitrogen concentration took place independent of the method bias.

General trend patterns on four of the five streams are similar, despite differences in land use and the presence or absence of point discharges to the streams. Nationally, air emissions of nitrogen oxide rose from the 1940's to the mid-1970's, and have remained relatively stationary through the 1990's (U.S. Environmental Protection Agency, 2000, table 3-13, p. 3-19 to 3-20; fig. 3-3, p. 3-23). Investigation of changes in atmospheric emission of nitrogen oxides and atmospheric deposition of nitrogen could yield information on possible causes of trends in total nitrogen in streams.

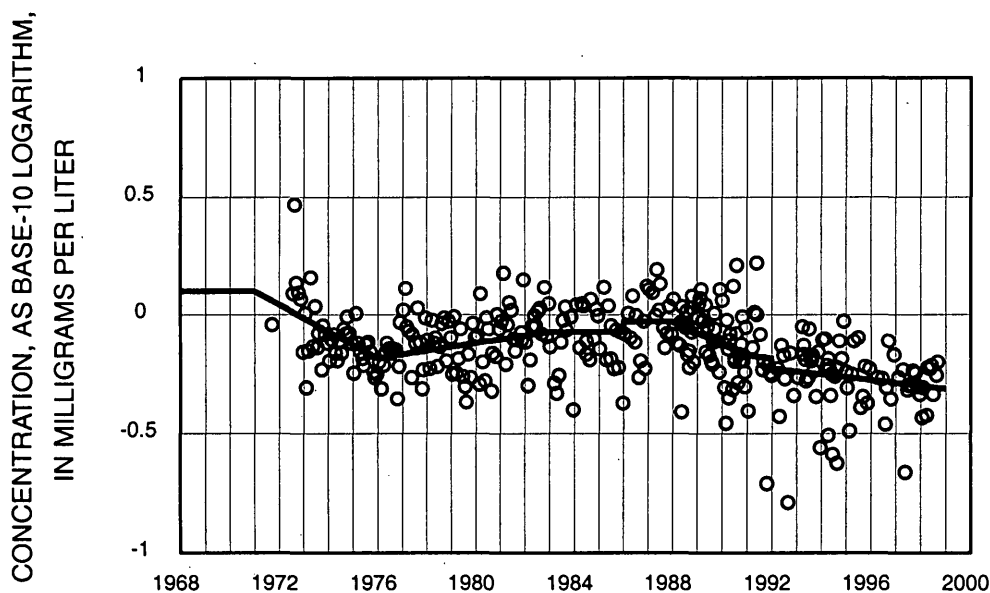


Figure 11a. Trends (lines) in flow-adjusted concentrations of total nitrogen (points), Connecticut River at Thompsonville, Conn., 1971–98.

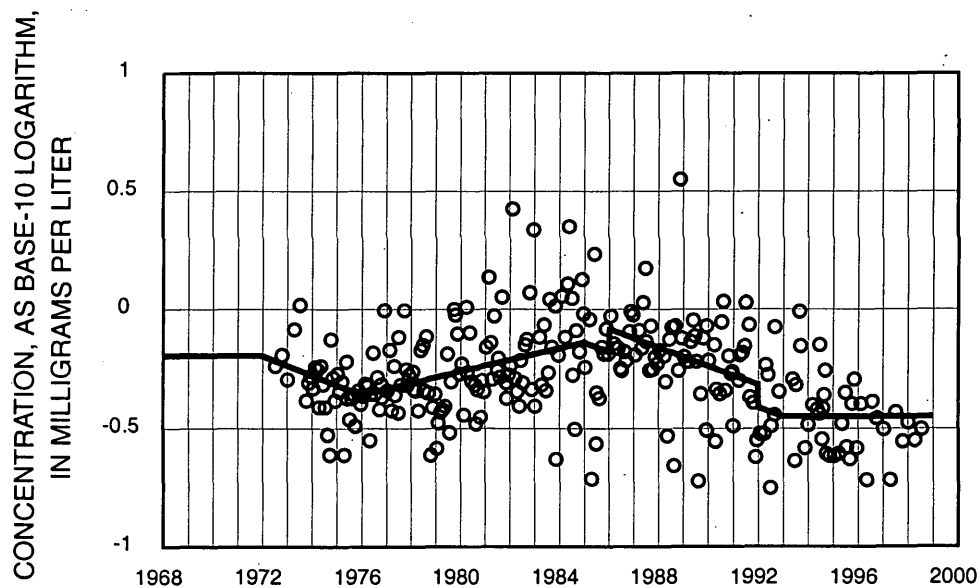


Figure 11b. Trends (lines) in flow-adjusted concentrations of total nitrogen (points), Salmon River near East Hampton, Conn., 1972–98.

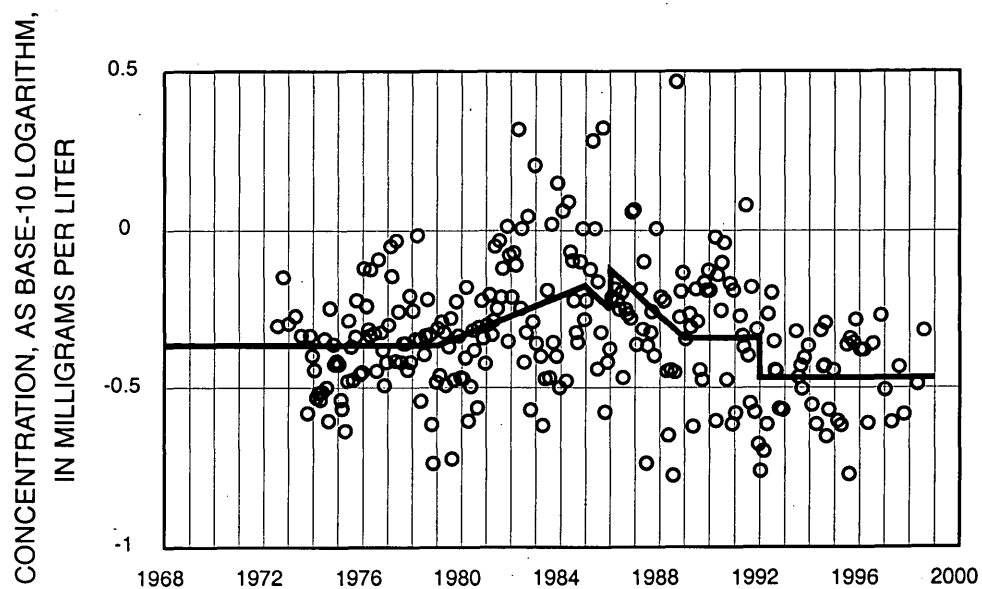


Figure 11c. Trends (lines) in flow-adjusted concentrations of total nitrogen (points), Saugatuck River near Redding, Conn., 1972–98.

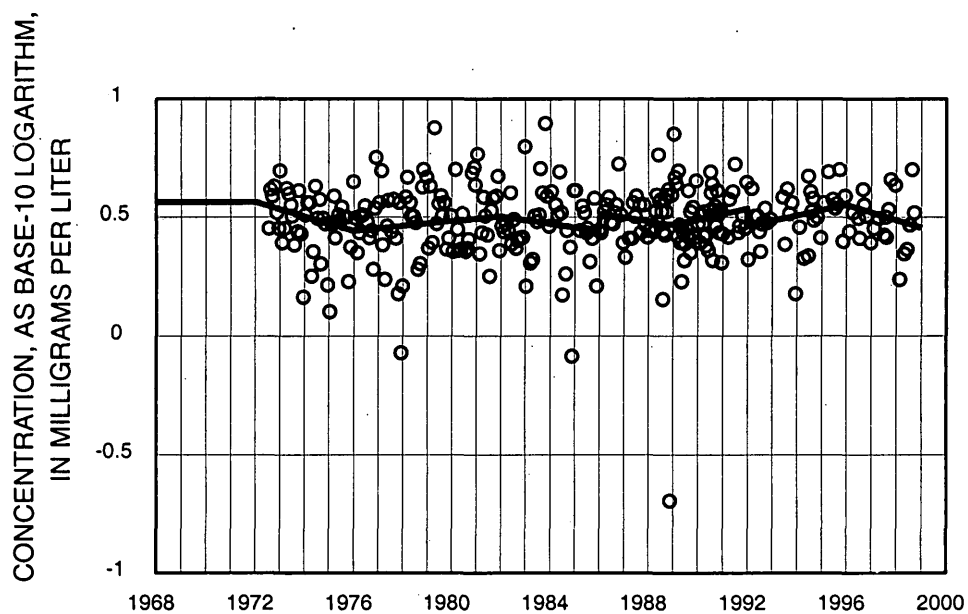


Figure 11d. Trends (lines) in flow-adjusted concentrations of total nitrogen (points), Quinnipiac River at Wallingford, Conn., 1972–98.

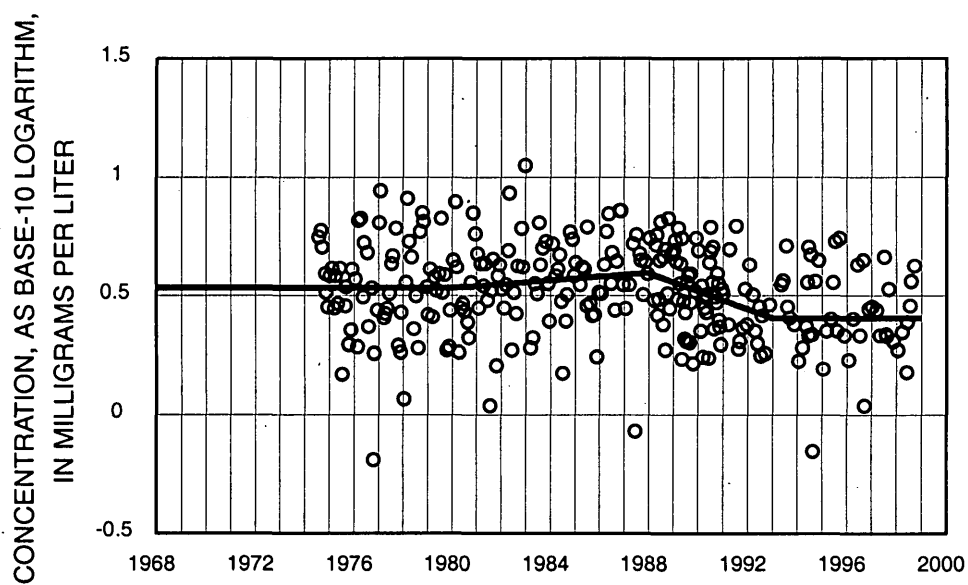


Figure 11e. Trends (lines) in flow-adjusted concentrations of total nitrogen (points), Naugatuck River at Beacon Falls, Conn., 1974–98.

Total Phosphorus

Trend models for total phosphorus revealed distinctly different patterns over time among the five streams (fig. 12). In comparing the trend plots, the actual concentration ranges in the different streams should be kept in mind. Almost all phosphorus concentrations for the Salmon and Saugatuck Rivers are less than 0.1 mg/L (log of 0.1 is -1 in figs. 12b–c), with many values near the detection limit of 0.01 mg/L (log of 0.01 is -2 in figs. 12b–c). By contrast, almost all concentrations for the Quinnipiac and Naugatuck Rivers are greater than 0.1 mg/L (log of 0.1 is -1 in figs. 12d–e).

The most pronounced change in total phosphorus concentrations took place on the Connecticut River (fig. 12a) with a highly significant downward trend for the entire period of record. Additional detailed modeling (not shown in fig. 12a) showed there are actually three periods of downward trends alternating with plateaus of no trend, including a period of no trend for 1993–98. The concentration variance has increased during the 1990's (fig. 12a).

Total phosphorus data for the Salmon and Saugatuck Rivers show a distinct cyclical pattern of increases and decreases during similar time periods on an approximately decadal scale (figs. 12b and 12c). The drainage basins of these two streams are primarily forested, and further study could investigate whether these trend cycles represent some subtle signal, not detectable in systems with larger human impacts, such as natural variability in phosphorus concentrations in forested ecosystems, or variability in atmospheric deposition of phosphorus. Agricultural practices (table 2) also may affect total phosphorus concentrations. These trend results for total phosphorus should be evaluated with caution, because censored data values exceed 10 percent of both data sets, the threshold that has been recommended for application of time-series analysis. The stations are included in the phosphorus analysis as a form of exploratory data analysis.

Concentrations of total phosphorus declined with some variability on the Quinnipiac and Naugatuck Rivers from the mid-1970's to the mid-1990's (figs. 12d and 12e). Decreases were neither as large nor as sustained as on the Connecticut River. In both streams, a significant upward trend was detected from the mid-1990's to 1998. This result was unexpected, given the improvements in wastewater treatment that have taken place in both basins in the early to mid-1990's. These upward trends are based on a small number of years, and the steepness of the trends shown in the plots (figs. 12d and e) may not be sustained when additional years of record are available, as is the case for any short-term trend detected at the end of the period of record.

The probable negative bias in total phosphorus concentrations prior to 1992, documented by the USGS Office of Water Quality (OWQ), led the OWQ to recommend against analyzing trends in data produced by the NWQL for time periods that cross the 1991–92 time boundary (D.A. Rickert, U.S. Geological Survey, written commun., 1992). The concern is that artificial upward trends would be detected because of the negative bias prior to 1992. The possible effects of this bias were taken into consideration in selecting time periods for trend analysis and evaluating detected trends in total phosphorus. However, strong downward trends in total phosphorus were detected in several Connecticut streams during the 1970's, 1980's, and early 1990's, and in some cases, these downward trends continue through or past 1992 (Trench 1996, 2000; Zimmerman, 1997). In the case of the Connecticut, Quinnipiac, and Naugatuck Rivers (figs. 12a, d–e), where water quality is affected by point sources, downward trends extend through or beyond 1992, and the reported phosphorus bias has not affected trend results by causing false upward trends. The actual environmental downward trends may be steeper in some cases than the detected downward trends, if the negative method bias has affected the data prior to 1992.

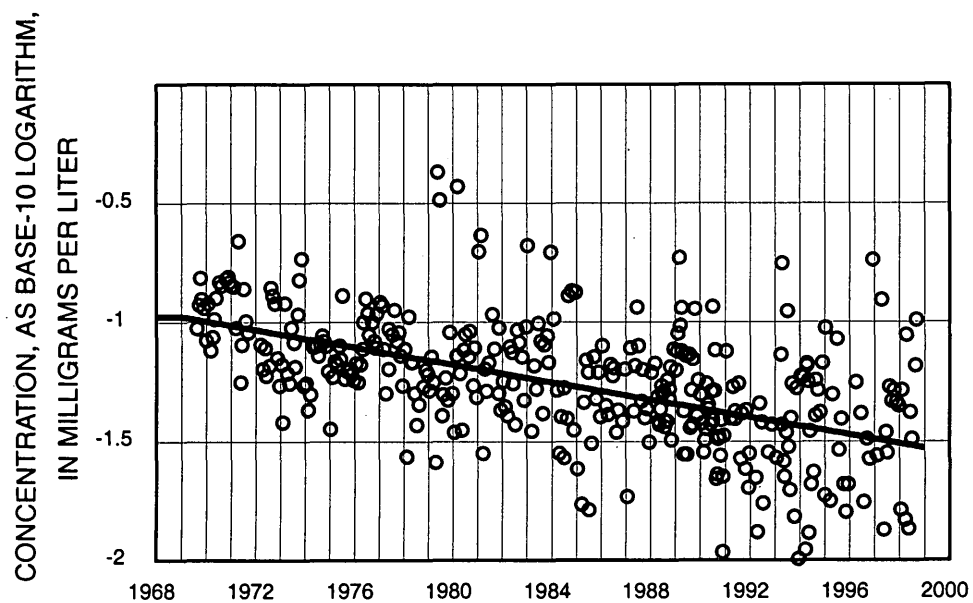


Figure 12a. Trends (lines) in flow-adjusted concentrations of total phosphorus (points), Connecticut River at Thompsonville, Conn., 1969-98.

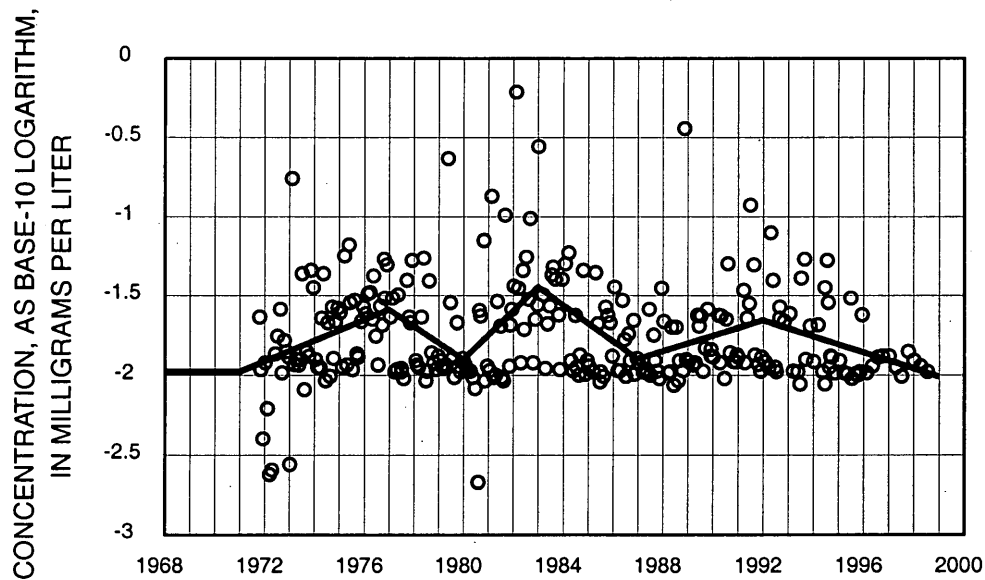


Figure 12b. Trends (lines) in flow-adjusted concentrations of total phosphorus (points), Salmon River near East Hampton, Conn., 1971-98.

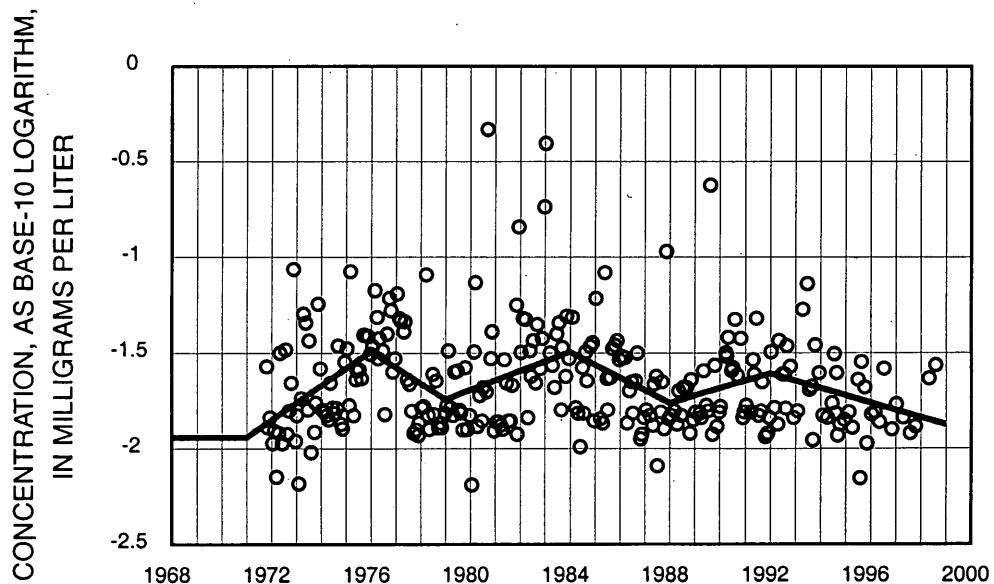


Figure 12c. Trends (lines) in flow-adjusted concentrations of total phosphorus (points), Saugatuck River near Redding, Conn., 1971-98.

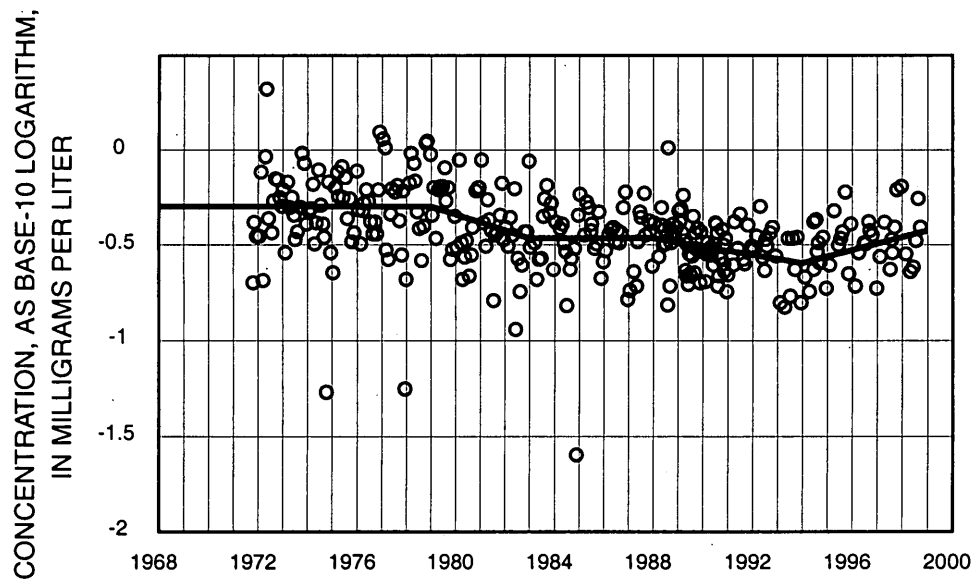


Figure 12d. Trends (lines) in flow-adjusted concentrations of total phosphorus (points), Quinnipiac River at Wallingford, Conn., 1971-98.

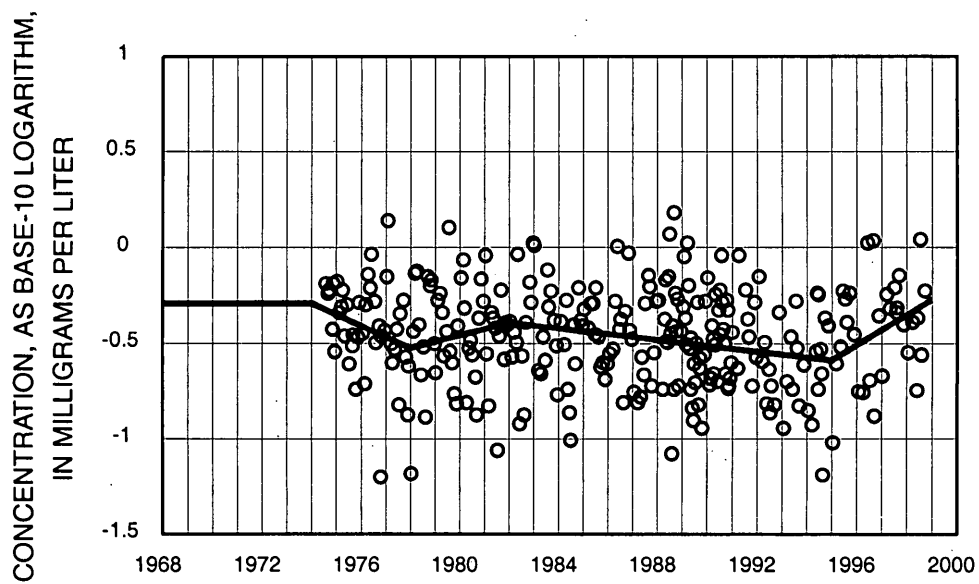


Figure 12e. Trends (lines) in flow-adjusted concentrations of total phosphorus (points), Naugatuck River at Beacon Falls, Conn., 1974-98.

Total Organic Carbon

Preliminary trend models showed that variability of total organic carbon concentrations and trend model residuals was generally substantially greater during the 1970's than later in the period of record. Consequently, although no information was available on method changes that might have caused this change in variability, the data sets were analyzed for possible step trends from the beginning of each period of record to the late 1970's or early 1980's. Positive step trends for this early period were detected for the Salmon, Saugatuck, and Quinnipiac Rivers (figs. 13b, c, and d). When linear trends were evaluated concurrently, no significant step trends were detected for this period for the Connecticut and Naugatuck Rivers (figs. 13a and e).

On the Connecticut and Quinnipiac Rivers, concentrations of total organic carbon increased during the 1970's, and then decreased from the late 1970's to

the early 1980's (figs. 13a and 13d). No significant trends were present during the 1990's. On the Salmon River, concentrations increased during the late 1970's and slightly again in the mid-1980's, then declined gradually during the remainder of the period of record (fig. 13b). On the Saugatuck River, concentrations increased during the 1970's, decreased in the early 1980's, and then increased again in the mid-1980's; a downward trend in the early 1990's was followed by a period of no trend for the remainder of the 1990's (fig. 13c). Concentrations of total organic carbon on the Naugatuck River decreased substantially during the 1970's and early 1980's (fig. 13e). Trend results for the Naugatuck show statistically significant increases and decreases throughout the period of record, with a sawtooth pattern overlaid on a general decrease in concentration. With the exception of the Naugatuck River, there were no large overall trends in the concentration of total organic carbon during the 1980's and 1990's.

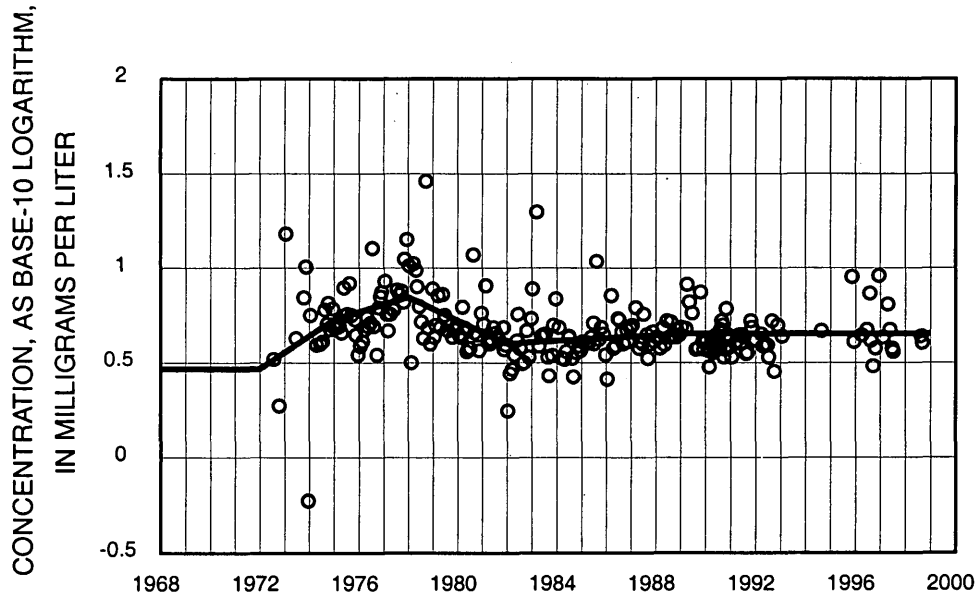


Figure 13a. Trends (lines) in flow-adjusted concentrations of total organic carbon (points), Connecticut River at Thompsonville, Conn., 1972–98.

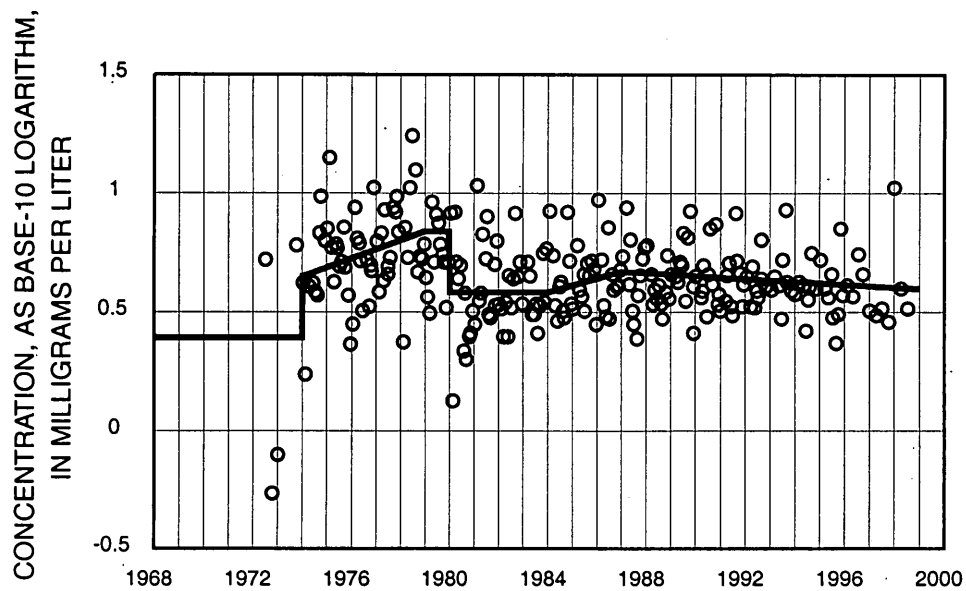


Figure 13b. Trends (lines) in flow-adjusted concentrations of total organic carbon (points), Salmon River near East Hampton, Conn., 1974–98.

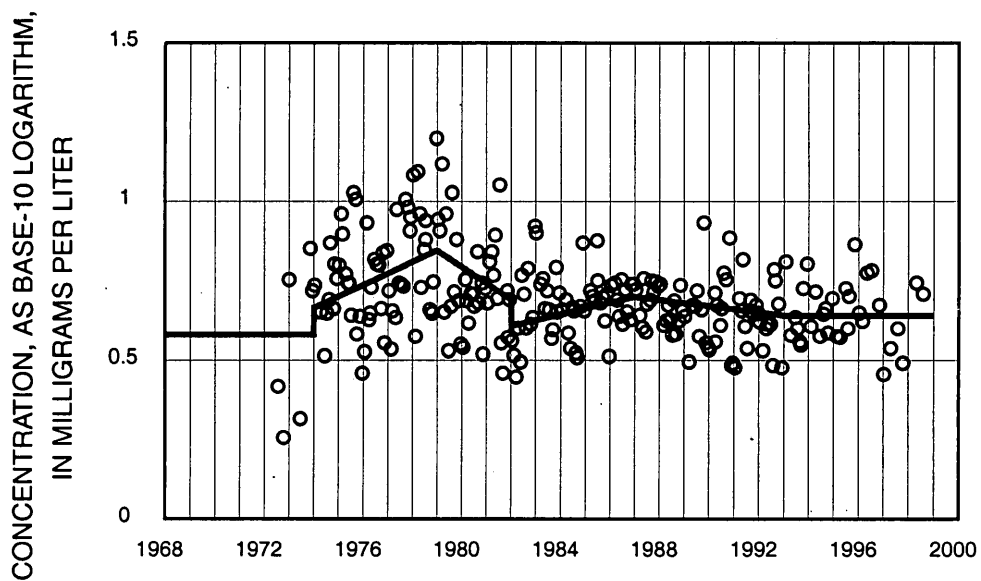


Figure 13c. Trends (lines) in flow-adjusted concentrations of total organic carbon (points), Saugatuck River near Redding, Conn., 1974–98.

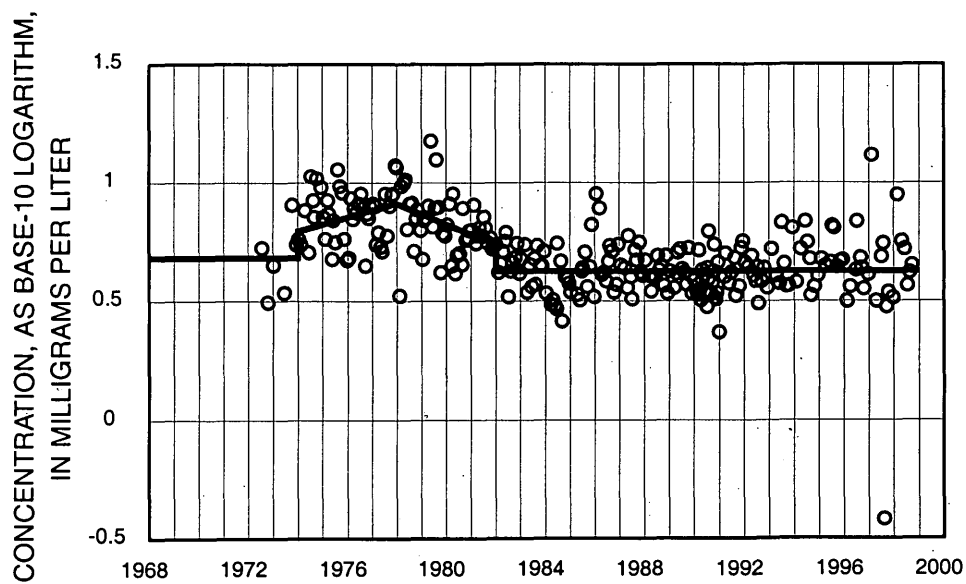


Figure 13d. Trends (lines) in flow-adjusted concentrations of total organic carbon (points), Quinnipiac River at Wallingford, Conn., 1974–98.

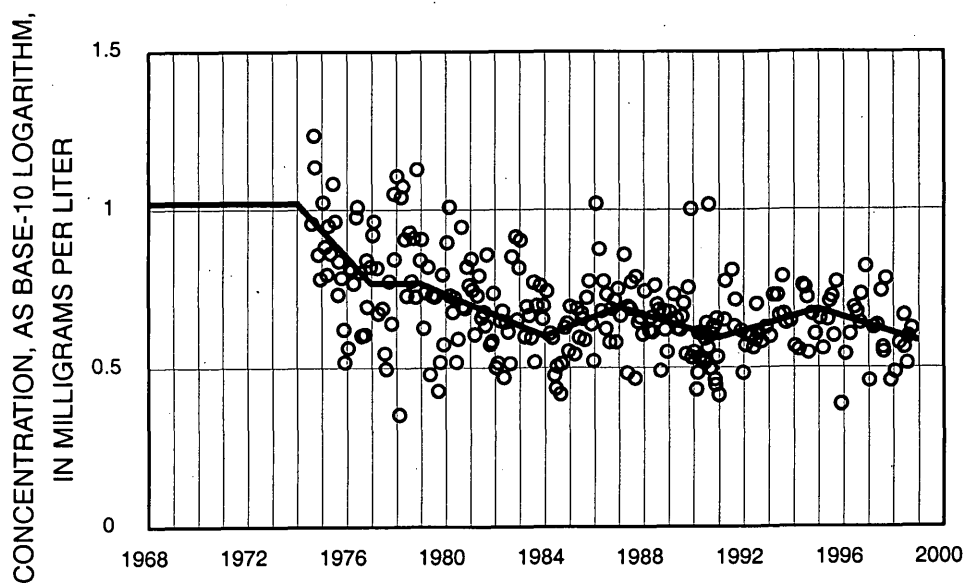


Figure 13e. Trends (lines) in flow-adjusted concentrations of total organic carbon (points), Naugatuck River at Beacon Falls, Conn., 1974–98.

Turbidity

Trend patterns for turbidity share similarities at all five stations (fig. 14); however, it should be noted that the ranges of turbidity values are sometimes distinctly different. Among these five streams, the Salmon and Saugatuck Rivers generally have the lowest turbidity values, the Connecticut and Naugatuck Rivers are in an intermediate range, and the Quinnipiac River has the highest values (table 4).

A statistically significant positive step trend is present for the period 1978–82 in the data for all five stations (fig. 14). The appearance of the data plots (shown in the appendix) suggests a change in method, instrument, or method sensitivity, although documentation is not available to substantiate this.

All five stations have a sawtooth pattern of decreases and increases in turbidity (fig. 14). Decreases in turbidity were detected in the early part of the period of record for the Connecticut, Salmon, Saugatuck Rivers, and Naugatuck Rivers. The longest periods of decrease, from 1978 to 1987, were on the Saugatuck and Naugatuck Rivers (figs. 14c and 14e). Turbidity at the five stations generally showed some increase between the mid-1980's and the early 1990's. The longest period of increase, from 1984 to 1992, took place on the Salmon River (fig. 14b). Turbidity decreased at all five stations during the early or mid-1990's, and then increased during the mid- to late 1990's. In some cases, the most recent upward trend is based on a small number of years and few samples, and the steepness of the trend shown in the plots (fig. 14) may not be sustained when additional years of record are available for analysis.

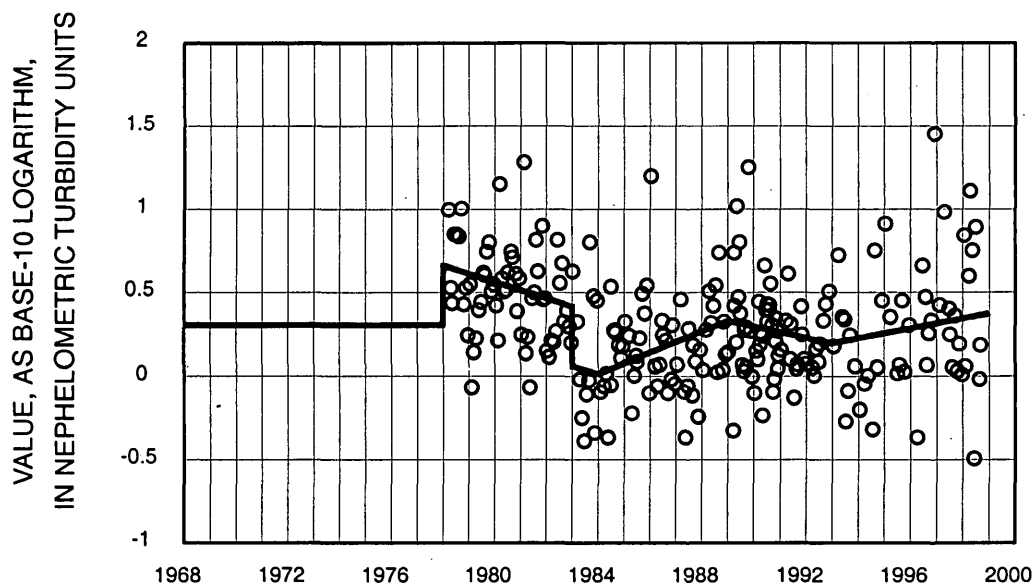


Figure 14a. Trends (lines) in flow-adjusted values of turbidity (points), Connecticut River at Thompsonville, Conn., 1978–98.

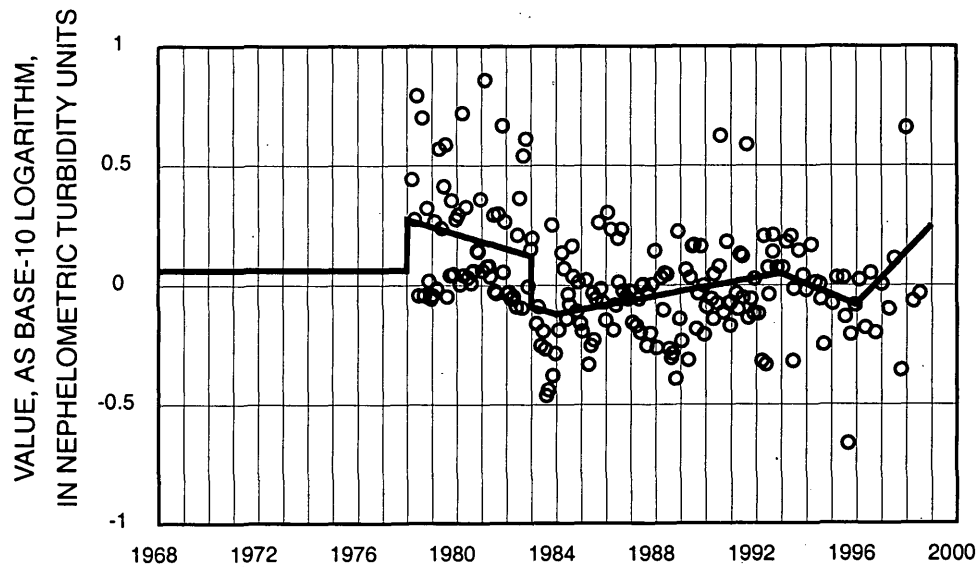


Figure 14b. Trends (lines) in flow-adjusted values of turbidity (points), Salmon River near East Hampton, Conn., 1978–98.

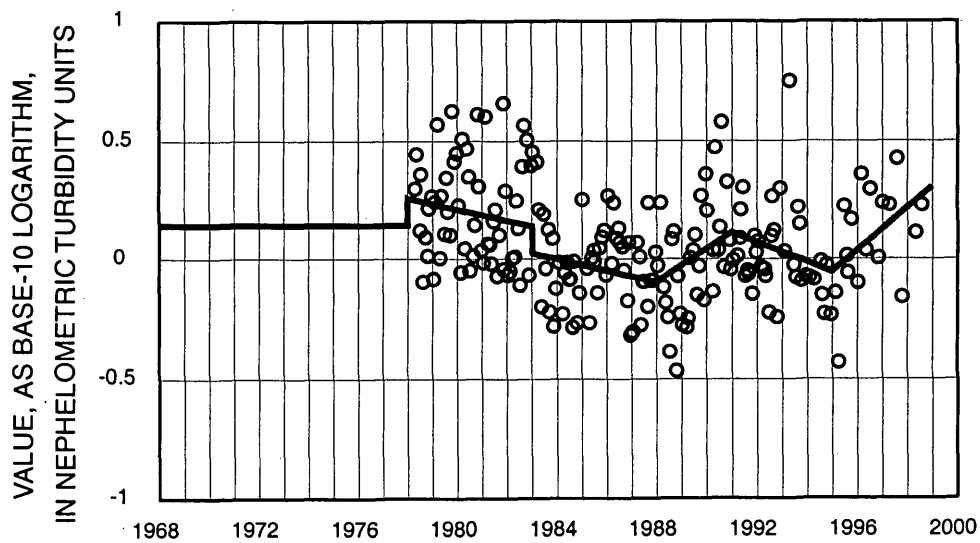


Figure 14c. Trends (lines) in flow-adjusted values of turbidity (points), Saugatuck River near Redding, Conn., 1978–98.

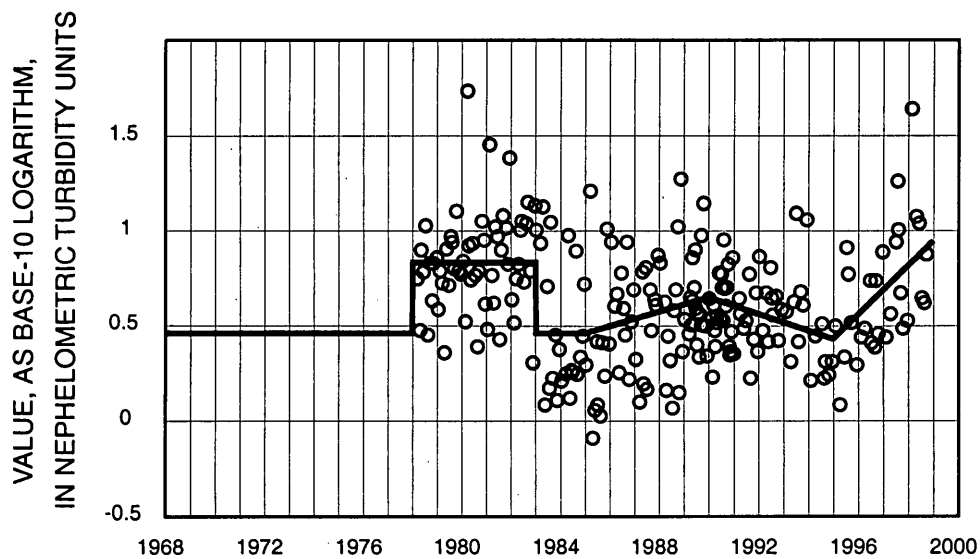


Figure 14d. Trends (lines) in flow-adjusted values of turbidity (points), Quinnipiac River at Wallingford, Conn., 1978–98.

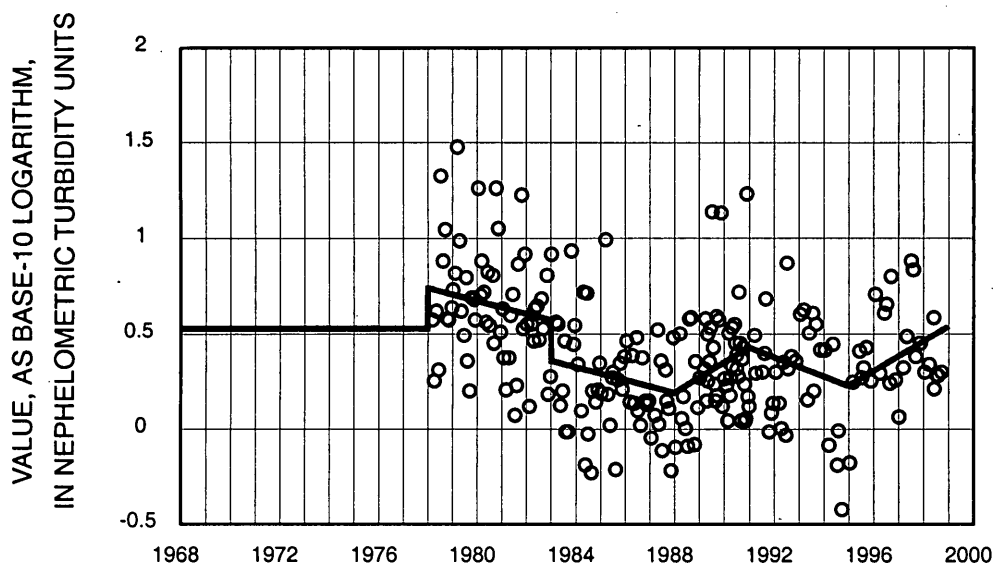


Figure 14e. Trends (lines) in flow-adjusted values of turbidity (points), Naugatuck River at Beacon Falls, Conn., 1978–98.

EVALUATION OF SAMPLING DESIGNS

Sampling design information can be evaluated in numerous ways, depending on the objectives and needs of the water-quality monitoring program. For example,

- * Optimal sampling designs, averaged for all constituents, can be identified.
- * Effects of one constituent on composite designs can be evaluated.
- * Optimal sampling designs for different constituents can be evaluated.
- * Sampling frequencies for a given design period can be evaluated relative to the size of detectable trends.
- * Sampling frequencies can be evaluated over time relative to the size of detectable trends.
- * The current monitoring program can be evaluated in terms of overall optimal sampling frequency and seasonal distribution of samples for future trend analysis.

Selected optimal sampling designs for the five monitoring stations evaluated in this report are discussed in the following sections, and examples of additional types of interpretation are presented. Design

results for specific stations are used to introduce and illustrate general characteristics of the designs. A complete analysis and presentation of all aspects of the design information is beyond the scope of this report.

Alternative sampling designs presented in this report include designs that work well for most constituents, are logistically feasible to implement, and maintain the ability to detect trends at a level comparable to or better than that provided by the current sampling schedule. The information gained or lost by increasing or decreasing the number of samples from current levels is discussed.

General Characteristics of Design Results

Selected design results for the Quinnipiac River are presented here in detail to introduce and clarify the general characteristics of design results for all stations. As noted in the Methods section of this report, designs are evaluated in terms of their efficiency and power. Design efficiency is measured in terms of the size (in percent change) of the detectable trend over the design period. The more efficient the design, the smaller the trend that can be detected. Power refers to the probability of detecting a trend of given size.

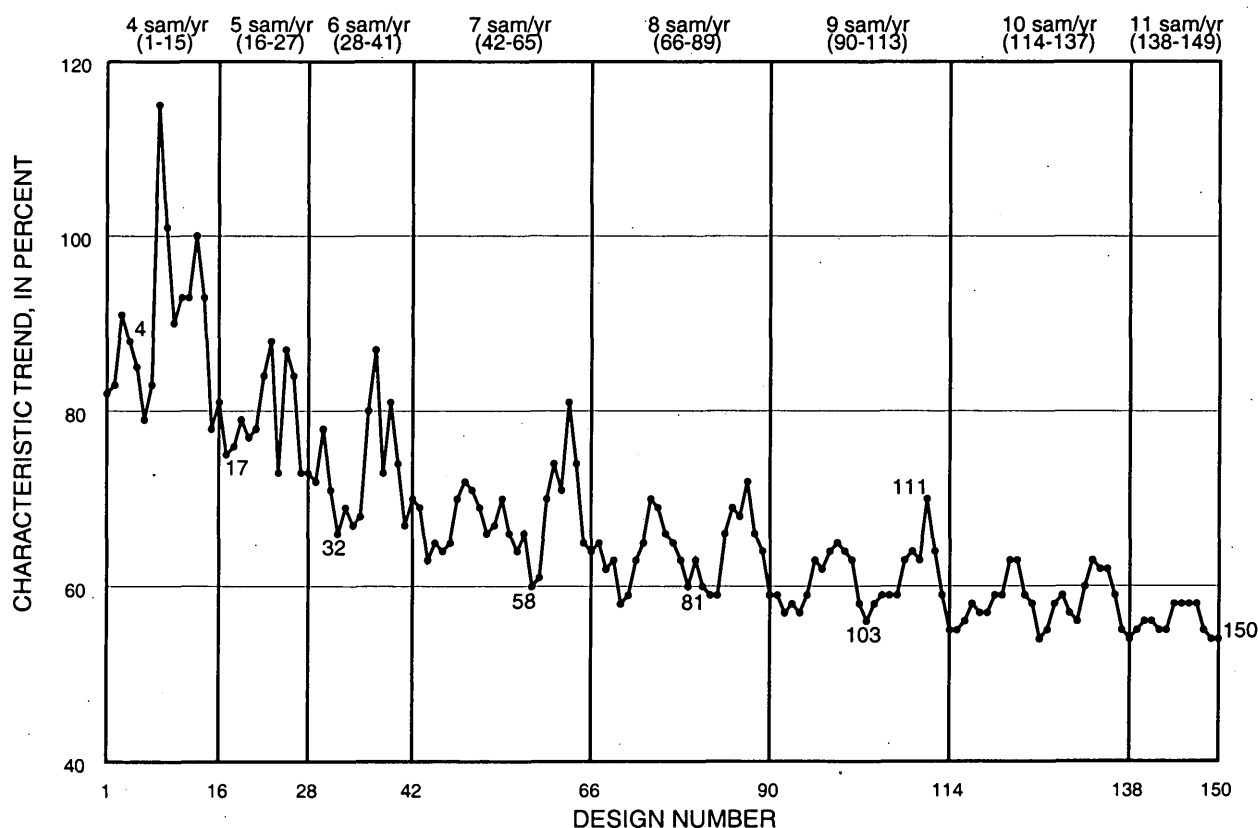


Figure 15. Magnitude of detectable characteristic trends in total phosphorus, at a fixed power of 0.8, for sample designs based on 4 through 12 samples per year, for the Quinnipiac River at Wallingford, Conn.

The sensitivity of a design can be looked at in two ways. One way is to fix the power to detect trends (for example, at 0.8, or 80 percent probability) and see how the characteristic (detectable) trend changes for the different designs (fig. 15). The other way is to fix the size of the trend and see how the power changes for the different designs (fig. 16).

Design efficiencies for detecting a trend in total phosphorus for the Quinnipiac River over a 5-year design period are shown in figure 15. Points on the graph represent the efficiencies of 150 specific sampling designs, numbered from left to right (fig. 15). The probability of detecting the characteristic trend is fixed at 0.8 (80 percent) for each design. Design 150, the only design with monthly sampling, is the most sensitive; that is, it has the smallest characteristic trend, about 56 percent over the design period. Designs with only four samples per year include the least efficient designs; that is, the characteristic trends must be quite large in order to have an 80 percent probability of detection with this low frequency of sampling.

Designs with larger numbers of samples are not necessarily more efficient. For example, the character-

istic trend for total phosphorus for design 32 (one of the 6-sample designs) is about 66 percent (fig. 15). If phosphorus concentrations at the end of the 5-year design period are 66 percent higher than at the beginning of the design period, there is an 80 percent chance that the increase will be detected using design 32. The characteristic trend for design 111 (one of the 9-sample designs) is about 70 percent (fig. 15). Consequently, design 111 is less efficient for detecting a trend in total phosphorus than design 32, even though design 111 has 3 more samples per year than design 32. Determining when to sample is therefore as important as determining how many samples to collect.

The power of designs for detecting a trend in total phosphorus concentration over a 5-year design period is shown in figure 16. Points on the graph in figure 16 plot the power for 150 specific sampling designs, numbered from left to right. In figure 15, the power is fixed at 0.8 and the size of the characteristic trend varies, whereas in figure 16, the size of the characteristic trend is fixed at 56 percent (the size of the detectable trend with monthly sampling, design 150),

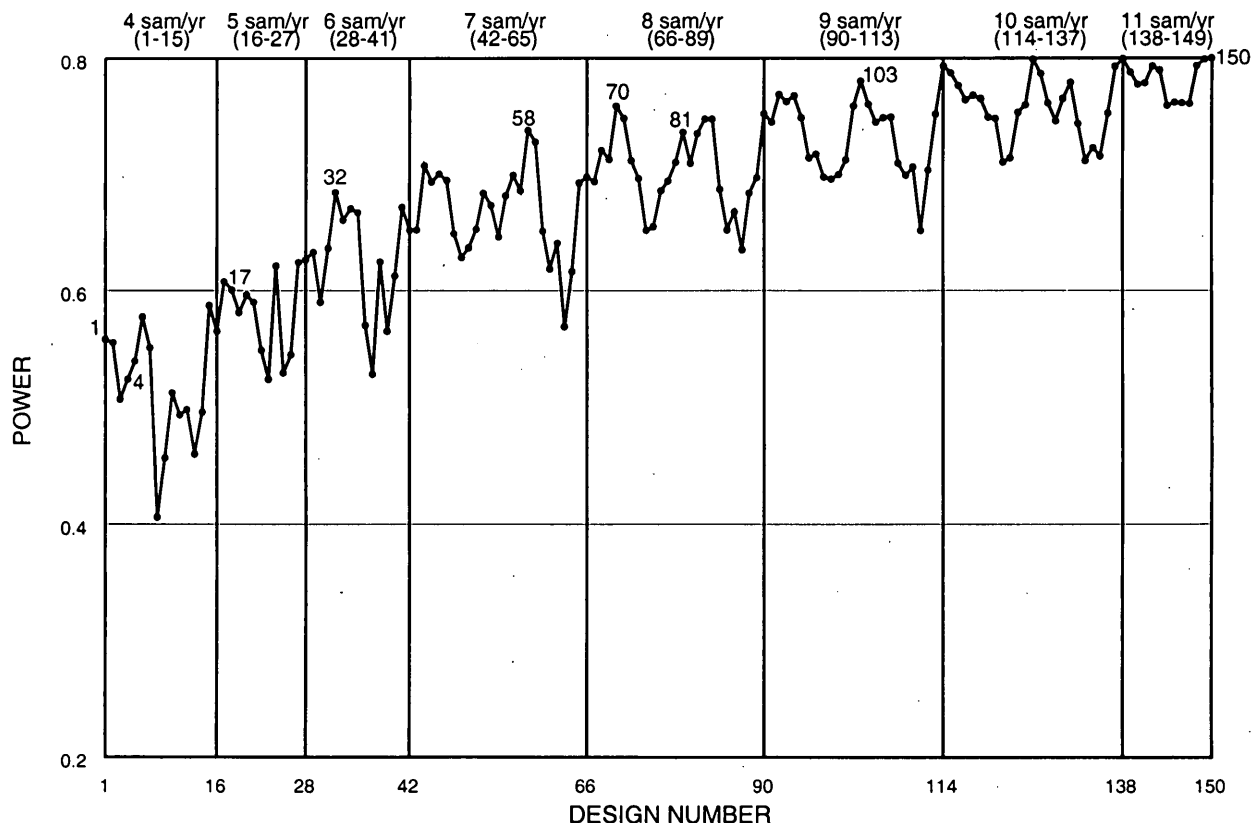


Figure 16. Power of the statistical test to detect trends in total phosphorus, at a fixed characteristic trend of 56 percent, for sample designs based on 4 through 12 samples per year, for the Quinnipiac River at Wallingford, Conn.

and the power, or probability of detection, varies. Designs with fewer than 12 samples per year have less than an 80 percent probability of detecting a change in total phosphorus concentration of 56 percent over the 5-year design period. Figure 16 conveys essentially the same information as figure 15, except that efficient designs for a given sample size are now those with high power. Thus, for example, several of the 8-sample designs have power nearly as high as design 150, and these designs correspond to the designs in figure 15 with low characteristic trends.

When comparing designs for different constituents, figure 16 has a distinct advantage over figure 15. Namely, the power is dimensionless and can be plotted on the same scale for every constituent. For example, the power of the designs for dissolved sulfate for the

Quinnipiac station are shown in figure 17. Efficient designs for different constituents may not coincide. Design number 70, which was the most efficient 8-sample design for total phosphorus (fig. 16), is a relatively inefficient design for dissolved sulfate (fig. 17).

Because the power of the various designs can be plotted on the same scale for all constituents, the overall power of a specific design for several constituents can be defined as the average (mean) power of the design over all the constituents. The optimal design for a given sample size is defined as the design with the highest average power over all constituents in the design. The optimal multi-constituent design is thus a composite design, and the power for that design is termed the composite power in the following discussion of design results.

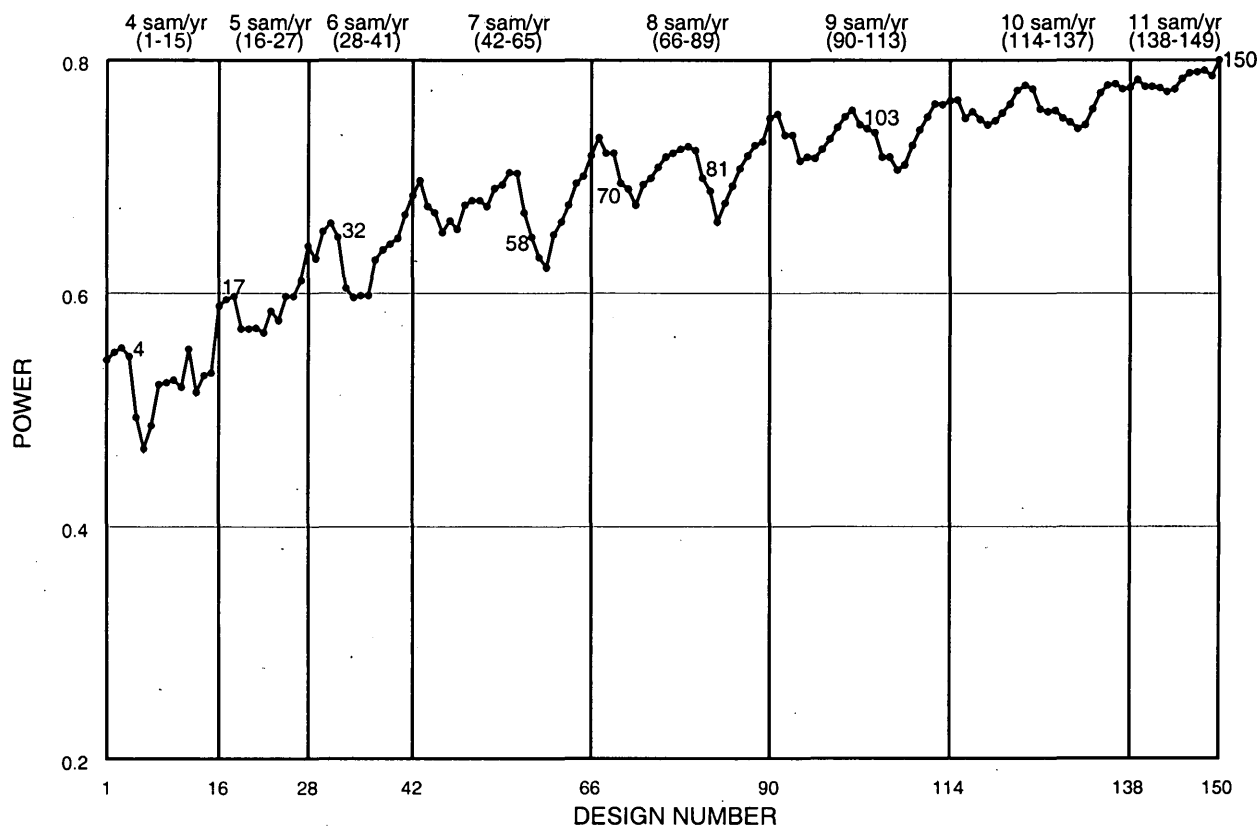


Figure 17. Power of the statistical test to detect trends in dissolved sulfate, for sample designs based on 4 through 12 samples per year, for the Quinnipiac River at Wallingford, Conn.

Evaluation of Optimal Sampling Designs at Different Stations

Optimal sampling designs for the Connecticut, Salmon, Saugatuck, Quinnipiac, and Naugatuck Rivers are shown in figures 18 through 22. Information shown for each station includes six designs, each being the optimal design for one of six sampling frequencies ranging from 4 through 9 samples per year. Each figure presents the length of the design period and a list of constituents with their characteristic trends in a box in the upper-left corner. A plot labeled "Optimal Designs" in the upper-right corner of each figure shows the sampling schedules for the numbered designs; months with water-quality samples are indicated by the solid boxes. Bar graphs show the power of the trend test for each individual constituent and the composite power (labeled C) for each of the six designs. The results for each station use a design period of 5 years and include the six constituents evaluated for trend in this study. The size of the characteristic trend is fixed at the indicated number for all designs for that station, while the power of each design varies.

The size of the characteristic trend for each constituent is defined so that the power equals 0.8 with monthly sampling. For example, the best 4-sample design for the Quinnipiac River (design 4 in fig. 21) shows considerable loss of power compared to monthly sampling, especially for total organic carbon and turbidity. The best 8-sample design for the Quinnipiac River (design 81) has much better power relative to the monthly design, with the power for all constituents well above 0.6. Thus, design 81 might be selected as a good overall design for the Quinnipiac River. An increase in sampling frequency to 9 samples per year does not provide a large increase in power.

Comparison of optimal sampling designs for the five stations evaluated shows some differences among stations. A 5-year design period with eight observations per year is considered in the following discussion. Other sampling frequencies or design periods also could be compared.

A qualitative look at the design summary plots for the five stations (figs. 18–22) shows that for an 8-sample design, the optimal time for monthly sampling is from spring to fall for the Connecticut and Quinnipiac Rivers (design number 81 on figs. 18 and 21),

and from winter to spring or summer for the Salmon, Saugatuck, and Naugatuck Rivers. The six best 8-sample designs (only the first-ranked design is shown in the figures) were compared among the five stations. The two best 8-sample designs for both the Salmon and Saugatuck Rivers are design numbers 89 and 78, with monthly sampling from December to May or January to June (figs. 19 and 20). These two designs also are among the best 8-sample designs for the Naugatuck River (ranking 5th and 6th), although design number 67 is ranked as best (fig. 22), with monthly sampling from February to June.

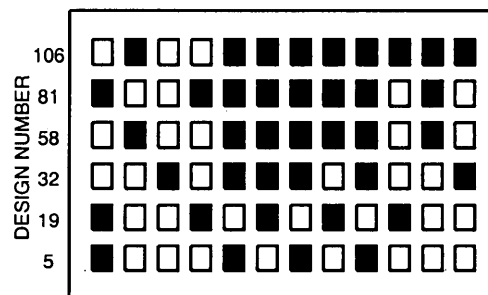
The Connecticut and Quinnipiac Rivers have the most similarities in results for the six best 8-sample designs, with five of six of the best designs in common. Three of the best 8-sample designs for the Quinnipiac River also are shared by the Salmon and Naugatuck Rivers. The Saugatuck River has the fewest similarities to other stations in its design requirements, with four of the six best 8-sample designs not shared by other stations.

The magnitude of trends that can be detected usually varies widely among constituents. For example, for the Connecticut River, the trend in turbidity (109 percent) would need to be more than 5 times as large as the trend in dissolved sulfate (20 percent) to have a comparable power of detection (fig. 18). This difference is due largely to much higher variability of the noise in turbidity than the noise in sulfate, as shown in trend plots of flow-adjusted values over time (compare figs. 10a and 14a). It is not feasible to develop a design with the same trend magnitude for all constituents. For example, referring to the data shown in figure 10a, a 109-percent increase in sulfate over a 5-year period (an increase of 0.32 in the base-10 logarithm) would result in concentrations well outside the range of historical values. Similarly, referring to figure 14a, a 20-percent increase in turbidity over a 5-year period (an increase of 0.08 in the base-10 logarithm) would barely be discernible. Varying the trend magnitudes so that the power of detection is comparable for each constituent provides a convenient way to remove differences due to variability in the noise. Thus, bar heights in the graphs of design power (figs. 18 to 22) are generally similar for a given sampling frequency at a station.

CONSTITUENT AND DESIGN INFORMATION

Design period: 5 years
 Label - Constituent - Characteristic trend (percent)
 C - Composite - 52.8
 1 - DISSOLVED CHLORIDE - 31.1
 2 - DISSOLVED SULFATE - 19.8
 3 - TOTAL NITROGEN - 38.9
 4 - TOTAL PHOSPHORUS - 85.7
 5 - TOTAL ORGANIC CARBON - 32.2
 6 - TURBIDITY - 109.1

OPTIMAL DESIGNS



J F M A M J J A S O N D
 (Solid box indicates that water-quality sample is collected)

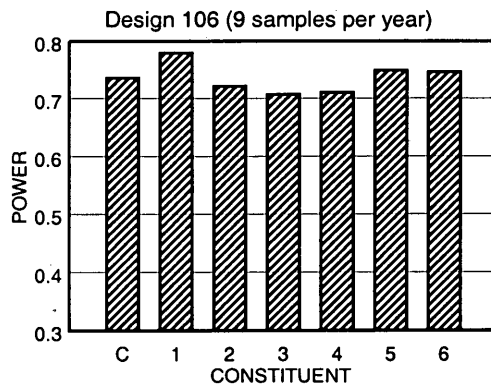
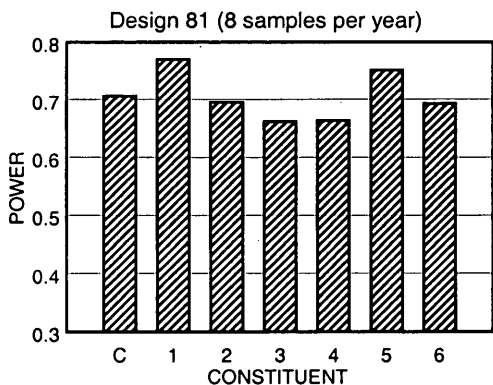
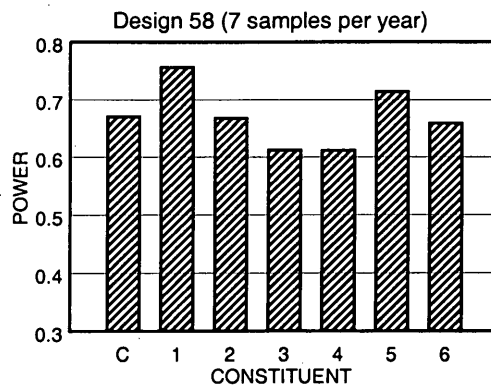
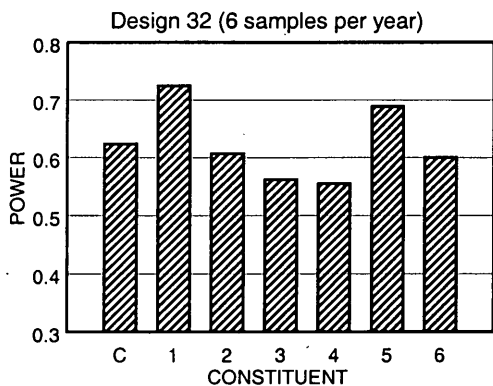
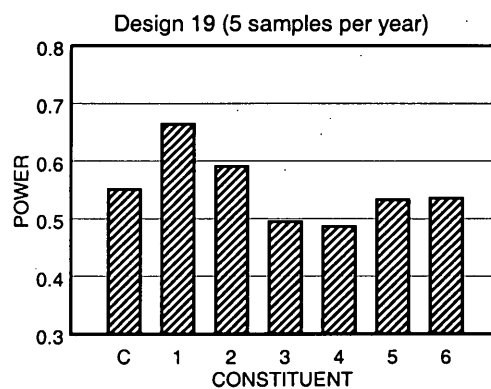
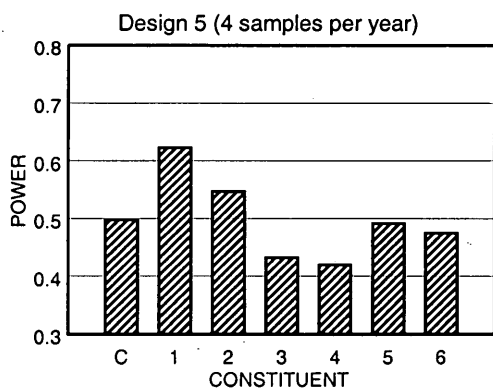


Figure 18. Optimal sampling designs, based on the average (or composite, C) power of the design over all constituents, for 4 through 9 samples per year, for the Connecticut River at Thompsonville, Conn.

CONSTITUENT AND DESIGN INFORMATION

Design period: 5 years
 Label - Constituent - Characteristic trend (percent)
 C - Composite - 49.5
 1 - DISSOLVED CHLORIDE - 16.6
 2 - DISSOLVED SULFATE - 12.7
 3 - TOTAL NITROGEN - 55.2
 4 - TOTAL PHOSPHORUS - 111.6
 5 - TOTAL ORGANIC CARBON - 42.4
 6 - TURBIDITY - 58.4

OPTIMAL DESIGNS

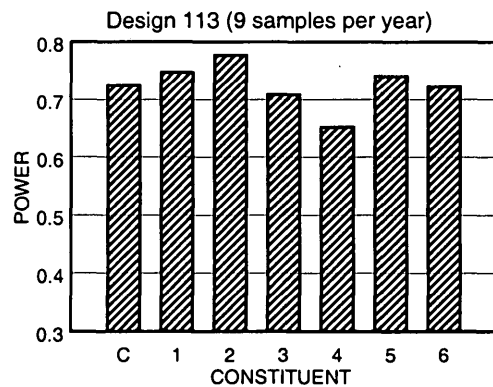
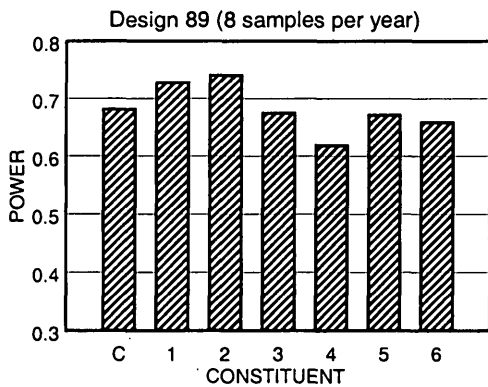
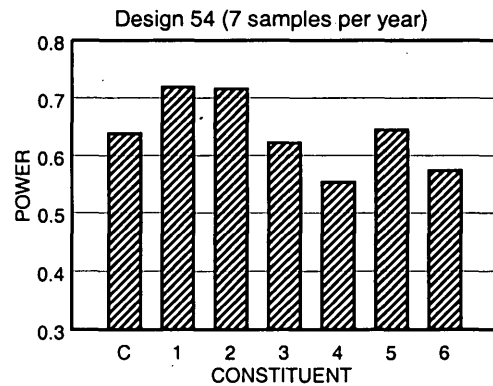
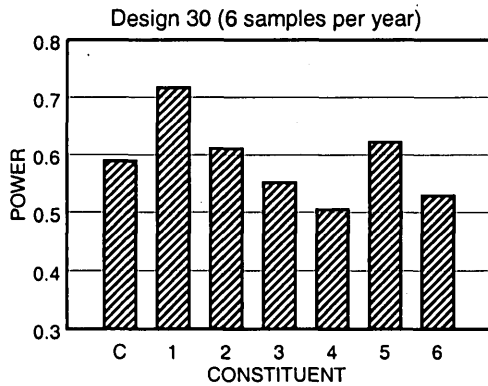
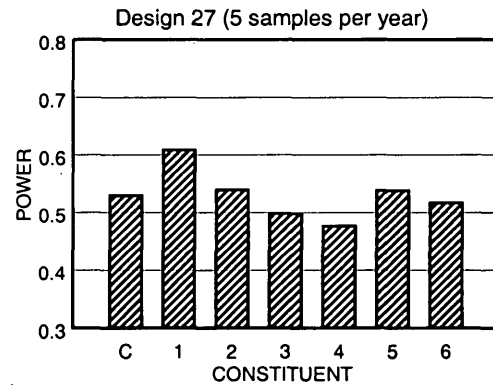
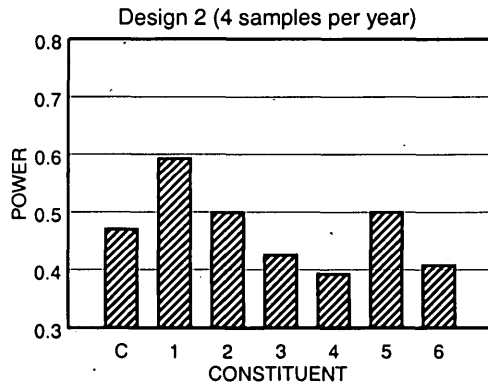
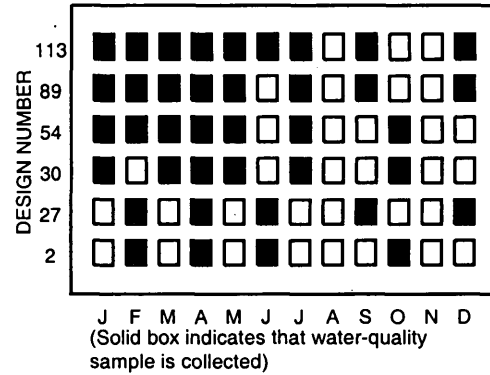
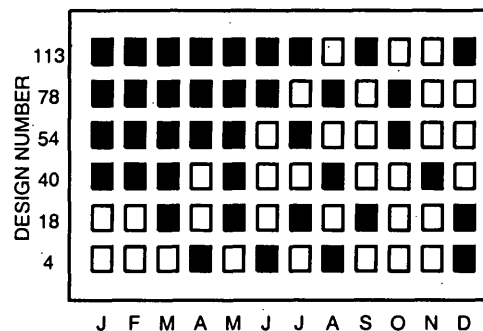


Figure 19. Optimal sampling designs, based on the average (or composite, C) power of the design over all constituents, for 4 through 9 samples per year, for the Salmon River at East Hampton, Conn.

CONSTITUENT AND DESIGN INFORMATION

Design period: 5 years
 Label - Constituent - Characteristic trend (percent)
 C - Composite - 44.9
 1 - DISSOLVED CHLORIDE - 24.5
 2 - DISSOLVED SULFATE - 13.3
 3 - TOTAL NITROGEN - 56
 4 - TOTAL PHOSPHORUS - 81.7
 5 - TOTAL ORGANIC CARBON - 35.3
 6 - TURBIDITY - 58.6

OPTIMAL DESIGNS



(Solid box indicates that water-quality sample is collected)

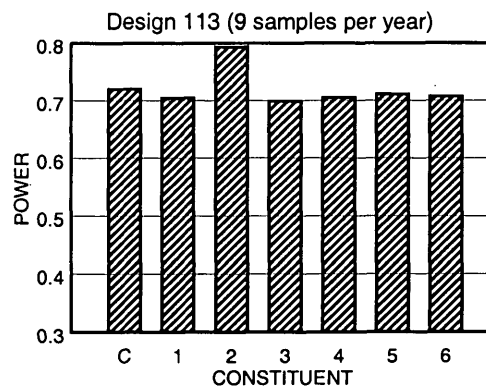
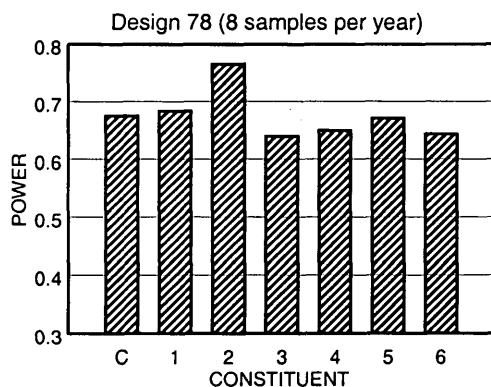
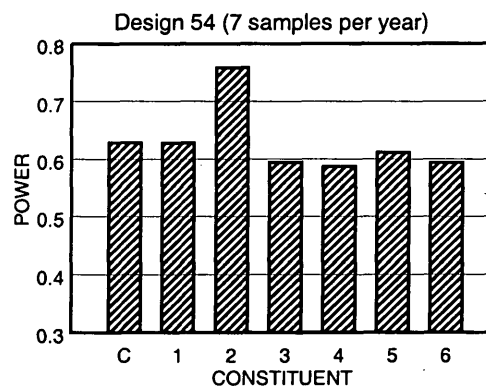
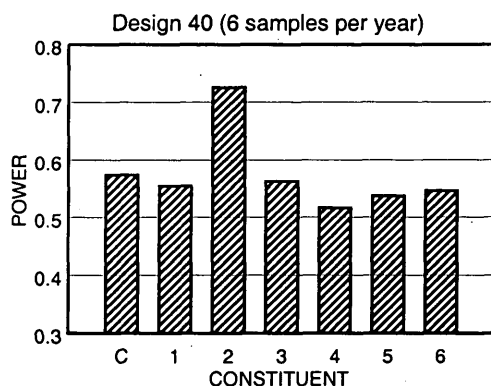
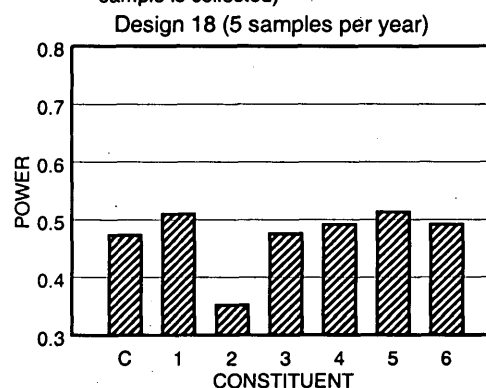
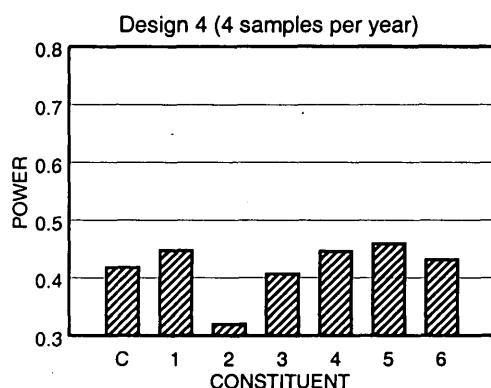
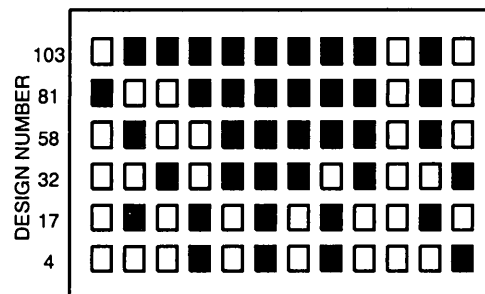


Figure 20. Optimal sampling designs, based on the average (or composite, C) power of the design over all constituents, for 4 through 9 samples per year, for the Saugatuck River near Redding, Conn.

CONSTITUENT AND DESIGN INFORMATION

Design period: 5 years
 Label - Constituent - Characteristic trend (percent)
 C - Composite - 41.7
 1 - DISSOLVED CHLORIDE - 24.2
 2 - DISSOLVED SULFATE - 19.3
 3 - TOTAL NITROGEN - 35.3
 4 - TOTAL PHOSPHORUS - 54.1
 5 - TOTAL ORGANIC CARBON - 31.2
 6 - TURBIDITY - 86.3

OPTIMAL DESIGNS



J F M A M J J A S O N D

(Solid box indicates that water-quality sample is collected)

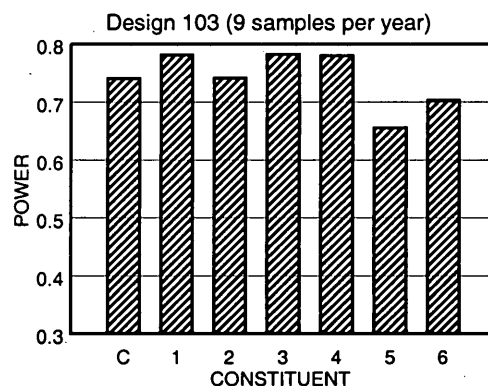
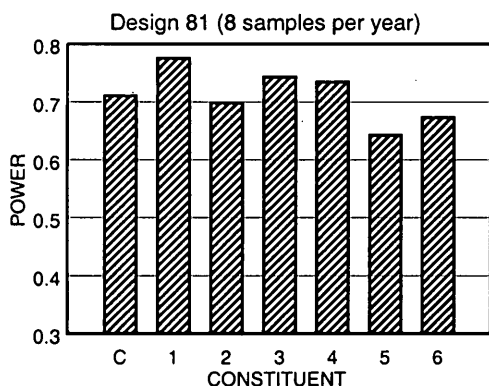
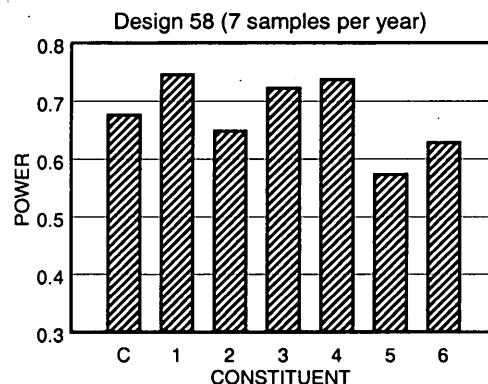
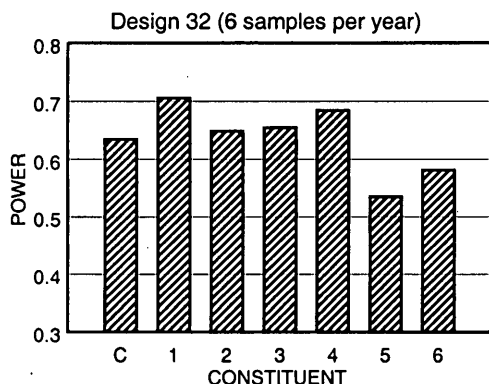
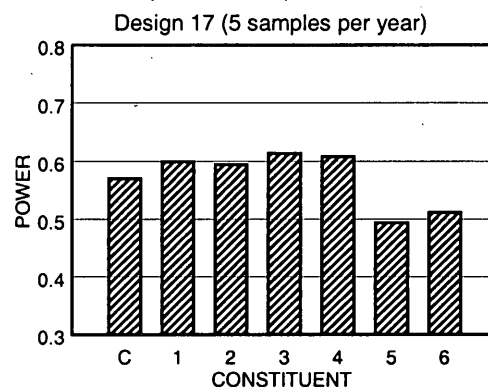
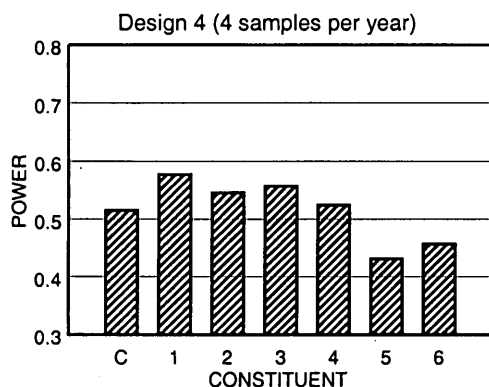


Figure 21. Optimal sampling designs, based on the average (or composite, C) power of the design over all constituents, for 4 through 9 samples per year, for the Quinnipiac River at Wallingford, Conn.

CONSTITUENT AND DESIGN INFORMATION

Design period: 5 years
 Label - Constituent - Characteristic trend (percent)
 C - Composite - 57.2
 1 - DISSOLVED CHLORIDE - 43.4
 2 - DISSOLVED SULFATE - 33.7
 3 - TOTAL NITROGEN - 55
 4 - TOTAL PHOSPHORUS - 83.6
 5 - TOTAL ORGANIC CARBON - 35.9
 6 - TURBIDITY - 91.5

OPTIMAL DESIGNS

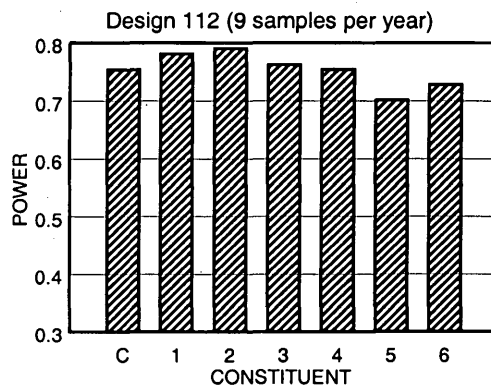
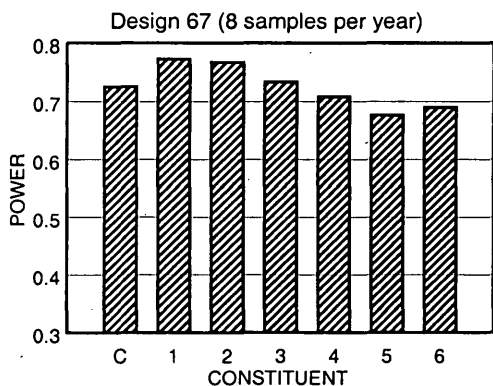
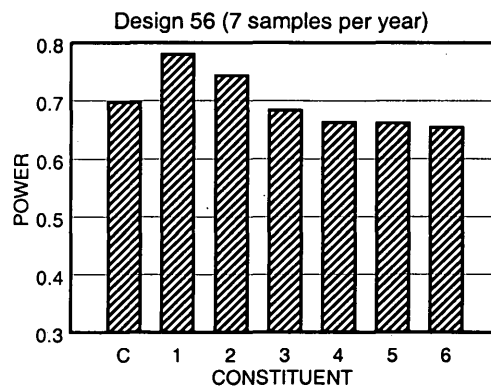
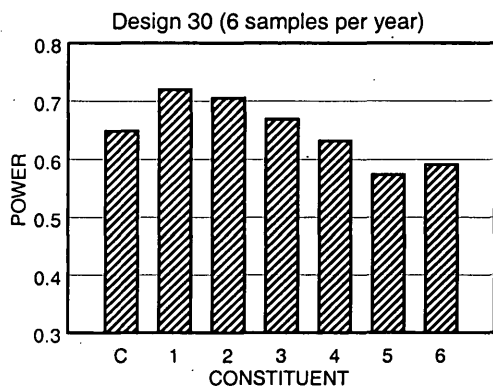
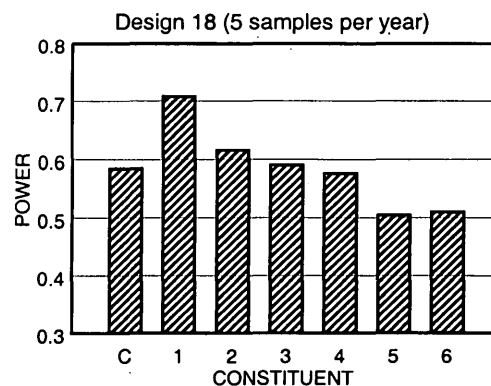
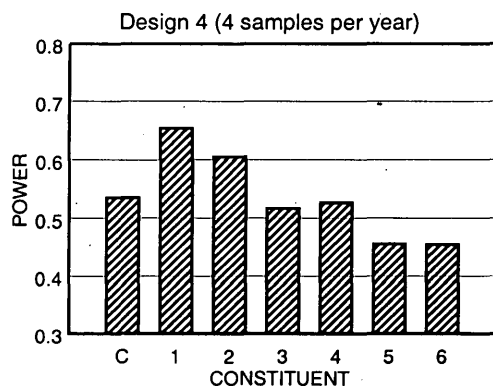
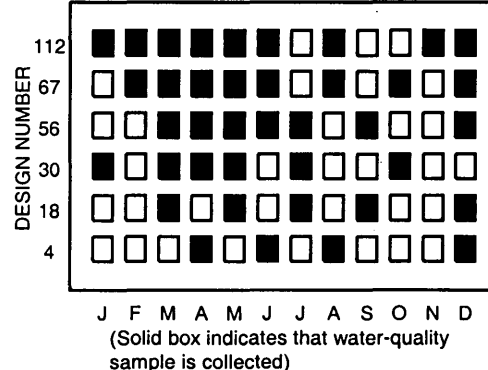


Figure 22. Optimal sampling designs, based on the average (or composite, C) power of the design over all constituents, for 4 through 9 samples per year, for the Naugatuck River at Beacon Falls, Conn.

Effects of One Constituent on Optimal Composite Designs

Design results were investigated in more detail to determine potential causes for the differences in sample timing among the five stations. For both the Salmon and Saugatuck stations, it was determined that sulfate was a key constituent in selection of designs. Designs with high power for sulfate tended to include monthly sampling during winter. As an exercise to examine the effect of one constituent on composite designs, optimal designs for the Salmon and Saugatuck stations were redetermined by maximizing the average power over all constituents except sulfate (figs. 23 and 24). When sulfate is excluded, design 81 is the best 8-sample design for both stations—the same 8-sample design that was the best design for the Connecticut and Quinnipiac stations. Although design 81 was not the best 8-sample design for the Naugatuck station, further examination of the design results showed that design 81 may be an acceptable design for the Naugatuck, depending on the importance of specific constituents at that station. Thus, design 81 may be a reasonable design for all 5 stations under some circumstances. This exercise demonstrates how design requirements for individual constituents can affect optimal composite designs; it is not intended as an evaluation of the relative importance of sulfate.

Evaluation of Optimal Sampling Designs for Different Constituents

Optimal sampling designs that balance the power of the trend test over all constituents provide an efficient way to plan sampling for a complex program that involves numerous stations and constituents. Optimal composite designs, however, may not represent the best designs for specific constituents. Optimal designs for individual constituents can be examined to augment the information provided by composite designs. Implementation of optimal sampling designs

for individual constituents may not be logistically feasible however, because every constituent may have a different optimal sampling schedule (Vecchia, 2000, p. 26).

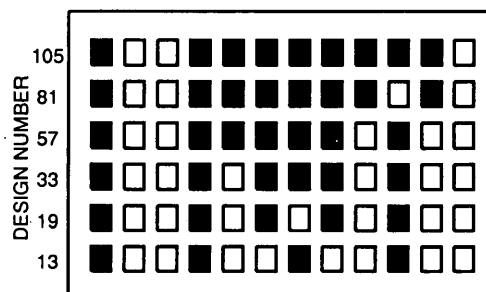
For example, for the Connecticut River at Thompsonville, the optimal 8-sample design, balancing all six constituents in the analysis, includes monthly sampling from April to September and additional samples in November and January (fig. 18). When the 8-sample designs for total nitrogen alone are examined, however, the six best 8-sample designs include monthly sampling in November, December, and January, and five of the six designs include sampling in February as well. Four of the six designs include sampling in March, and four of the six include October. Thus, optimal monthly sampling for total nitrogen is during the winter months, the opposite of the optimal composite design for this station.

Using one of most efficient 8-sample designs for total nitrogen results in a relatively small gain, in terms of the reduction in the size of the characteristic trend for the Connecticut River, compared to other 8-sample designs (fig. 25). One way to improve trend detection for total nitrogen, while retaining the composite schedule, would be to add one or more sampling dates for nutrients in the winter months. In the case of the Salmon River, by contrast, the optimal composite design in the absence of sulfate, as described previously, is design number 81, which is the least efficient 8-sample design for total nitrogen (fig. 26). Although the inclusion of total nitrogen does not drive the optimal designs to the extent that sulfate does, power for total nitrogen in design 81 for the Salmon River (without sulfate, fig. 23) is less than the power for other constituents in design 81, and less than the power for total nitrogen in design 89 (all constituents, fig. 19). The power for total nitrogen in design 81 also is less than the power for most other constituents for the Connecticut and Saugatuck Rivers (figs. 18 and 24).

CONSTITUENT AND DESIGN INFORMATION

Design period: 5 years
 Label - Constituent - Characteristic trend (percent)
 C - Composite - 49.5
 1 - DISSOLVED CHLORIDE - 16.6
 2 - DISSOLVED SULFATE - 12.7
 3 - TOTAL NITROGEN - 55.2
 4 - TOTAL PHOSPHORUS - 111.6
 5 - TOTAL ORGANIC CARBON - 42.4
 6 - TURBIDITY - 58.4

OPTIMAL DESIGNS



J F M A M J J A S O N D
 (Solid box indicates that water-quality sample is collected)

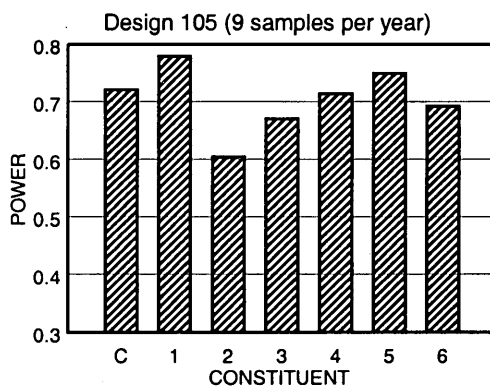
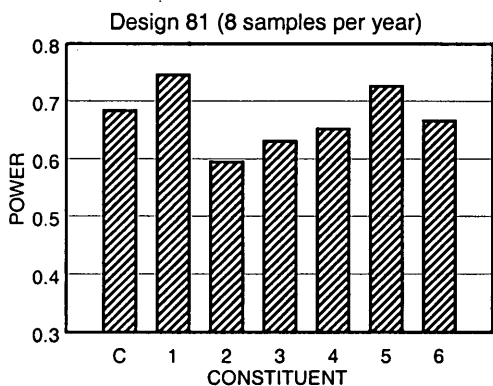
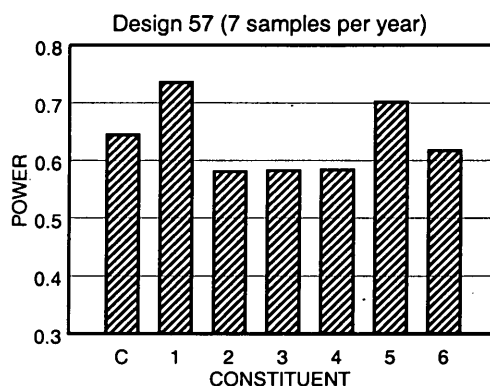
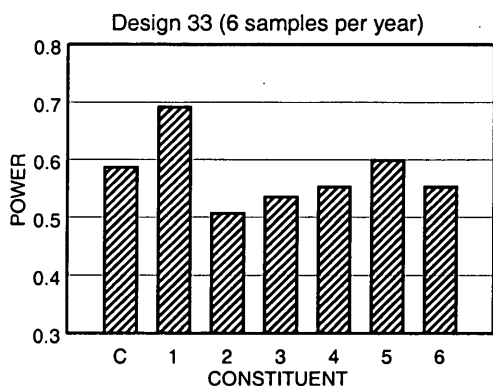
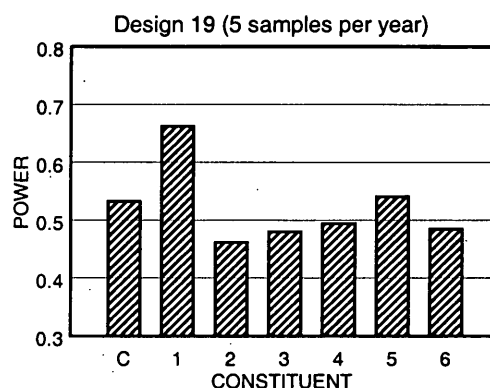
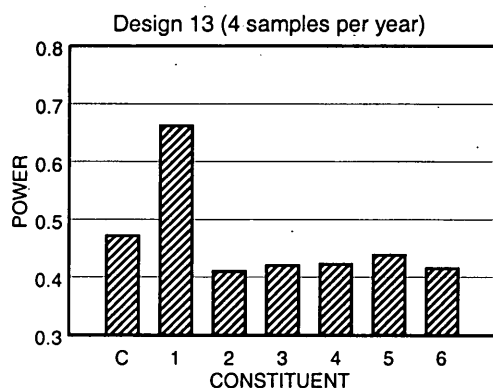


Figure 23. Optimal sampling designs, based on the average (or composite, C) power of the design over all constituents (excluding sulfate), for 4 through 9 samples per year, for the Salmon River at East Hampton, Conn..

CONSTITUENT AND DESIGN INFORMATION

Design period: 5 years
 Label - Constituent - Characteristic trend (percent)
 C - Composite - 44.9
 1 - DISSOLVED CHLORIDE - 24.5
 2 - DISSOLVED SULFATE - 13.3
 3 - TOTAL NITROGEN - 56
 4 - TOTAL PHOSPHORUS - 81.7
 5 - TOTAL ORGANIC CARBON - 35.3
 6 - TURBIDITY - 58.6

OPTIMAL DESIGNS

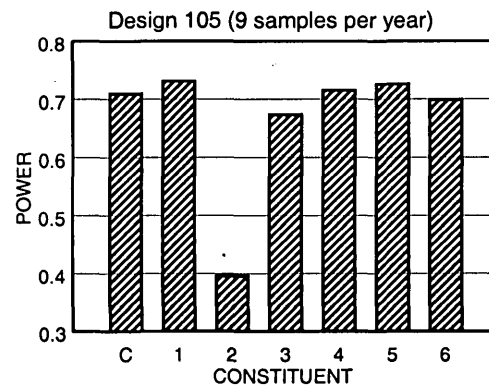
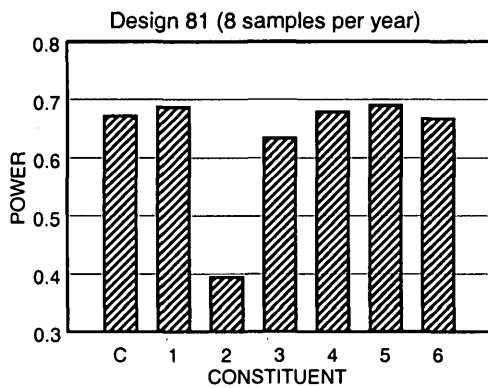
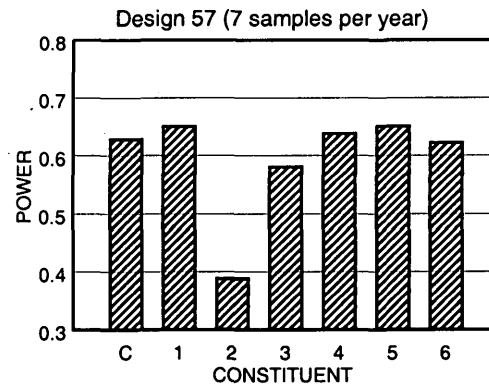
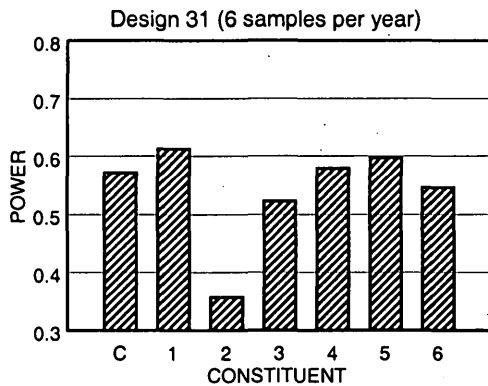
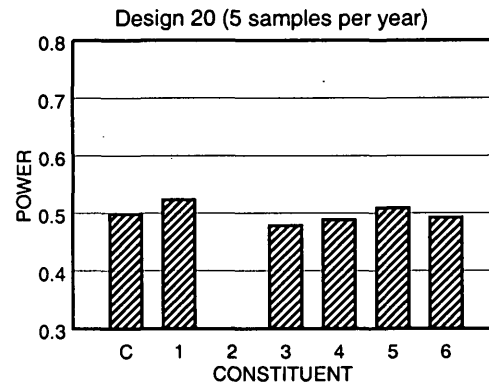
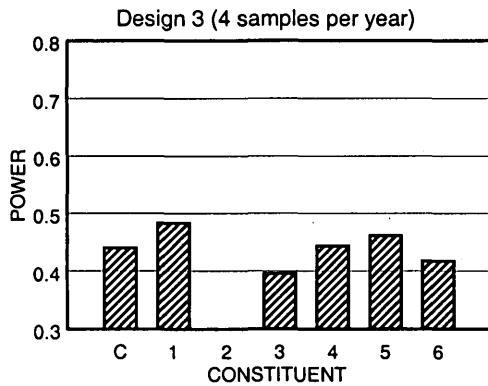
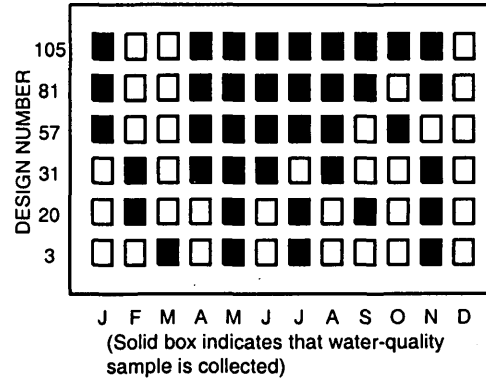


Figure 24. Optimal sampling designs, based on the average (or composite, C) power of the design over all constituents (excluding sulfate), for 4 through 9 samples per year, for the Saugatuck River near Redding, Conn.

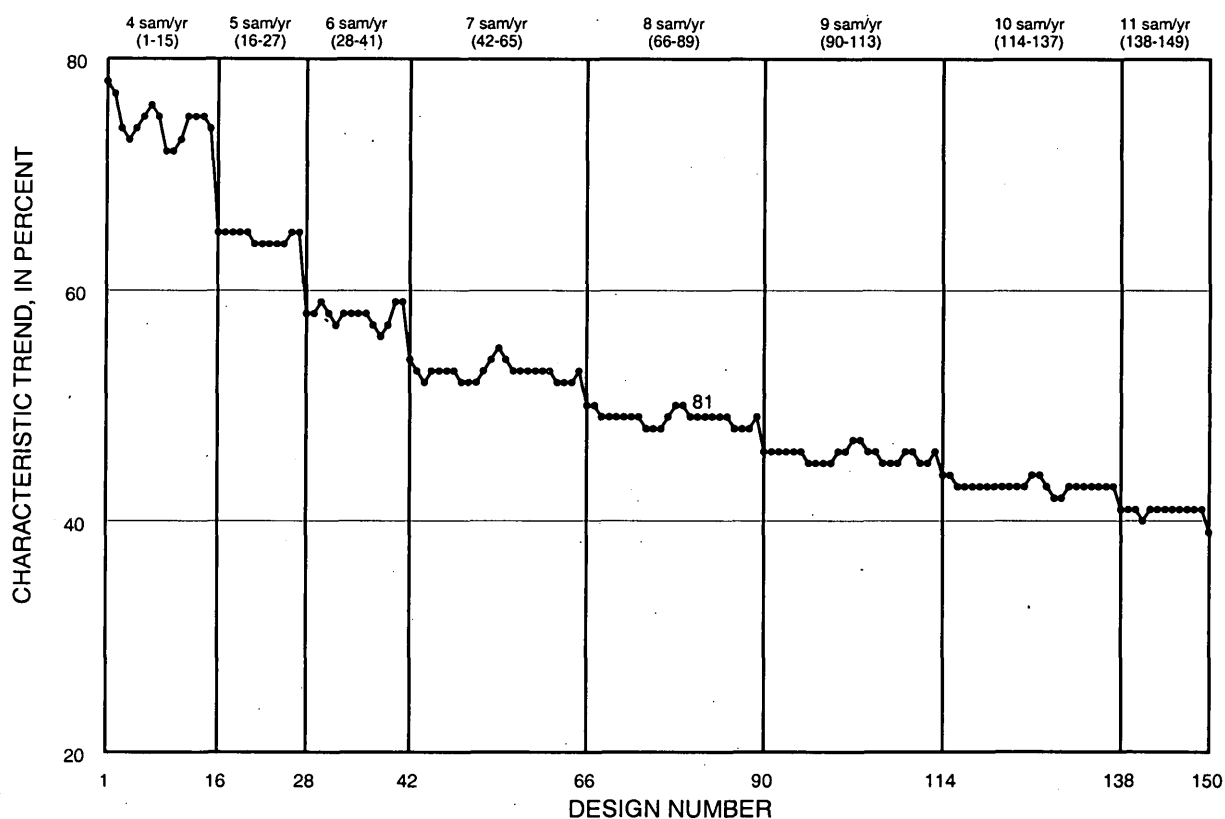


Figure 25. Magnitude of characteristic (detectable) trends in total nitrogen for sample designs based on 4 through 12 samples per year for the Connecticut River at Thompsonville, Conn.

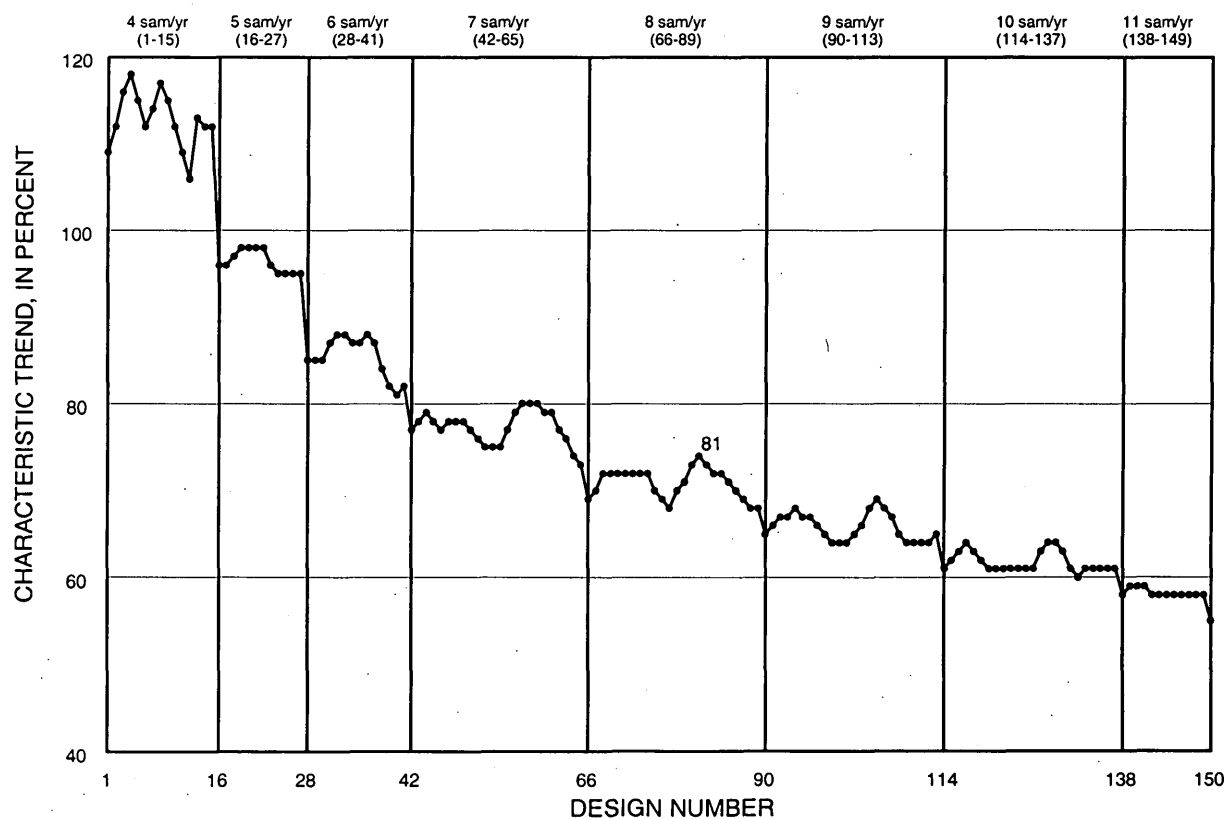


Figure 26. Magnitude of characteristic (detectable) trends in total nitrogen for sample designs based on 4 through 12 samples per year for the Salmon River at East Hampton, Conn.

Examination of characteristic trend plots for the Naugatuck River further illustrates the variation of design efficiency for some constituents within the same sampling frequency. The efficiency of 8-sample designs for total nitrogen for the Naugatuck River (fig. 27) is considerably more variable than for the Connecticut River (fig. 25). Design 81, identified as a highly efficient composite design for some stations, is one of the least efficient of the 8-sample designs for total nitrogen at the Naugatuck station. Design 67, the optimal composite 8-sample design for the Naugatuck River (fig. 22), is also one of the most efficient designs for total nitrogen (fig. 27). Likewise, the efficiency of 8-sample designs for sulfate for the Naugatuck River is highly variable (fig. 28), and design 67, the optimal composite design, is one of the best designs for sulfate. Design 81 is only moderately efficient for sulfate, and several 6-sample and 7-sample designs are more efficient for sulfate than design 81 at this station (fig. 28). Results such as these indicate that additional evaluation for specific constituents may be useful in determining sampling designs that address specific water-quality concerns.

Effects of Sampling Frequency

The smaller the characteristic trend is for a given sample size, the more efficient the design. Design efficiency varies among designs within a given sampling frequency, but also changes as sampling frequency changes. The characteristic trends of the most efficient designs (lowest points on the graph in figs. 15, 25–28) generally decrease rapidly in progressing from 4 to 8 samples per year, but decrease slowly in progressing from 8 to 12 samples per year (design 150 is monthly sampling, the right-most point in each graph). Thus, although more designs of greater efficiency are available as sampling frequency increases, the rate of improvement in design efficiency generally decreases at frequencies above 8 samples per year. For some stations and constituents, several designs with 8 samples per year are nearly as efficient as design 150 with 12 samples per year (figs. 27–28). A higher sampling frequency is not a guarantee of an increased capability for detecting trends.

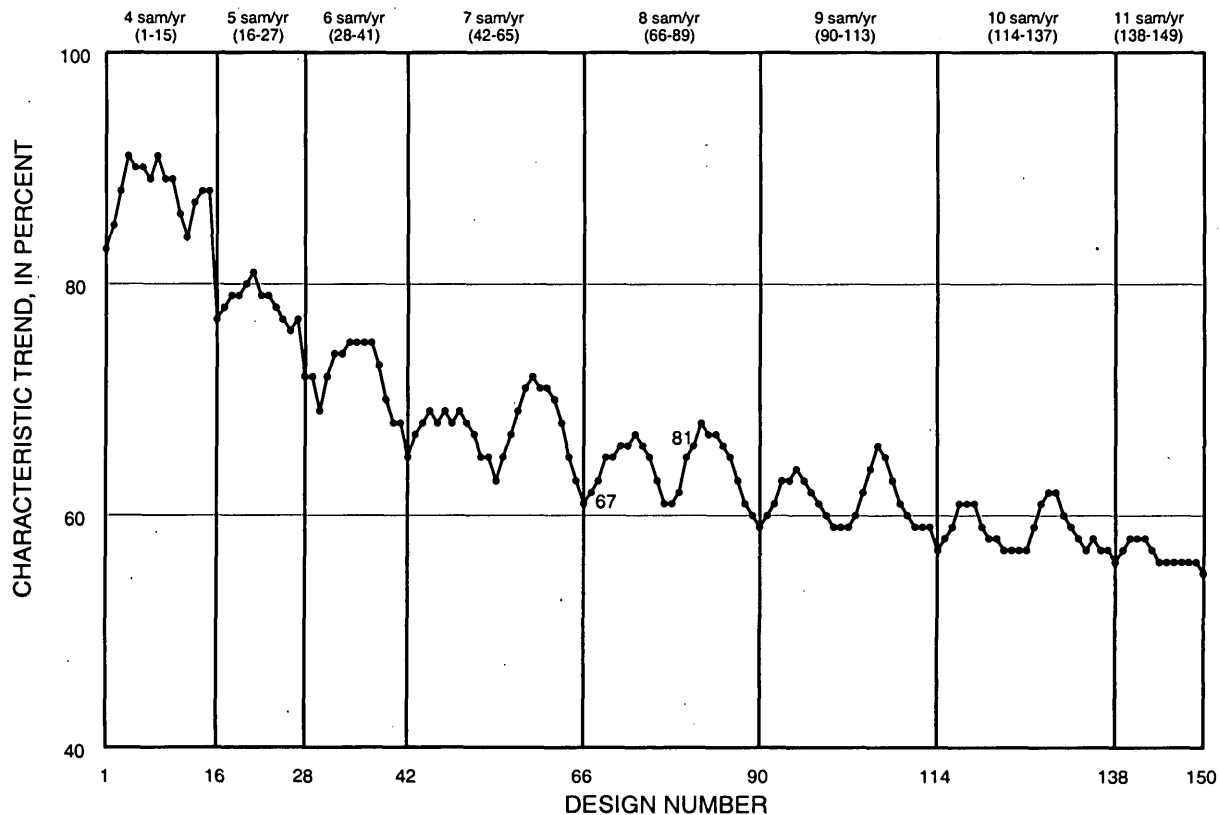


Figure 27. Magnitude of characteristic (detectable) trends in total nitrogen for sample designs based on 4 through 12 samples per year for the Naugatuck River at Beacon Falls, Conn.

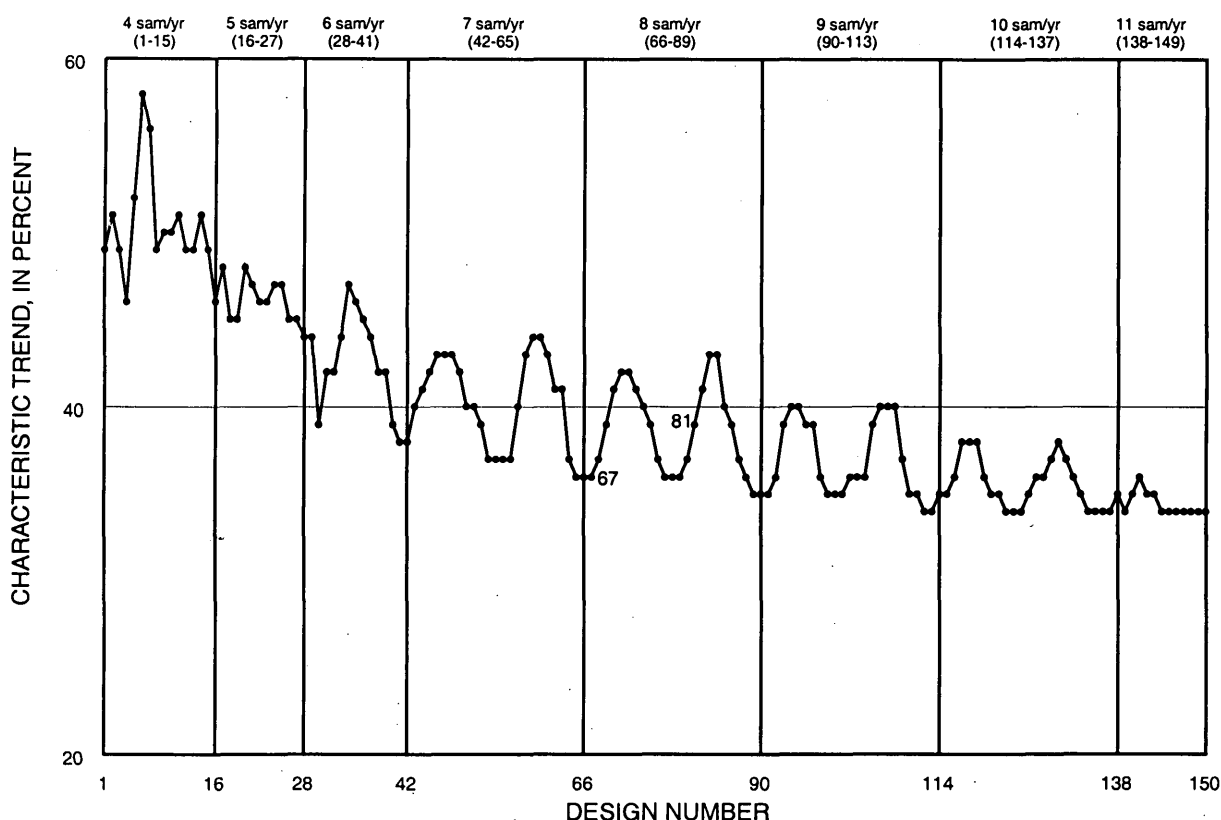


Figure 28. Magnitude of characteristic (detectable) trends in sulfate for sample designs based on 4 through 12 samples per year for the Naugatuck River at Beacon Falls, Conn.

Effects of Record Length

For each sampling frequency, the size of the characteristic trend for optimal designs decreases as the length of record increases. Characteristic trends for monthly sampling for the Quinnipiac River were plotted as a function of the number of years of water-quality record in the design (fig. 29). The size of the characteristic trend varies widely among constituents, but in all cases, it decreases as the length of the design period increases. Characteristic trends for lower sampling frequencies (not shown) are larger than those shown for monthly sampling, but there is a similar pattern of variability among constituents and a decrease in size as the length of the design period increases.

Decreases in the size of the characteristic trend tend to be largest in the change from a 5-year to a 7-year design period and smallest in the change from an 8-year to a 10-year design period. Decreases in the size of the characteristic trend over time are minimal for some constituents and substantial for others, indicating that further evaluation could provide information on the optimal length of record necessary to detect trends in specific constituents.

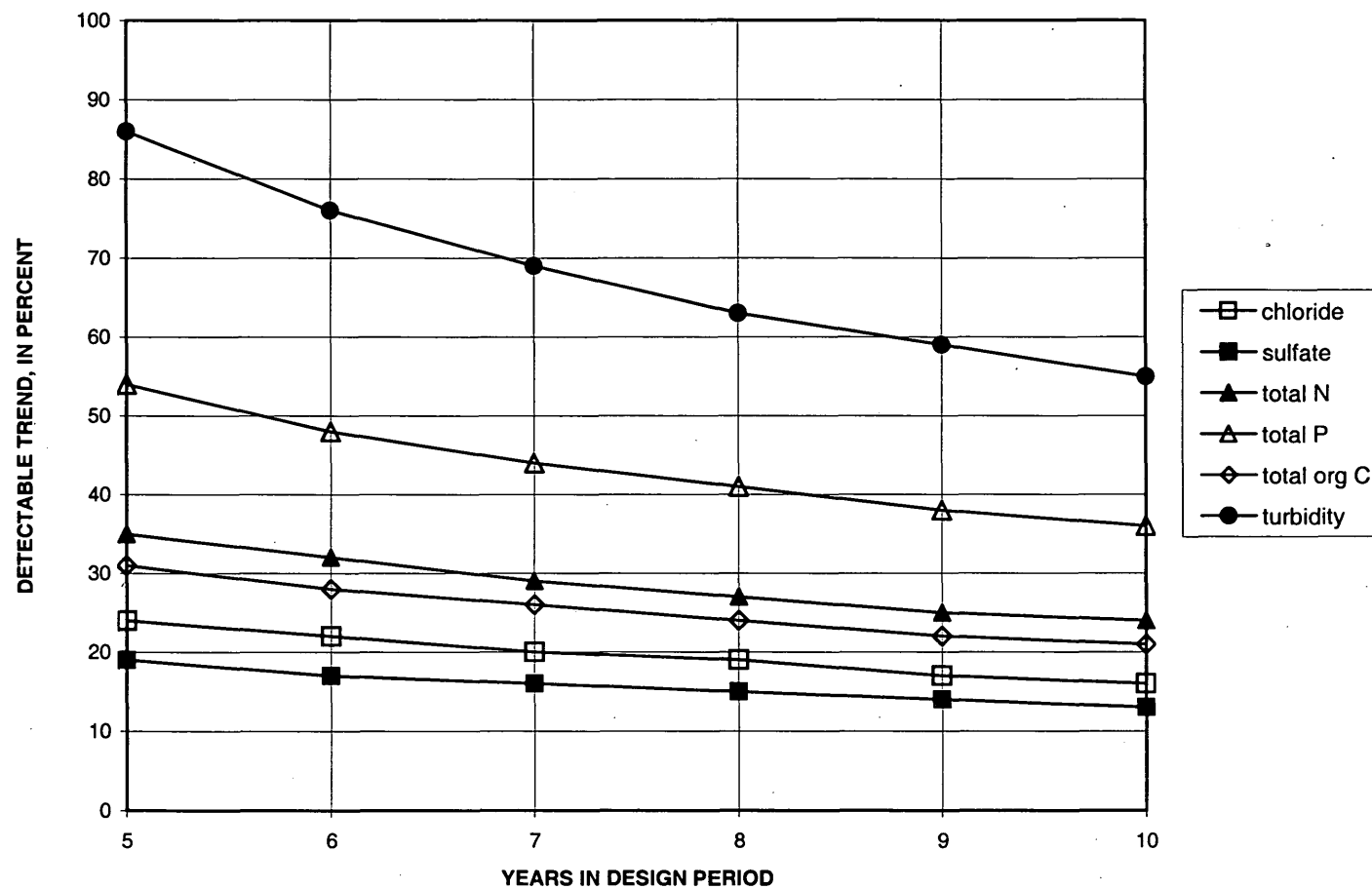


Figure 29. Magnitude of characteristic (detectable) trends in six constituents for monthly sampling, plotted as a function of the number of years in the design period for the Quinnipiac River at Wallingford, Conn.

Evaluation of Current Monitoring Program in Terms of Optimal Designs

The cooperative water-quality monitoring program between the USGS and the CTDEP included 34 monitoring stations at the time of this study (1998). During that year, one station was monitored 11 times per year; 23 stations, including the Connecticut River at Thompsonville, the Quinnipiac River at Wallingford, and the Naugatuck River at Beacon Falls, were monitored 8 times per year; 3 stations were monitored 6 times per year; and 7 stations, including the Salmon and Saugatuck Rivers, were monitored 4 times per year. At stations where the sampling frequency is 8 times per year, samples are collected monthly during summer (generally May to September or June to October) and bimonthly in winter. The selection of months for bimonthly sampling in winter, and the beginning and ending months of the monthly sampling period, is currently a logistical decision, based on station location and other factors affecting field operations.

Results from the design analysis indicate that monthly sampling during the growing season will meet general data requirements for long-term trend analysis at some stations, including the Connecticut and Quinnipiac, but may not be well-suited for the detection of trends in all situations. Monthly sampling in the winter and spring months yields more information for trend analysis on the Naugatuck, Salmon, and Saugatuck Rivers. This difference is due largely to the effects of one constituent, sulfate, on the design requirements for these three stations. Trend detection for other constituents, including total nitrogen, also may benefit from additional winter sampling, as indicated by examination of optimal designs for nitrogen for the Connecticut, Salmon, and Saugatuck Rivers.

A general characteristic of the designs, and a general concept of design evaluation, is "diminishing incremental benefits" as the number of samples, and therefore costs, increase. The power of the designs for the Connecticut River increases fairly rapidly in progressing from 4 to 6 samples per year, then increases more slowly in progressing from 6 to 9 samples per year (fig. 18). Diminishing incremental benefits is a property of any analysis of design efficiency, because samples with the highest incremental benefit (for example, the highest increase in power) are included first, followed by samples with lower incremental benefit. As samples are added, the incremental

benefit of an additional sample eventually becomes less than the cost of the sample, and further increases in the sample size are not warranted. Although the cost of each additional sample should be easy to determine, a rigorous definition of the incremental benefit (in dollars) is more difficult to determine. Potential benefits of increasing sample size, such as reducing the size of trend that can be detected for a fixed time interval or decreasing the time it takes to detect a trend of a given magnitude, may outweigh the increased cost.

Rigorous cost-benefit analysis of the designs is beyond the scope of this study; therefore, power is used as a surrogate for benefits. It is reasonable to assume that the overall benefit of a design is proportional to the power for detecting a trend. Criteria for acceptable designs can be determined relative to the power and cost of monthly sampling. For example, suppose that a design is deemed acceptable if, for every constituent, it provides at least 75 percent of the power of monthly sampling at less than 70 percent of the cost. With this cost criterion, all acceptable designs will have 8 or fewer samples per year. As indicated in figure 18 for the Connecticut River, designs 58 (with 7 samples per year) and 81 (with 8 samples per year) are both acceptable designs based on these criteria. The power for all constituents in both designs is greater than 0.6 (75 percent of 0.8, the power for monthly sampling). The cost of design 58 is assumed to be 58 percent (7/12) of monthly sampling, and the cost of design 81 is 67 percent (8/12) of monthly sampling. None of the other designs meets these criteria.

Using the above criteria for an acceptable design, design 89 (8 samples per year) is the only acceptable design of those shown for the Salmon River (fig. 19). For the Saugatuck River (fig. 20), design 78 (8 samples) is the only acceptable design. For the Quinnipiac River (fig. 21), design 81 (8 samples) is the only acceptable design. For the Naugatuck River (fig. 22), designs 56 (7 samples) and 67 (8 samples) are both acceptable designs.

The greatest difference between the optimal designs presented here and the existing monitoring program is in the current monitoring schedule for the Salmon and Saugatuck Rivers, both of which are monitored quarterly. Quarterly sampling designs for these stations have very low power for trend detection for all constituents evaluated in this study, and all designs with less than 8 samples per year have low power for trend detection for constituents other than chloride or sulfate (figs. 19 and 20).

SUMMARY AND CONCLUSIONS

Five water-quality stations, part of a larger network of 34 stations throughout Connecticut, were selected for trend analysis. The five stations all have more than 20 years of water-quality record, and their drainage areas represent the major land uses and hydrologic conditions in Connecticut. The five drainage basins range in size from about 20 mi² to almost 10,000 mi². Time-series analysis, a mathematically complex parametric approach, was used to analyze flow-adjusted trends in concentration and evaluate the efficiency of various sampling designs for monitoring trends in water quality.

Numerous trends were detected in the data for the five stations, which represent water quality in the Connecticut River, the largest river in New England; the Quinnipiac and Naugatuck Rivers, which have large urban areas and receive substantial point discharges; and the Salmon and Saugatuck Rivers, which are largely forested and receive no point discharges, but which have undergone increasing suburban development during the period of the study.

Long-term or short-term trends were detected in the records of all constituents at all stations evaluated in this study. Improvement in water quality is indicated by downward trends in sulfate concentrations at all stations and in recent decreases in total nitrogen and long-term decreases in total phosphorus concentrations at some stations. Deterioration in water quality is indicated by long-term upward trends in chloride concentration at all five stations.

Chloride concentrations generally increased in all five streams during the period of the study. Upward chloride trends in the two forested basins with no point sources indicate that nonpoint sources may be a factor affecting chloride trends in suburban areas.

Significant decreases in dissolved sulfate concentrations were detected in the data for all five streams during one or more time periods within the period of record. Typical flow-adjusted concentrations of dissolved sulfate were lower at the end of the period of record than at the beginning in all five streams. Declines in sulfate at all five stations, in some cases spanning three decades, suggest a regional cause. The relation of trends in streamwater quality in Connecticut to trends in regional air quality is a potential area for further investigation.

Flow-adjusted concentrations of total nitrogen in the Connecticut, Naugatuck, Salmon, and Saugatuck Rivers increased during variable periods from the mid-1970's to the mid- or late 1980's, and then decreased to a time in the early 1990's. On the Naugatuck, Salmon, and Saugatuck Rivers, concentrations remained relatively constant during the mid- to late 1990's. Total nitrogen in the Connecticut River declined slightly during the 1990's. Several significant short-term increases and decreases in total nitrogen concentration were detected for the Quinnipiac River, but no overall trend was detected for the period of record, 1972–98. General trend patterns on four of the five streams are similar, despite differences in land use and the presence or absence of point discharges to the streams. Investigation of changes in atmospheric emission of nitrogen oxides and atmospheric deposition of nitrogen could yield information on possible causes of trends in total nitrogen in streams.

Trend models for total phosphorus revealed distinctly different patterns over time among the five streams. The most pronounced change in total phosphorus concentrations took place on the Connecticut River. Overall, there is a highly significant downward trend for the period of the study, with typical concentrations in the 1990's substantially lower than concentrations in the 1970's. Data for the Salmon and Saugatuck Rivers show a distinct cyclical pattern of increases and decreases during similar time periods on an approximately decadal scale. Concentrations of total phosphorus declined with some variability on the Quinnipiac and Naugatuck Rivers from the mid-1970's to the mid-1990's. In both streams, a significant upward trend was detected from the mid-1990's to 1998.

Concentrations of total organic carbon on the Naugatuck River decreased substantially during the 1970's and early 1980's and fluctuated substantially in the 1980's and 1990's. With the exception of the Naugatuck River, no large overall trends in total organic carbon were detected during the 1980's and 1990's. Positive step trends, possibly related to a method change, were detected for the Salmon, Saugatuck, and Quinnipiac Rivers for a period during the 1970's into the early 1980's.

Trend patterns for turbidity share similarities at all five stations. A statistically significant positive step trend is present for 1978–82 in the data for all five stations. The appearance of the data plots (shown in the

appendix) suggests that a change in method, instrument, or method sensitivity accounts for this step change in 1982. All five stations display a sawtooth pattern of decreases and increases in turbidity over the period of record.

The statistical time-series model of historical discharge and concentration data was used to evaluate the efficiency of various sampling designs for monitoring trends in water quality. Optimal sampling designs for the Connecticut, Salmon, Saugatuck, Quinnipiac, and Naugatuck Rivers were identified for each of six sampling frequencies ranging from 4 through 9 samples per year, based on a design period of 5 years.

The cooperative water-quality monitoring program between the USGS and the CTDEP currently includes 34 monitoring stations, with sampling frequencies of 4, 6, 8, and 11 times per year at different stations. The Connecticut, Quinnipiac, and Naugatuck Rivers are monitored 8 times per year, and the Salmon and Saugatuck Rivers are monitored on a quarterly basis. All stations had a higher sampling frequency earlier in the period of record.

Results from the design analysis indicate that monthly sampling from approximately April to September (with two winter samples) will meet general data requirements for long-term trend analysis at some stations, including the Connecticut and Quinnipiac, but may not be well-suited for the detection of trends in all situations. Monthly sampling in the winter and spring months yields optimal information for trend analysis on the Naugatuck, Salmon, and Saugatuck Rivers. This difference is due largely to the effects of one constituent, sulfate, on the design requirements for these three stations. Trend detection for other constituents, however, including total nitrogen, also benefits from additional winter sampling, as indicated by examination of optimal designs for total nitrogen for the Connecticut, Salmon, and Saugatuck Rivers.

Sampling designs with reasonable power to detect trends in the constituents evaluated can be identified at the sampling frequency of 8 samples per year; however, the timing of samples for the 8-sample designs differs among the stations. These results indicate that improved information for trend detection can be obtained by using optimal design information in planning the monitoring program.

The greatest difference between the optimal designs presented here and the existing monitoring program is in the current quarterly monitoring schedule

for the Salmon and Saugatuck Rivers. Quarterly sampling designs for these two rivers have very low power for trend detection for all constituents evaluated in this study, and all designs with less than 8 samples per year have low power for trend detection for constituents other than chloride or sulfate.

The high frequency of trend detection in this study, and the high significance of many of the trends detected, indicate the variability and complexity of water-quality conditions in Connecticut. Evaluation of optimal sampling designs provides an approach for improving the efficiency of the monitoring program for detecting water-quality trends in the future.

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APPENDIXES

Appendix 1. Summary of trend results for selected water-quality stations, 1968–98

[L, linear trend; S, step trend; p-value, attained significance level of the trend test; model coefficient is the slope of linear trend or size of step trend, in base-10 logarithmic units; for linear trends, trend is percent change in the median per year; for step trends, trend is percent change in the median from before to after the time of the step trend]

Water-quality station name	Type of trend	Trend period	Test statistic	P-value	Model coefficient	Trend, in percent
CHLORIDE						
Connecticut River at Thompsonville, Conn.	L	1968-73	-3.025	0.0012	-0.010	-2.3
	L	1974-98	9.516	<0.0001	.005	1.2
Salmon River near East Hampton, Conn.	L	1972-74	-6.619	<0.0001	-.031	-6.9
	L	1975-77	7.292	<0.0001	.043	10
	L	1978-80	-4.990	<0.0001	-.029	-6.5
	L	1981-84	4.456	<0.0001	.019	4.5
	L	1985-89	2.492	0.0064	.006	1.4
Quinnipiac River at Wallingford, Conn.	L	1968-71	3.783	0.0001	0.031	7.4
	L	1972-76	-4.842	<0.0001	-0.014	-3.2
	L	1984-95	16.204	<0.0001	0.017	4.0
	L	1996-98	-3.114	0.0009	-0.029	-6.5
Naugatuck River at Beacon Falls, Conn.	L	1974-79	2.874	0.0020	.011	2.6
	L	1980-88	8.808	<0.0001	.018	4.2
	L	1989-98	-2.219	0.0132	-.005	-1.1
Saugatuck River near Redding, Conn.	L	1968-70	3.417	0.0003	.060	15
	L	1971-73	-5.028	<0.0001	-.050	-11
	L	1974-76	4.840	<0.0001	.034	8.1
	L	1981-91	15.415	<0.0001	.019	4.5
SULFATE						
Connecticut River at Thompsonville, Conn.	S	1983-89	3.035	0.0012	.031	7.4
	L	1968-72	-2.036	0.0209	-.008	-1.8
	L	1973-76	-5.524	<0.0001	-.022	-4.9
	L	1977-81	1.694	0.0451	.005	1.2
	L	1988-98	-9.887	<0.0001	-.014	-3.2
Salmon River near East Hampton, Conn.	S	1987-89	8.210	<0.0001	.078	20
	L	1972-75	-5.693	<0.0001	-.017	-3.8
	L	1976-79	2.690	0.0036	.007	1.6
	L	1987-94	-10.873	<0.0001	-.013	-2.9
Quinnipiac River at Wallingford, Conn.	S	1983-89	6.568	<0.0001	.065	16
	L	1968-75	-5.353	<0.0001	-.010	-2.3
	L	1981-84	-12.158	<0.0001	-.033	-7.3
	L	1992-98	-2.899	0.0019	-.008	-1.8
Naugatuck River at Beacon Falls, Conn.	L	1974-77	-6.494	<0.0001	-.053	-11
	L	1978-81	3.904	<0.0001	.024	5.7
	L	1989-94	-5.279	<0.0001	-.013	-2.9

Appendix 1. Summary of trend results for selected water-quality stations, 1968-98—Continued

[L, linear trend; S, step trend; p-value, attained significance level of the trend test; model coefficient is the slope of linear trend or size of step trend, in base-10 logarithmic units; for linear trends, trend is percent change in the median per year; for step trends, trend is percent change in the median from before to after the time of the step trend]

Water-quality station name	Type of trend	Trend period	Test statistic	P-value	Model coefficient	Trend, in percent
Saugatuck River near Redding, Conn.	S	1983-89	2.806	0.0025	.052	13
	L	1973-76	-9.711	<0.0001	-.038	-8.4
	L	1977-89	7.827	<0.0001	.036	8.6
	L	1981-85	-5.351	<0.0001	-.034	-7.5
	L	1986-88	3.741	0.0001	.027	6.4
	L	1980-92	-4.480	<0.0001	-.027	-6.0
TOTAL NITROGEN						
Connecticut River at Thompsonville, Conn.	S	1986-91	2.004	0.0225	.043	10
	L	1971-75	-4.577	<0.0001	-.057	-12
	L	1976-82	3.561	0.0002	.016	3.8
	L	1988-90	-5.352	<0.0001	-.047	-10
	L	1991-98	-2.044	0.0205	-.012	-2.7
Salmon River near East Hampton, Conn.	S	1986-91	3.348	0.0004	.095	24
	L	1972-75	-1.721	0.0426	-.041	-9.0
	L	1976-84	4.833	<0.0001	.024	5.7
	L	1985-92	-8.663	<0.0001	-.039	-8.6
Quinnipiac River at Wallingford, Conn.	S	1986-91	2.656	0.0040	.073	18
	L	1972-75	-3.122	0.0009	-.030	-6.7
	L	1976-81	2.395	0.0083	.010	2.3
	L	1982-88	-2.692	0.0036	-.016	-3.6
	L	1989-95	3.592	0.0002	.022	5.2
	L	1996-98	-2.245	0.0124	-.032	-7.1
Naugatuck River at Beacon Falls, Conn.	L	1980-87	2.389	0.0084	.008	1.9
	L	1988-92	-6.745	<0.0001	-.038	-8.4
Saugatuck River near Redding, Conn.	S	1986-91	4.065	<0.0001	.125	33
	L	1979-84	4.776	<0.0001	.031	7.4
	L	1985-88	-7.533		-.073	-15
TOTAL PHOSPHORUS						
Connecticut River at Thompsonville, Conn.	L	1969-98	-11.751	<0.0001	-.018	-4.1
Salmon River near East Hampton, Conn.	L	1971-76	3.080	0.0010	.065	16
	L	1977-79	-3.288	0.0005	-.104	-21
	L	1980-82	4.509	<0.0001	.152	42
	L	1983-86	-4.824	<0.0001	-.113	-23
	L	1987-91	2.703	0.0034	.048	12
	L	1992-98	-2.580	0.0049	-.050	-11
Quinnipiac River at Wallingford, Conn.	L	1979-82	-8.232	<0.0001	-.042	-9.2
	L	1988-93	-4.521	<0.0001	-.022	-4.9
	L	1994-98	3.540	0.0002	.034	8.1

Appendix 1. Summary of trend results for selected water-quality stations, 1968–98—Continued

[L, linear trend; S, step trend; p-value, attained significance level of the trend test; model coefficient is the slope of linear trend or size of step trend, in base-10 logarithmic units; for linear trends, trend is percent change in the median per year; for step trends, trend is percent change in the median from before to after the time of the step trend]

Water-quality station name	Type of trend	Trend period	Test statistic	P-value	Model coefficient	Trend, in percent
Naugatuck River at Beacon Falls, Conn.	L	1974-77	-2.690	0.0036	-.059	-13
	L	1978-81	2.579	0.0050	.033	7.9
	L	1982-94	-4.147	<0.0001	-.015	-3.4
	L	1995-98	4.045	<0.0001	.079	20
Saugatuck River near Redding, Conn.	L	1971-75	4.102	<0.0001	.090	23
	L	1976-78	-3.369	0.0004	-.084	-18
	L	1979-83	3.460	0.0003	.050	12
	L	1984-87	-3.787	0.0001	-.067	-14
	L	1988-91	2.129	0.0166	.039	9.4
	L	1992-98	-2.367	0.0090	-.038	-8.4
TOTAL ORGANIC CARBON						
Connecticut River at Thompsonville, Conn.	L	1972-74	2.878	0.0020	.087	22
	L	1975-77	2.969	0.0015	.040	9.6
	L	1978-81	-7.973	<0.0001	-.062	-13
	L	1982-88	2.311	0.0104	.008	1.9
Salmon River near East Hampton, Conn.	S	1974-79	9.773	<0.0001	.255	80
	L	1974-78	4.675	<0.0001	.038	9.1
	L	1984-86	3.322	0.0004	.029	6.9
	L	1987-98	-1.731	0.0417	-.006	-1.4
Quinnipiac River at Wallingford, Conn.	S	1974-81	4.042	<0.0001	.111	29
	L	1974-77	2.711	0.0034	.030	7.2
	L	1978-81	-4.096	<0.0001	-.044	-9.6
Naugatuck River at Beacon Falls, Conn.	L	1974-76	-4.158	<0.0001	-.084	-18
	L	1979-83	-5.129	<0.0001	-.033	-7.3
	L	1984-86	2.396	0.0083	.031	7.4
	L	1987-90	-2.458	0.0070	-.024	-5.4
	L	1991-94	2.193	0.0142	.022	5.2
	L	1995-98	-1.723	0.0424	-.024	-5.4
Saugatuck River near Redding, Conn.	S	1974-81	2.218	0.0133	.084	21
	L	1974-78	4.078	<0.0001	.036	8.6
	L	1979-81	-3.146	0.0008	-.051	-11
	L	1982-86	2.525	0.0058	.018	4.2
	L	1987-92	-2.086	0.0185	-.010	-2.3
TURBIDITY						
Connecticut River at Thompsonville, Conn.	S	1978-82	3.825	0.0001	.354	126
	L	1978-83	-2.152	0.0157	-.049	-11
	L	1984-88	4.370	<0.0001	.063	16
	L	1989-92	-1.763	0.0390	-.033	-7.3
	L	1993-98	1.855	0.0318	.029	6.9

Appendix 1. Summary of trend results for selected water-quality stations, 1968–98—Continued

[L, linear trend; S, step trend; p-value, attained significance level of the trend test; model coefficient is the slope of linear trend or size of step trend, in base-10 logarithmic units; for linear trends, trend is percent change in the median per year; for step trends, trend is percent change in the median from before to after the time of the step trend]

Water-quality station name	Type of trend	Trend period	Test statistic	P-value	Model coefficient	Trend, in percent
Salmon River near East Hampton, Conn.	S	1978-82	3.352	0.0004	.211	63
	L	1978-83	-1.878	0.0302	-.030	-6.7
	L	1984-92	3.436	0.0003	.019	4.5
	L	1993-95	-1.841	0.0328	-.044	-9.6
	L	1996-98	2.305	0.0106	.110	29
Quinnipiac River at Wallingford, Conn.	S	1978-82	7.926	<0.0001	.373	136
	L	1985-89	3.428	0.0003	.038	9.1
	L	1990-94	-3.095	0.0010	-.044	-9.6
	L	1995-98	4.606	<0.0001	.129	35
Naugatuck River at Beacon Falls, Conn.	S	1978-82	2.344	0.0095	.213	63
	L	1978-87	-2.214	0.0134	-.034	-7.5
	L	1988-90	3.214	0.0007	.079	20
	L	1991-94	-2.224	0.0131	-.051	-11
	L	1995-98	2.419	0.0078	.078	20
Saugatuck River near Redding, Conn.	S	1978-82	1.677	0.0468	.115	30
	L	1978-87	-2.112	0.0173	-.024	-5.4
	L	1988-90	3.529	0.0002	.069	17
	L	1991-94	-2.279	0.0111	-.041	-9.0
	L	1995-98	2.924	0.0017	.090	23

Appendix 2. Selected trend models of water-quality data for streams in Connecticut by station and constituent

The water-quality data and selected time-series model output used in this report are shown in this appendix.

The graphs in each of the figures are interpreted as follows:

Upper-left graph: Recorded data, Y , and fitted trend $C + T$ (see eq. 2);

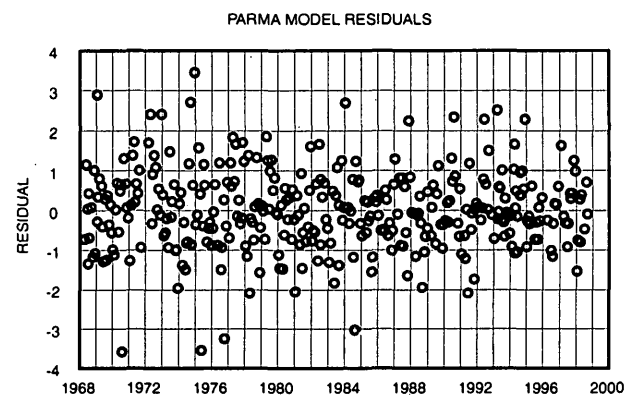
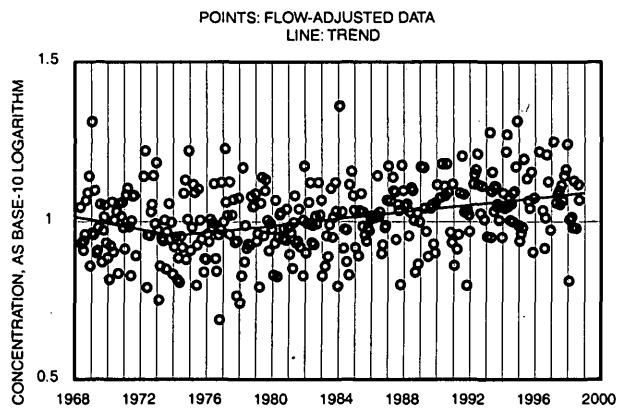
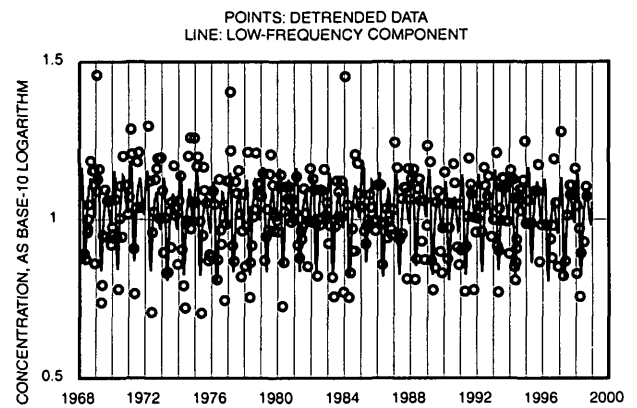
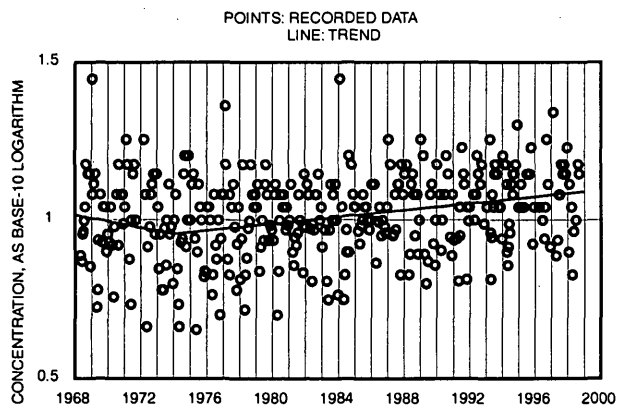
Upper-right graph: Detrended data, $Y - T$, and fitted low-frequency component, $C + A + S$ (see eq. 2);

Lower-left graph: Flow-adjusted data, $Y - A - S$, and fitted trend, $C + T$ (see eq. 2). These graphs correspond to figures 9 through 14 in the report.

Lower-right graph: PARMA model residuals (see Vecchia, 2000, Appendix A).

Connecticut River at Thompsonville, Connecticut

DISSOLVED CHLORIDE, IN MILLIGRAMS PER LITER (AS CL)



Appendix 2. Selected trend models of water-quality data for streams in Connecticut by station and constituent—Continued

The water-quality data and selected time-series model output used in this report are shown in this appendix.

The graphs in each of the figures are interpreted as follows:

Upper-left graph: Recorded data, Y , and fitted trend $C + T$ (see eq. 2);

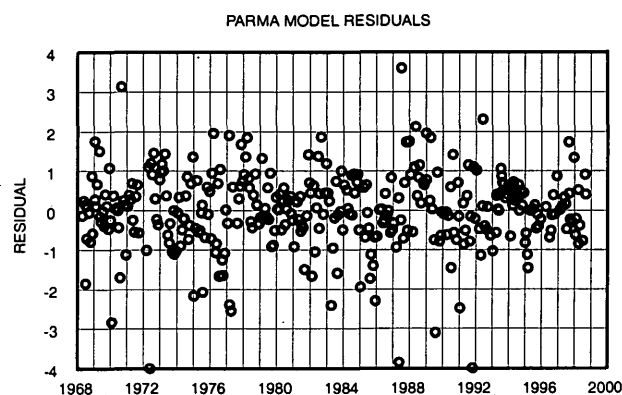
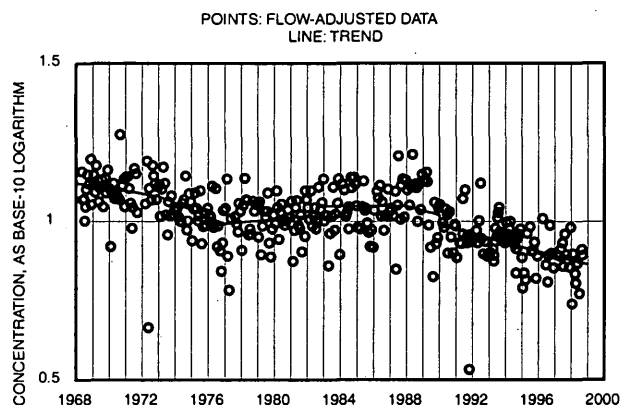
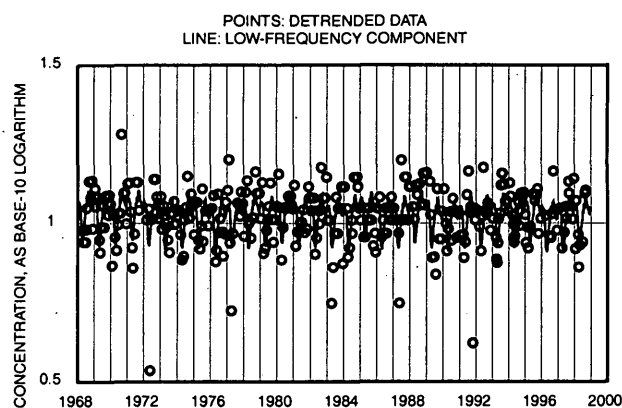
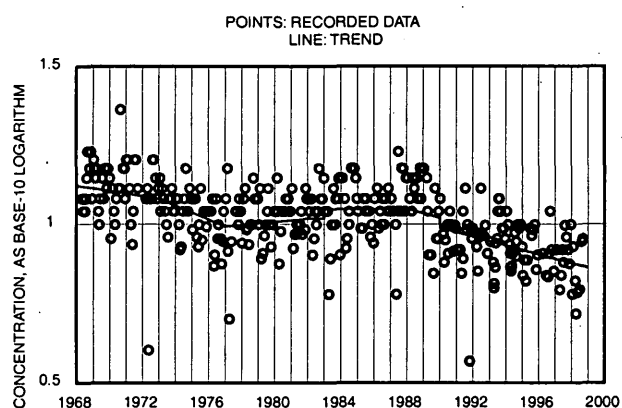
Upper-right graph: Detrended data, $Y - T$, and fitted low-frequency component, $C + A + S$ (see eq. 2);

Lower-left graph: Flow-adjusted data, $Y - A - S$, and fitted trend, $C + T$ (see eq. 2). These graphs correspond to figures 9 through 14 in the report.

Lower-right graph: PARMA model residuals (see Vecchia, 2000, Appendix A).

Connecticut River at Thompsonville, Connecticut

DISSOLVED SULFATE, IN MILLIGRAMS PER LITER (AS SO₄)



Appendix 2. Selected trend models of water-quality data for streams in Connecticut by station and constituent—Continued

The water-quality data and selected time-series model output used in this report are shown in this appendix.

The graphs in each of the figures are interpreted as follows:

Upper-left graph: Recorded data, Y , and fitted trend $C + T$ (see eq. 2);

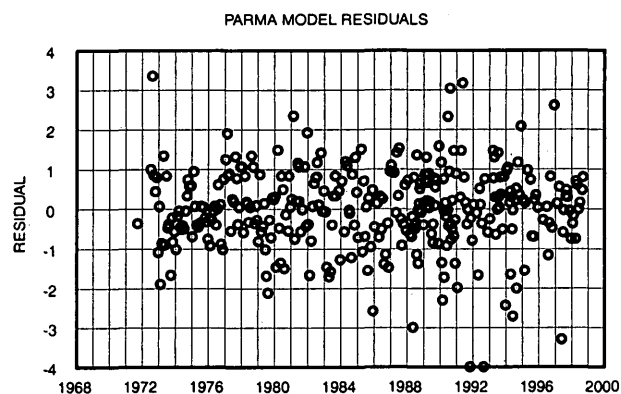
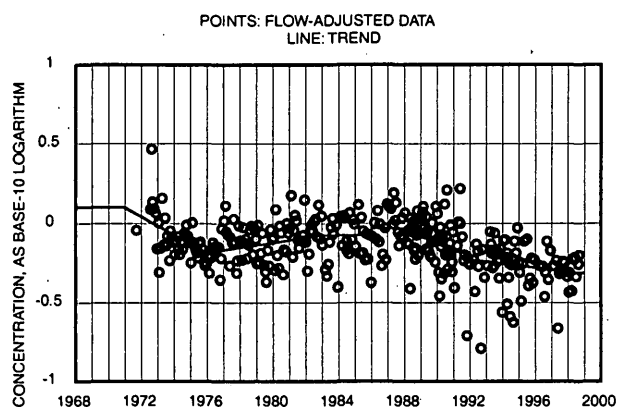
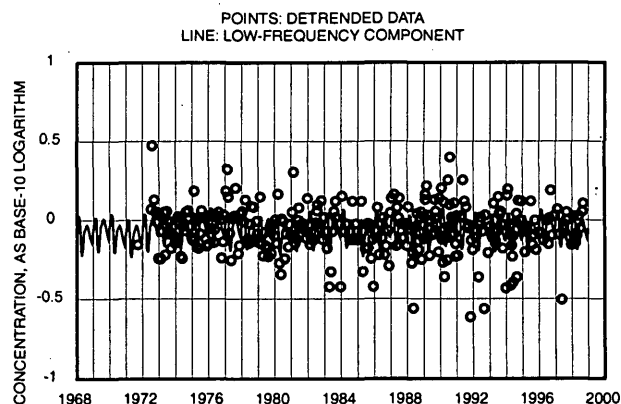
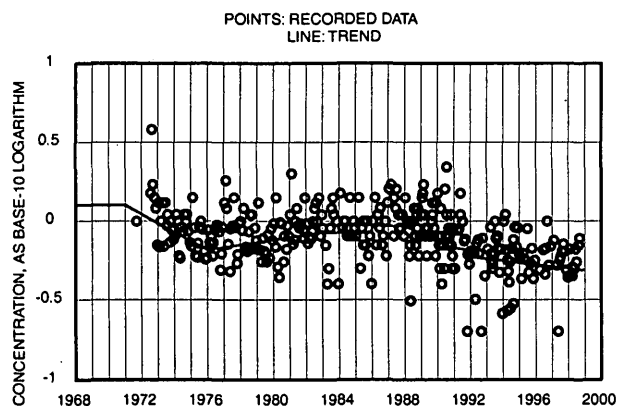
Upper-right graph: Detrended data, $Y - T$, and fitted low-frequency component, $C + A + S$ (see eq. 2);

Lower-left graph: Flow-adjusted data, $Y - A - S$, and fitted trend, $C + T$ (see eq. 2). These graphs correspond to figures 9 through 14 in the report.

Lower-right graph: PARMA model residuals (see Vecchia, 2000, Appendix A).

Connecticut River at Thompsonville, Connecticut

TOTAL NITROGEN, IN MILLIGRAMS PER LITER (AS N)



Appendix 2. Selected trend models of water-quality data for streams in Connecticut by station and constituent—Continued

The water-quality data and selected time-series model output used in this report are shown in this appendix.

The graphs in each of the figures are interpreted as follows:

Upper-left graph: Recorded data, Y , and fitted trend $C + T$ (see eq. 2);

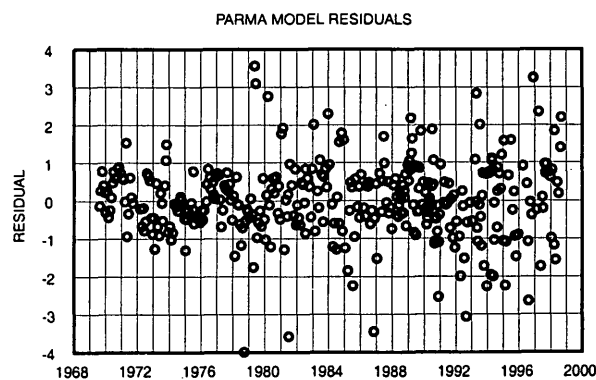
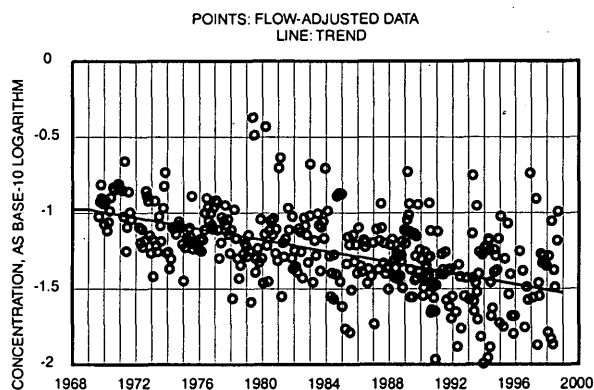
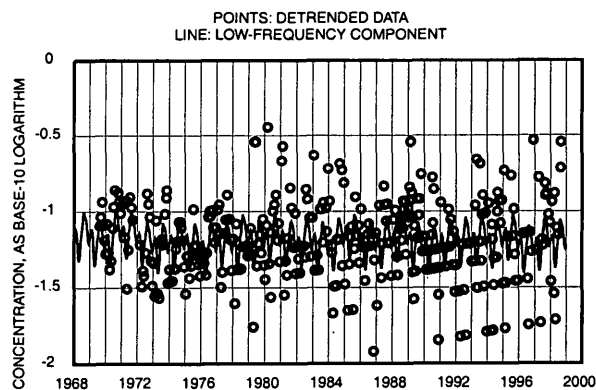
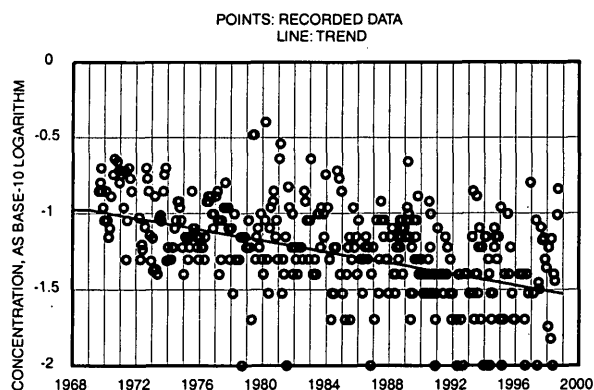
Upper-right graph: Detrended data, $Y - T$, and fitted low-frequency component, $C + A + S$ (see eq. 2);

Lower-left graph: Flow-adjusted data, $Y - A - S$, and fitted trend, $C + T$ (see eq. 2). These graphs correspond to figures 9 through 14 in the report.

Lower-right graph: PARMA model residuals (see Vecchia, 2000, Appendix A).

Connecticut River at Thompsonville, Connecticut

TOTAL PHOSPHORUS, IN MILLIGRAMS PER LITER (AS P)



Appendix 2. Selected trend models of water-quality data for streams in Connecticut by station and constituent—Continued

The water-quality data and selected time-series model output used in this report are shown in this appendix.

The graphs in each of the figures are interpreted as follows:

Upper-left graph: Recorded data, Y , and fitted trend $C + T$ (see eq. 2);

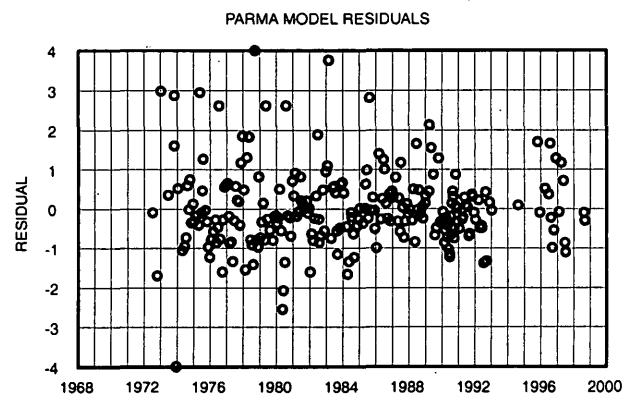
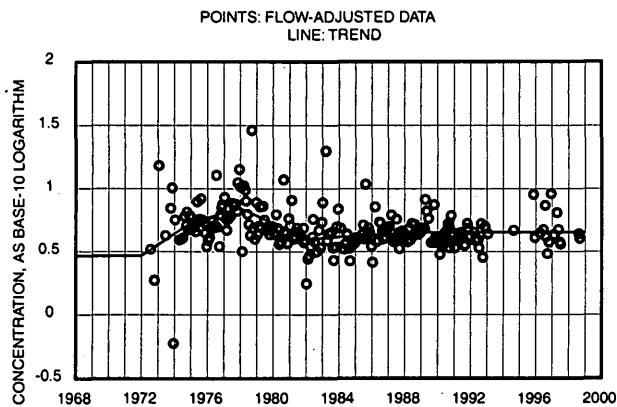
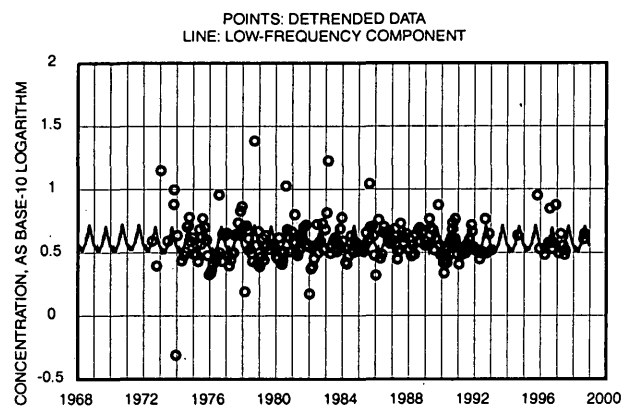
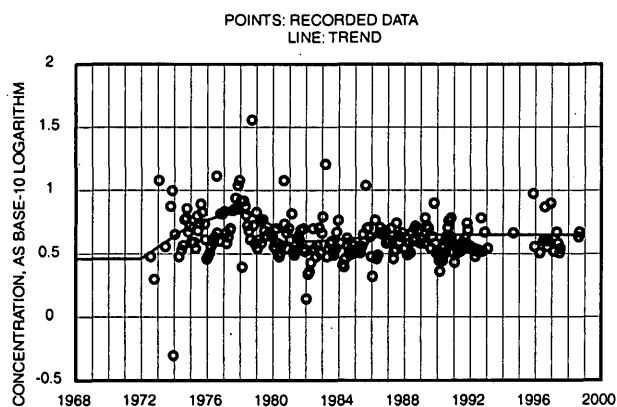
Upper-right graph: Detrended data, $Y - T$, and fitted low-frequency component, $C + A + S$ (see eq. 2);

Lower-left graph: Flow-adjusted data, $Y - A - S$, and fitted trend, $C + T$ (see eq. 2). These graphs correspond to figures 9 through 14 in the report.

Lower-right graph: PARMA model residuals (see Vecchia, 2000, Appendix A).

Connecticut River at Thompsonville, Connecticut

TOTAL ORGANIC CARBON, IN MILLIGRAMS PER LITER (AS C)



Appendix 2. Selected trend models of water-quality data for streams in Connecticut by station and constituent—Continued

The water-quality data and selected time-series model output used in this report are shown in this appendix.

The graphs in each of the figures are interpreted as follows:

Upper-left graph: Recorded data, Y , and fitted trend $C + T$ (see eq. 2);

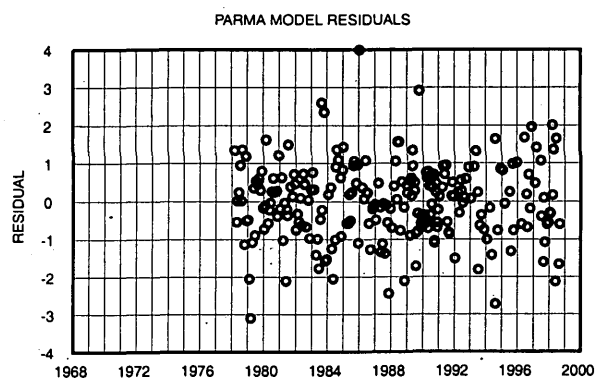
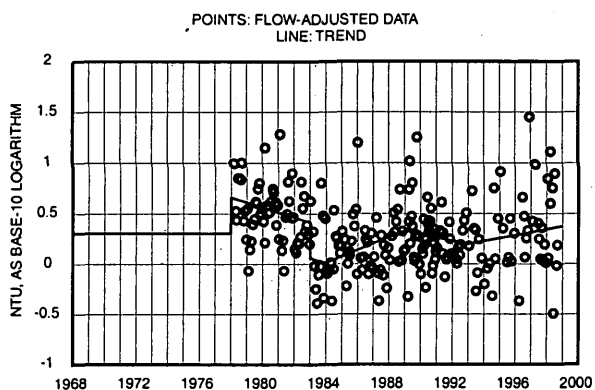
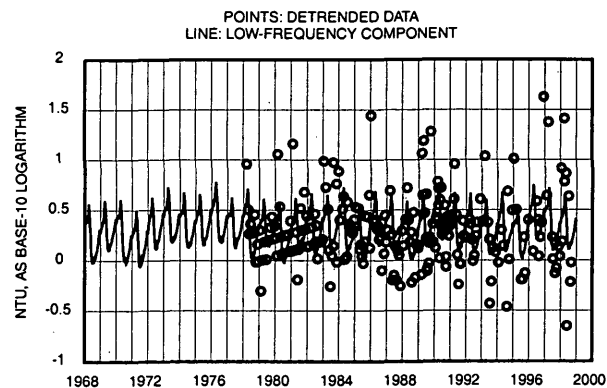
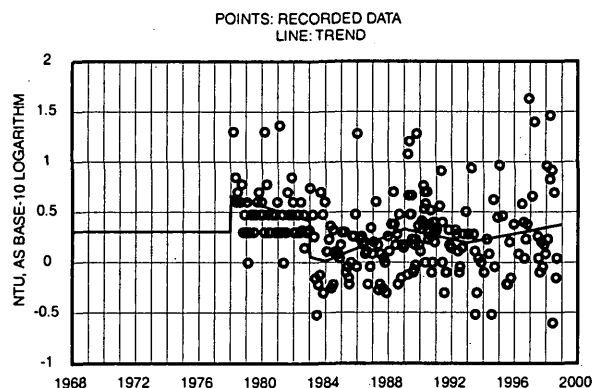
Upper-right graph: Detrended data, $Y - T$, and fitted low-frequency component, $C + A + S$ (see eq. 2);

Lower-left graph: Flow-adjusted data, $Y - A - S$, and fitted trend, $C + T$ (see eq. 2). These graphs correspond to figures 9 through 14 in the report.

Lower-right graph: PARMA model residuals (see Vecchia, 2000, Appendix A).

Connecticut River at Thompsonville, Connecticut

TURBIDITY, IN NEPHELOMETRIC TURBIDITY UNITS



Appendix 2. Selected trend models of water-quality data for streams in Connecticut by station and constituent—Continued

The water-quality data and selected time-series model output used in this report are shown in this appendix.

The graphs in each of the figures are interpreted as follows:

Upper-left graph: Recorded data, Y , and fitted trend $C + T$ (see eq. 2);

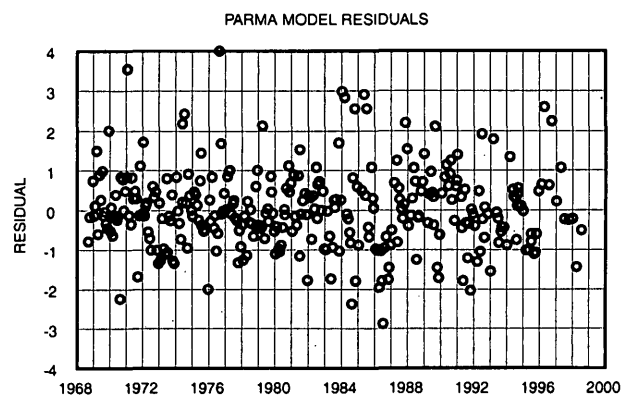
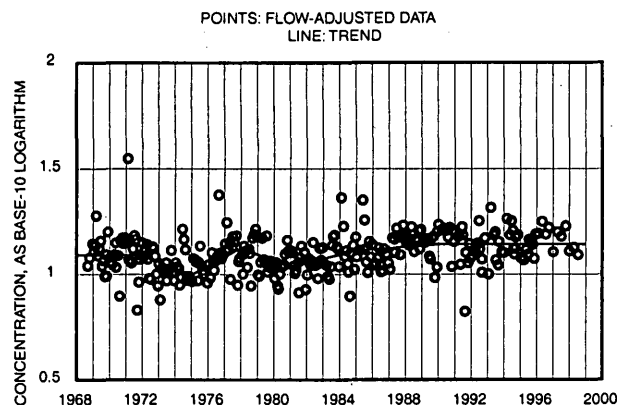
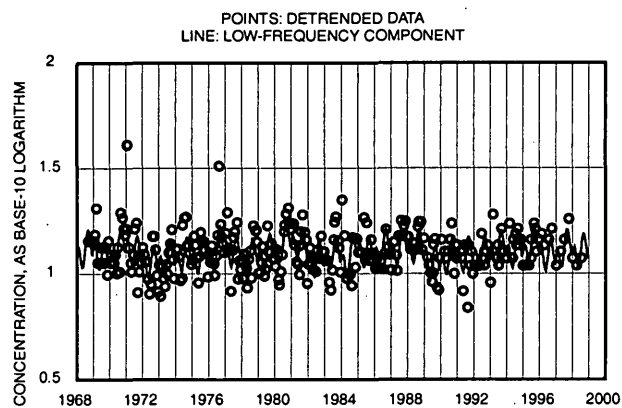
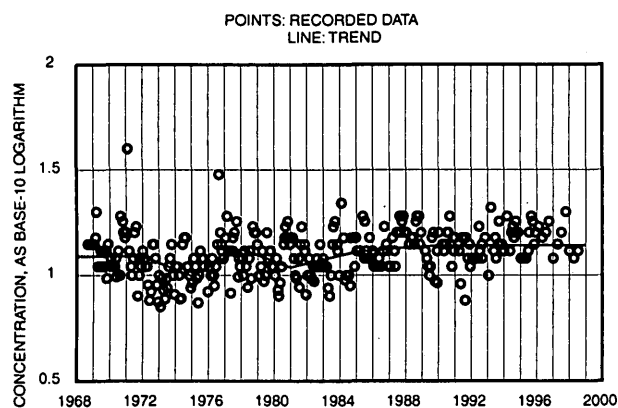
Upper-right graph: Detrended data, $Y - T$, and fitted low-frequency component, $C + A + S$ (see eq. 2);

Lower-left graph: Flow-adjusted data, $Y - A - S$, and fitted trend, $C + T$ (see eq. 2). These graphs correspond to figures 9 through 14 in the report.

Lower-right graph: PARMA model residuals (see Vecchia, 2000, Appendix A).

Salmon River near East Hampton, Connecticut

DISSOLVED CHLORIDE, IN MILLIGRAMS PER LITER (AS CL)



Appendix 2. Selected trend models of water-quality data for streams in Connecticut by station and constituent—Continued

The water-quality data and selected time-series model output used in this report are shown in this appendix.
The graphs in each of the figures are interpreted as follows:

Upper-left graph: Recorded data, Y , and fitted trend $C + T$ (see eq. 2);

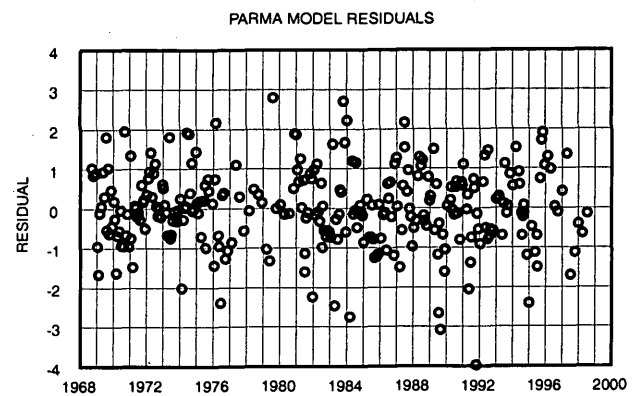
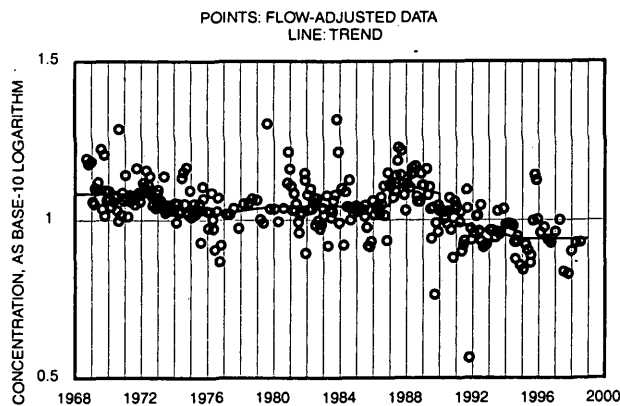
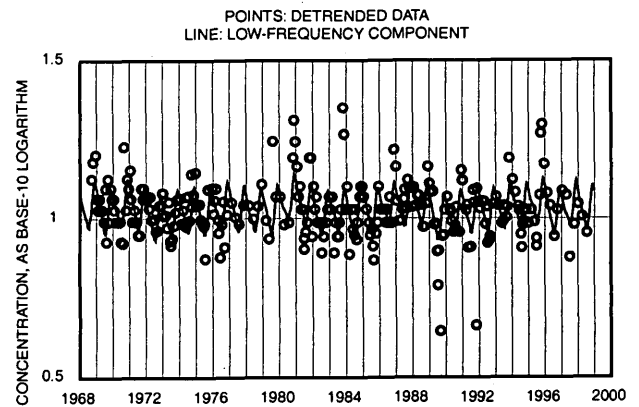
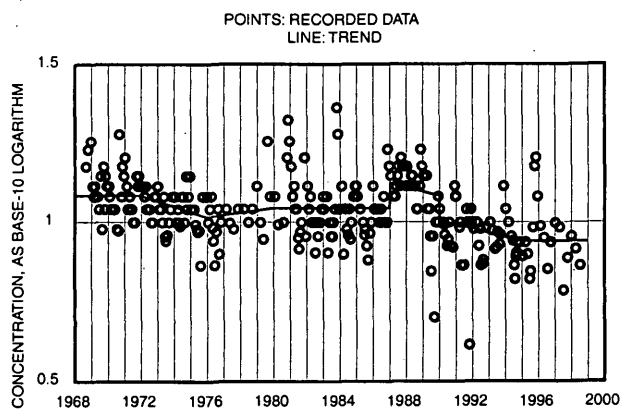
Upper-right graph: Detrended data, $Y - T$, and fitted low-frequency component, $C + A + S$ (see eq. 2);

Lower-left graph: Flow-adjusted data, $Y - A - S$, and fitted trend, $C + T$ (see eq. 2). These graphs correspond to figures 9 through 14 in the report.

Lower-right graph: PARMA model residuals (see Vecchia, 2000, Appendix A).

Salmon River near East Hampton, Connecticut

DISSOLVED SULFATE, IN MILLIGRAMS PER LITER (AS SO₄)



Appendix 2. Selected trend models of water-quality data for streams in Connecticut by station and constituent—Continued

The water-quality data and selected time-series model output used in this report are shown in this appendix.

The graphs in each of the figures are interpreted as follows:

Upper-left graph: Recorded data, Y , and fitted trend $C + T$ (see eq. 2);

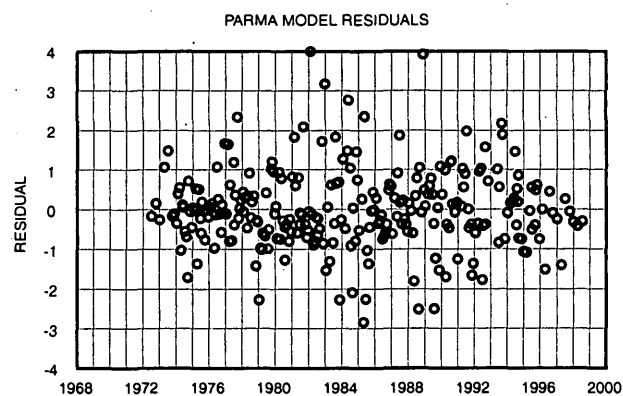
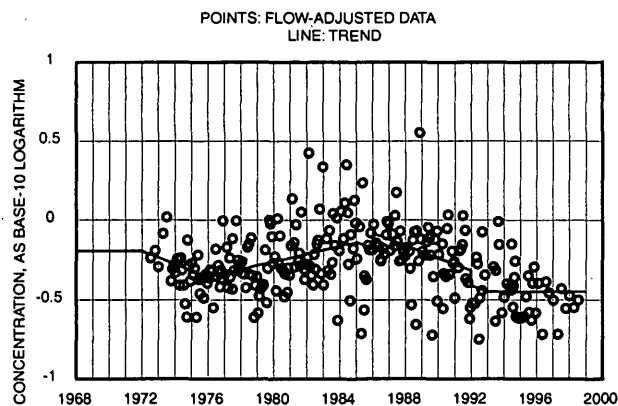
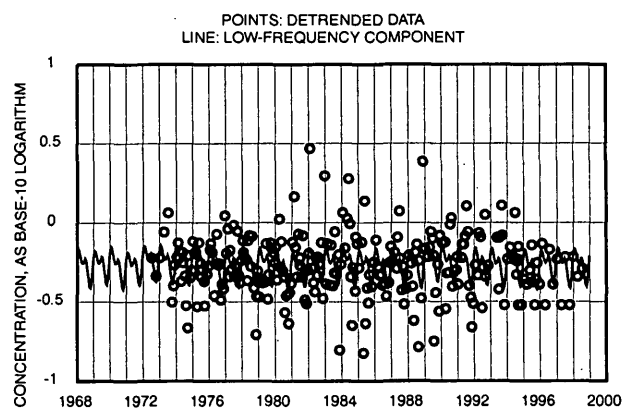
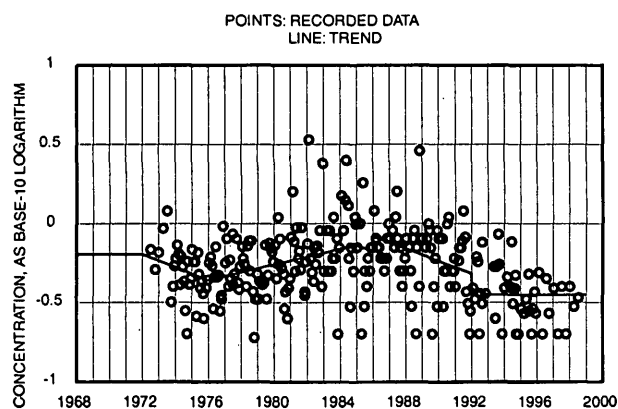
Upper-right graph: Detrended data, $Y - T$, and fitted low-frequency component, $C + A + S$ (see eq. 2);

Lower-left graph: Flow-adjusted data, $Y - A - S$, and fitted trend, $C + T$ (see eq. 2). These graphs correspond to figures 9 through 14 in the report.

Lower-right graph: PARMA model residuals (see Vecchia, 2000, Appendix A).

Salmon River near East Hampton, Connecticut

TOTAL NITROGEN, IN MILLIGRAMS PER LITER (AS N)



Appendix 2. Selected trend models of water-quality data for streams in Connecticut by station and constituent—Continued

The water-quality data and selected time-series model output used in this report are shown in this appendix.

The graphs in each of the figures are interpreted as follows:

Upper-left graph: Recorded data, Y , and fitted trend $C + T$ (see eq. 2);

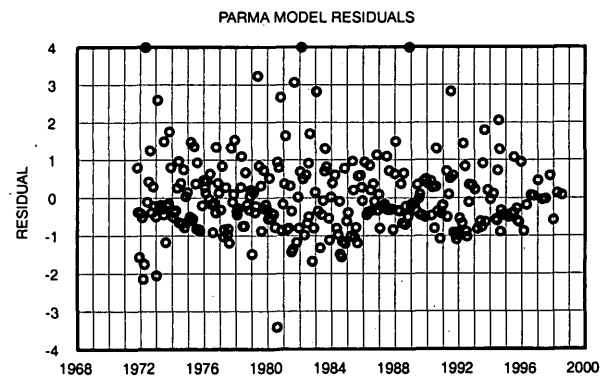
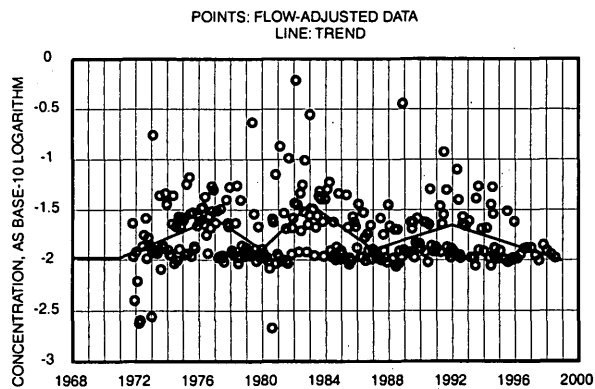
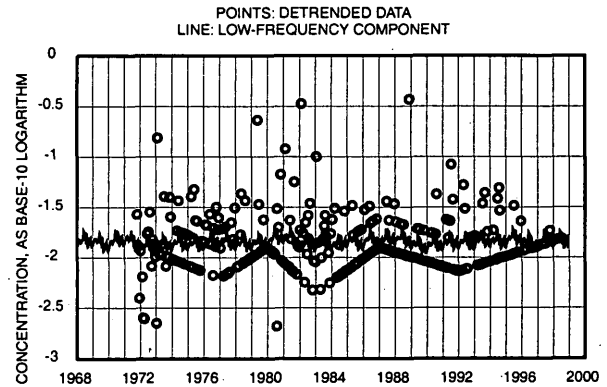
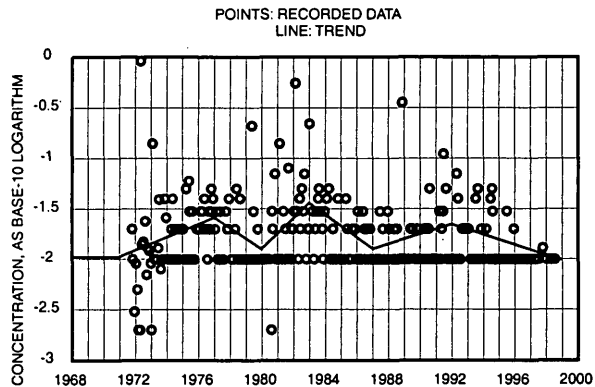
Upper-right graph: Detrended data, $Y - T$, and fitted low-frequency component, $C + A + S$ (see eq. 2);

Lower-left graph: Flow-adjusted data, $Y - A - S$, and fitted trend, $C + T$ (see eq. 2). These graphs correspond to figures 9 through 14 in the report.

Lower-right graph: PARMA model residuals (see Vecchia, 2000, Appendix A).

Salmon River near East Hampton, Connecticut

TOTAL PHOSPHORUS, IN MILLIGRAMS PER LITER (AS P)



Appendix 2. Selected trend models of water-quality data for streams in Connecticut by station and constituent—Continued

The water-quality data and selected time-series model output used in this report are shown in this appendix.

The graphs in each of the figures are interpreted as follows:

Upper-left graph: Recorded data, Y , and fitted trend $C + T$ (see eq. 2);

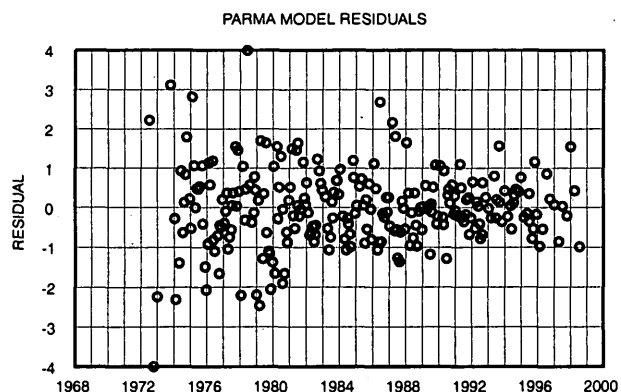
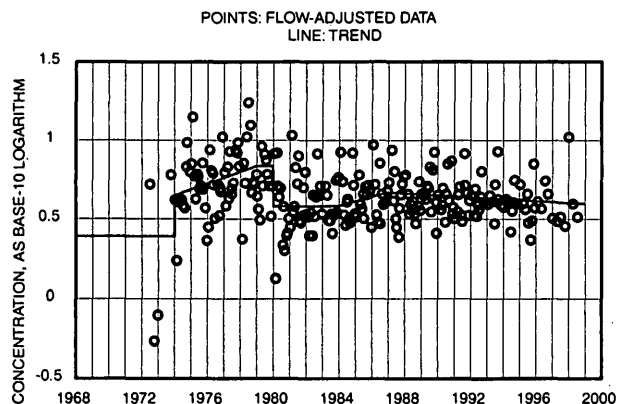
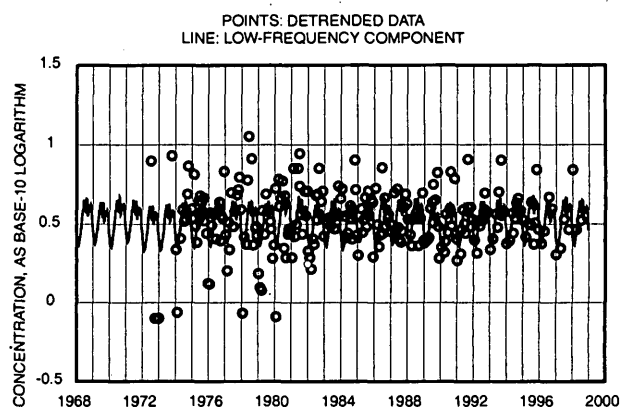
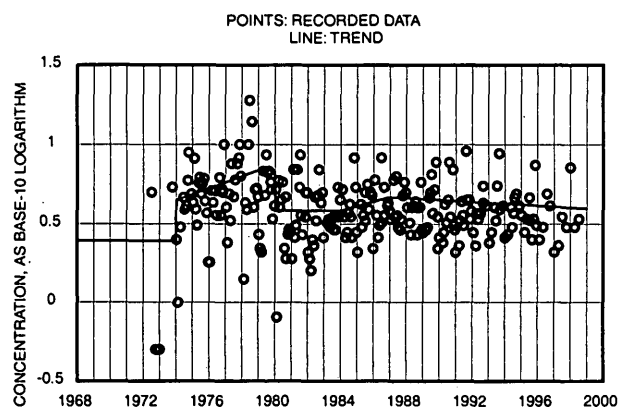
Upper-right graph: Detrended data, $Y - T$, and fitted low-frequency component, $C + A + S$ (see eq. 2);

Lower-left graph: Flow-adjusted data, $Y - A - S$, and fitted trend, $C + T$ (see eq. 2). These graphs correspond to figures 9 through 14 in the report.

Lower-right graph: PARMA model residuals (see Vecchia, 2000, Appendix A).

Salmon River near East Hampton, Connecticut

TOTAL ORGANIC CARBON, IN MILLIGRAMS PER LITER (AS C)



Appendix 2. Selected trend models of water-quality data for streams in Connecticut by station and constituent—Continued

The water-quality data and selected time-series model output used in this report are shown in this appendix.

The graphs in each of the figures are interpreted as follows:

Upper-left graph: Recorded data, Y , and fitted trend $C + T$ (see eq. 2);

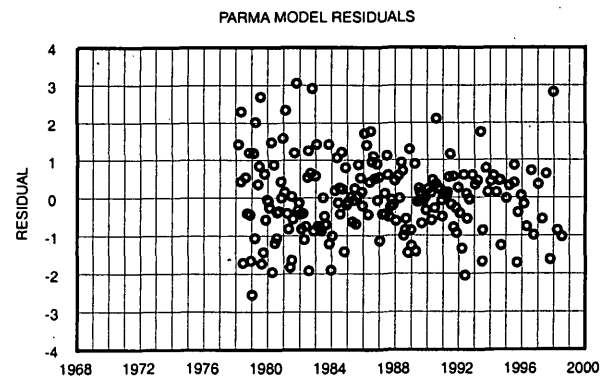
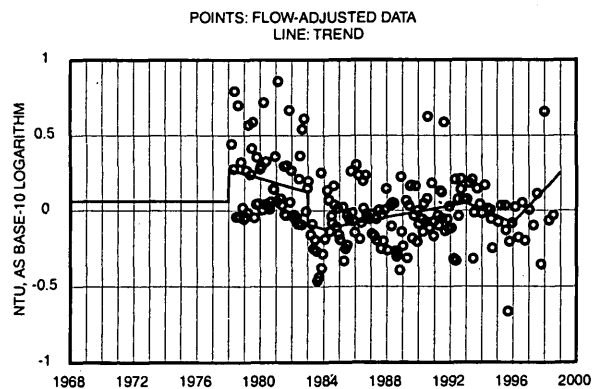
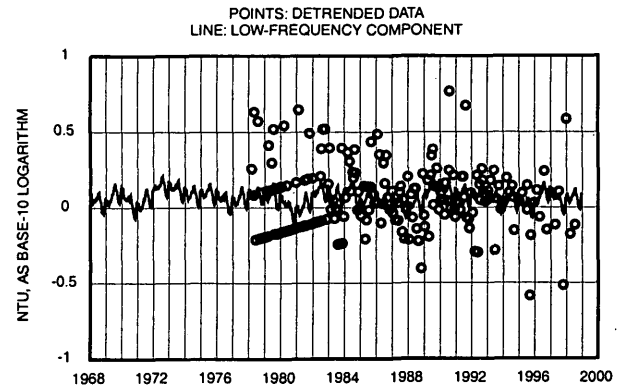
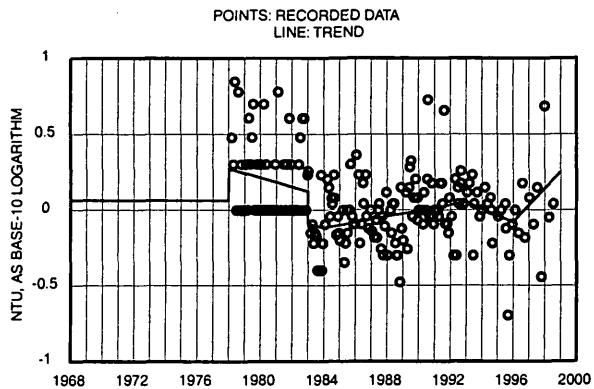
Upper-right graph: Detrended data, $Y - T$, and fitted low-frequency component, $C + A + S$ (see eq. 2);

Lower-left graph: Flow-adjusted data, $Y - A - S$, and fitted trend, $C + T$ (see eq. 2). These graphs correspond to figures 9 through 14 in the report.

Lower-right graph: PARMA model residuals (see Vecchia, 2000, Appendix A).

Salmon River near East Hampton, Connecticut

TURBIDITY, IN NEPHELOMETRIC TURBIDITY UNITS



Appendix 2. Selected trend models of water-quality data for streams in Connecticut by station and constituent—Continued

The water-quality data and selected time-series model output used in this report are shown in this appendix.

The graphs in each of the figures are interpreted as follows:

Upper-left graph: Recorded data, Y , and fitted trend $C + T$ (see eq. 2);

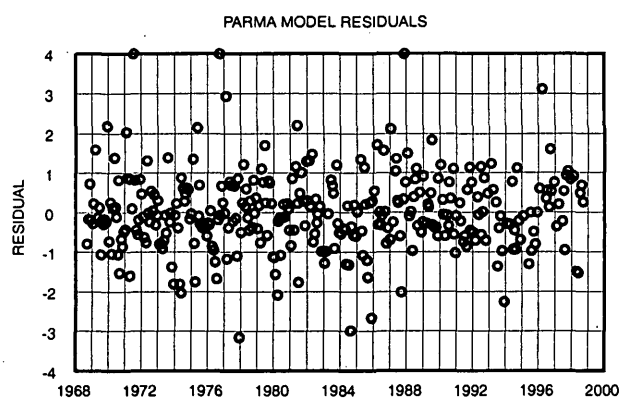
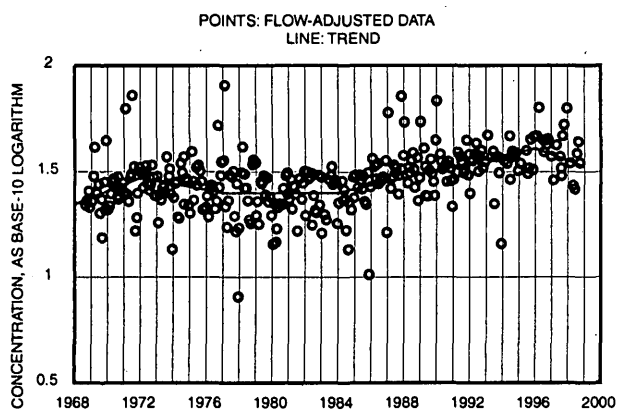
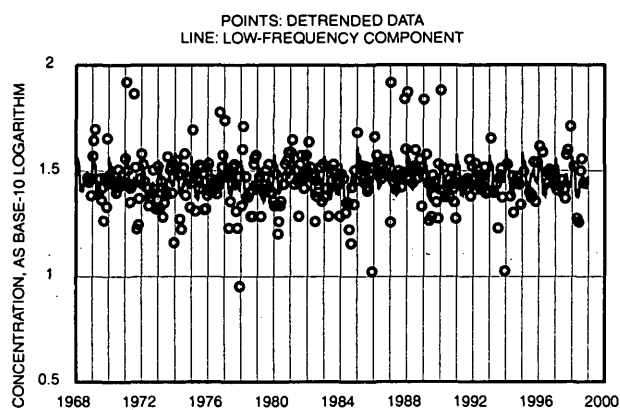
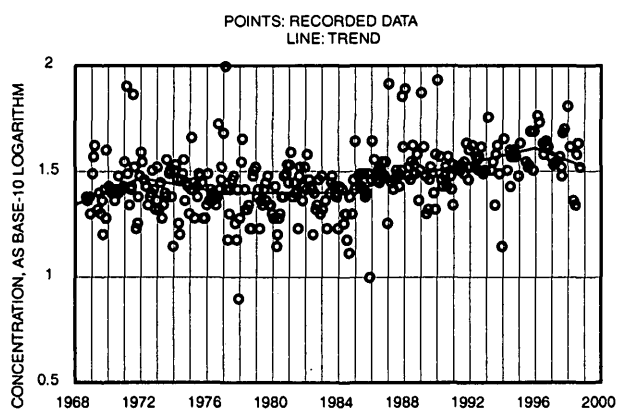
Upper-right graph: Detrended data, $Y - T$, and fitted low-frequency component, $C + A + S$ (see eq. 2);

Lower-left graph: Flow-adjusted data, $Y - A - S$, and fitted trend, $C + T$ (see eq. 2). These graphs correspond to figures 9 through 14 in the report.

Lower-right graph: PARMA model residuals (see Vecchia, 2000, Appendix A).

Quinnipiac River at Wallingford, Connecticut

DISSOLVED CHLORIDE, IN MILLIGRAMS PER LITER (AS CL)



Appendix 2. Selected trend models of water-quality data for streams in Connecticut by station and constituent—Continued

The water-quality data and selected time-series model output used in this report are shown in this appendix.

The graphs in each of the figures are interpreted as follows:

Upper-left graph: Recorded data, Y , and fitted trend $C + T$ (see eq. 2);

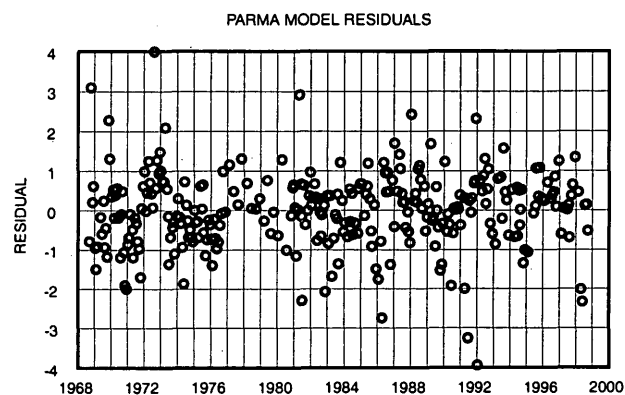
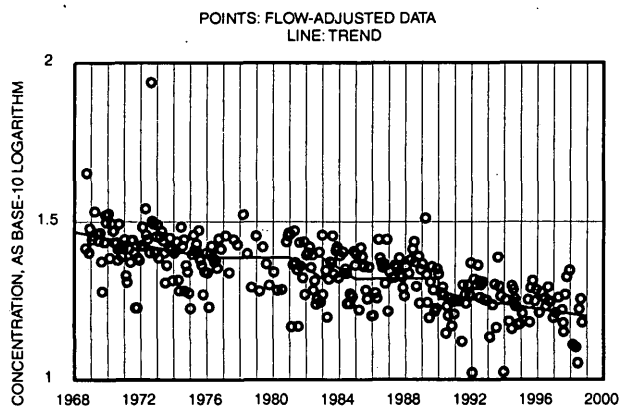
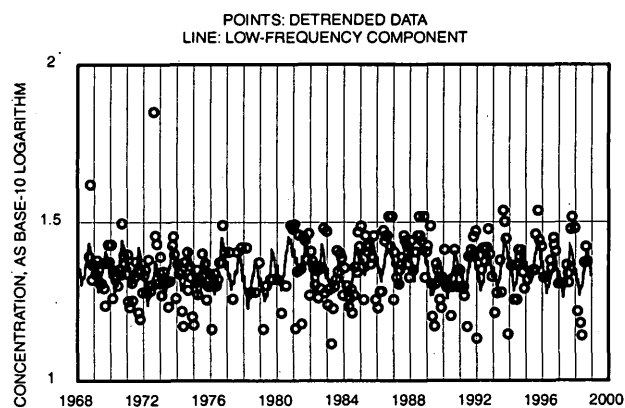
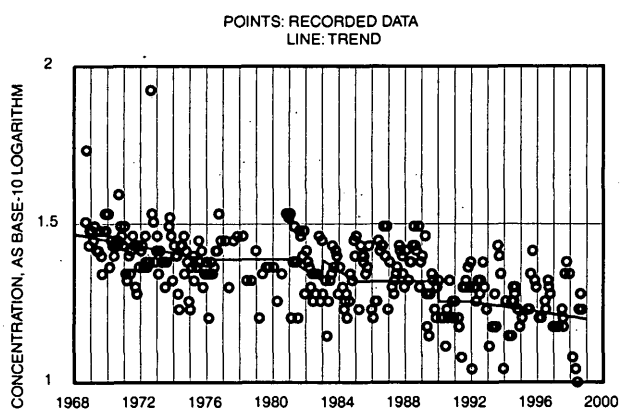
Upper-right graph: Detrended data, $Y - T$, and fitted low-frequency component, $C + A + S$ (see eq. 2);

Lower-left graph: Flow-adjusted data, $Y - A - S$, and fitted trend, $C + T$ (see eq. 2). These graphs correspond to figures 9 through 14 in the report.

Lower-right graph: PARMA model residuals (see Vecchia, 2000, Appendix A).

Quinnipiac River at Wallingford, Connecticut

DISSOLVED SULFATE, IN MILLIGRAMS PER LITER (AS SO₄)



Appendix 2. Selected trend models of water-quality data for streams in Connecticut by station and constituent—Continued

The water-quality data and selected time-series model output used in this report are shown in this appendix.

The graphs in each of the figures are interpreted as follows:

Upper-left graph: Recorded data, Y , and fitted trend $C + T$ (see eq. 2);

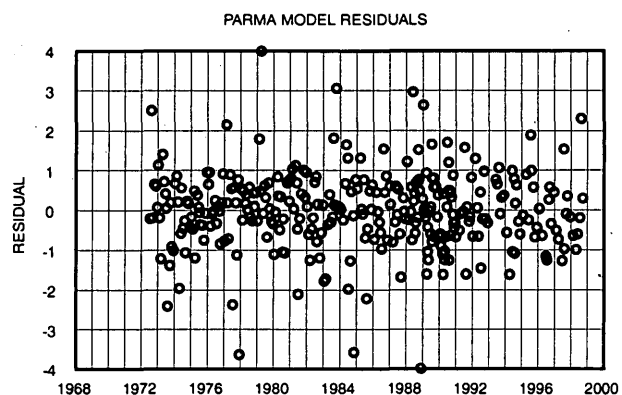
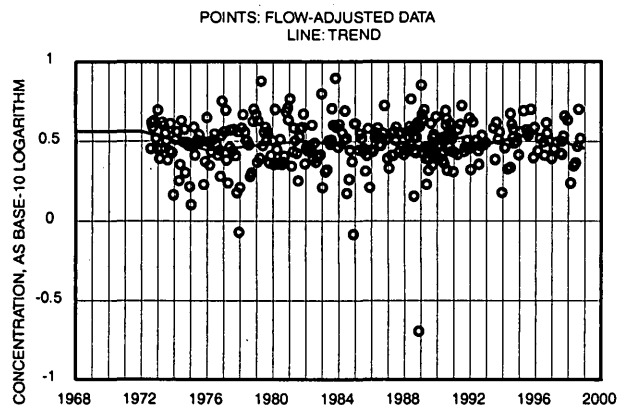
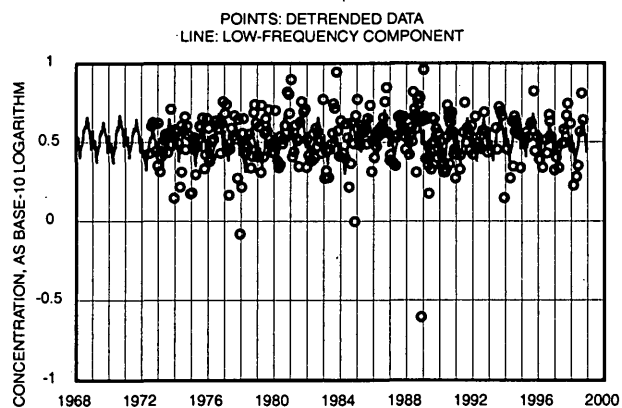
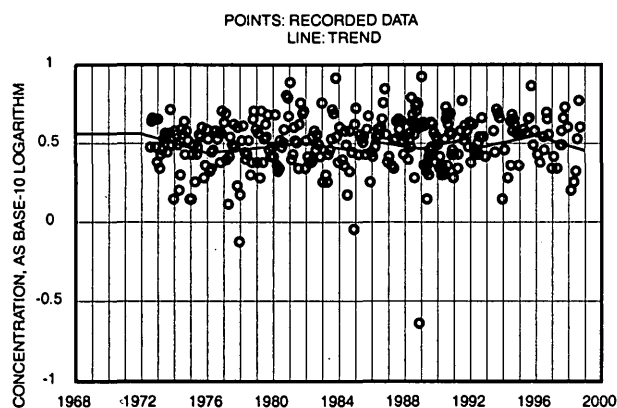
Upper-right graph: Detrended data, $Y - T$, and fitted low-frequency component, $C + A + S$ (see eq. 2);

Lower-left graph: Flow-adjusted data, $Y - A - S$, and fitted trend, $C + T$ (see eq. 2). These graphs correspond to figures 9 through 14 in the report.

Lower-right graph: PARMA model residuals (see Vecchia, 2000, Appendix A).

Quinnipiac River at Wallingford, Connecticut

TOTAL NITROGEN, IN MILLIGRAMS PER LITER (AS N)



Appendix 2. Selected trend models of water-quality data for streams in Connecticut by station and constituent—Continued

The water-quality data and selected time-series model output used in this report are shown in this appendix.

The graphs in each of the figures are interpreted as follows:

Upper-left graph: Recorded data, Y , and fitted trend $C + T$ (see eq. 2);

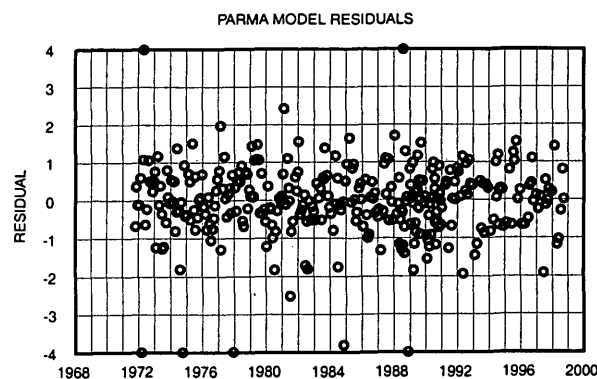
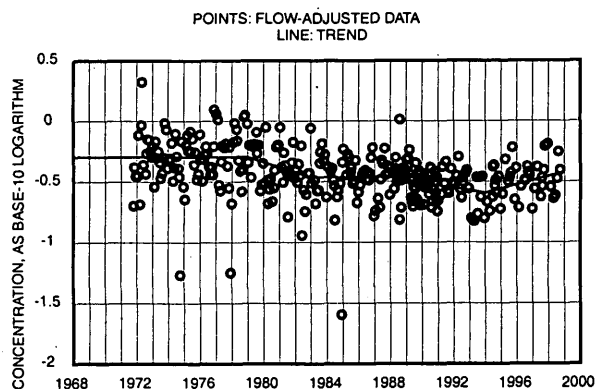
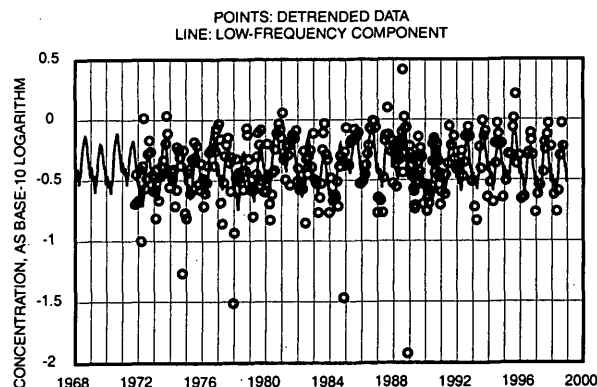
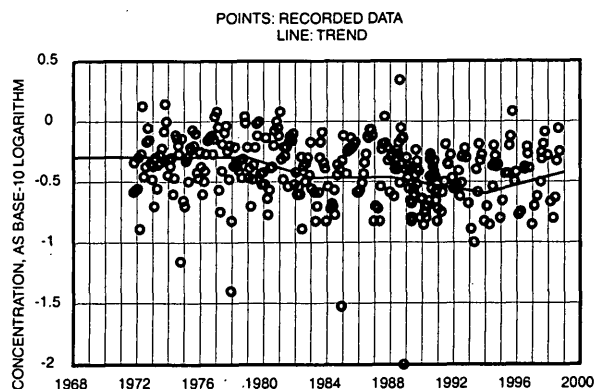
Upper-right graph: Detrended data, $Y - T$, and fitted low-frequency component, $C + A + S$ (see eq. 2);

Lower-left graph: Flow-adjusted data, $Y - A - S$, and fitted trend, $C + T$ (see eq. 2). These graphs correspond to figures 9 through 14 in the report.

Lower-right graph: PARMA model residuals (see Vecchia, 2000, Appendix A).

Quinnipiac River at Wallingford, Connecticut

TOTAL PHOSPHORUS, IN MILLIGRAMS PER LITER (AS P)



Appendix 2. Selected trend models of water-quality data for streams in Connecticut by station and constituent—Continued

The water-quality data and selected time-series model output used in this report are shown in this appendix.

The graphs in each of the figures are interpreted as follows:

Upper-left graph: Recorded data, Y , and fitted trend $C + T$ (see eq. 2);

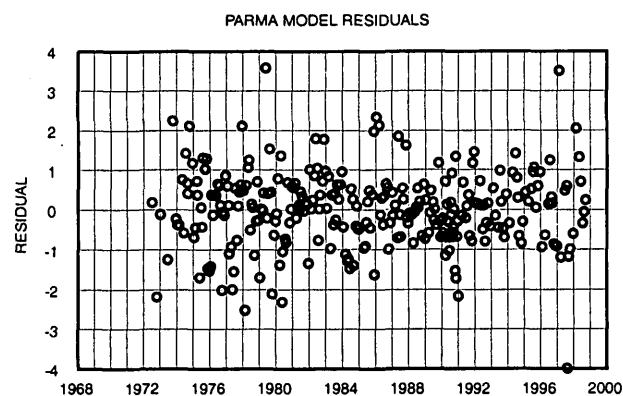
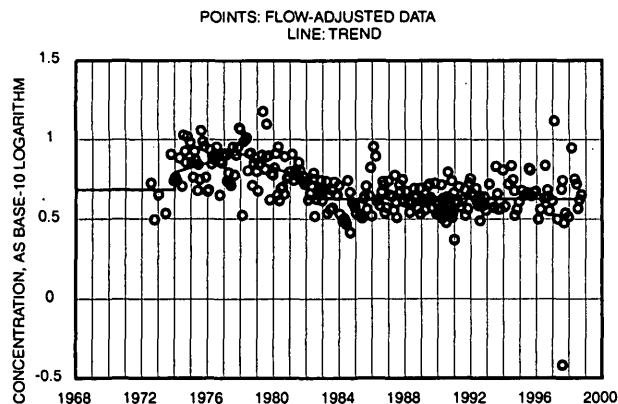
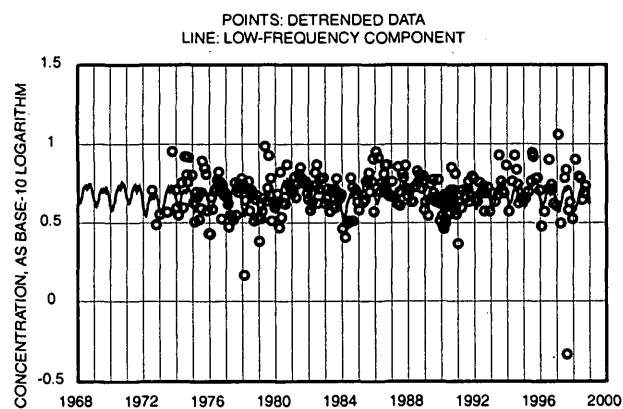
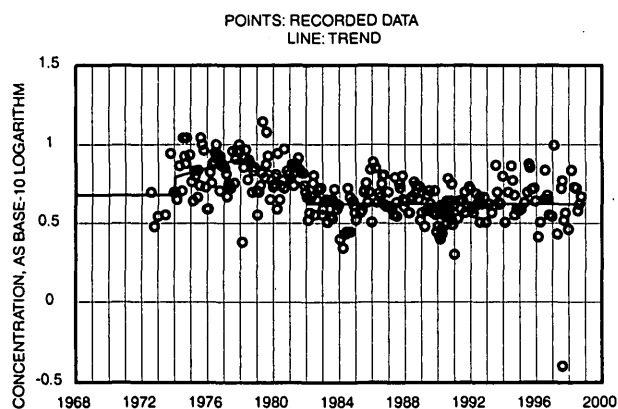
Upper-right graph: Detrended data, $Y - T$, and fitted low-frequency component, $C + A + S$ (see eq. 2);

Lower-left graph: Flow-adjusted data, $Y - A - S$, and fitted trend, $C + T$ (see eq. 2). These graphs correspond to figures 9 through 14 in the report.

Lower-right graph: PARMA model residuals (see Vecchia, 2000, Appendix A).

Quinnipiac River at Wallingford, Connecticut

TOTAL ORGANIC CARBON, IN MILLIGRAMS PER LITER (AS C)



Appendix 2. Selected trend models of water-quality data for streams in Connecticut by station and constituent—Continued

The water-quality data and selected time-series model output used in this report are shown in this appendix.

The graphs in each of the figures are interpreted as follows:

Upper-left graph: Recorded data, Y , and fitted trend $C + T$ (see eq. 2);

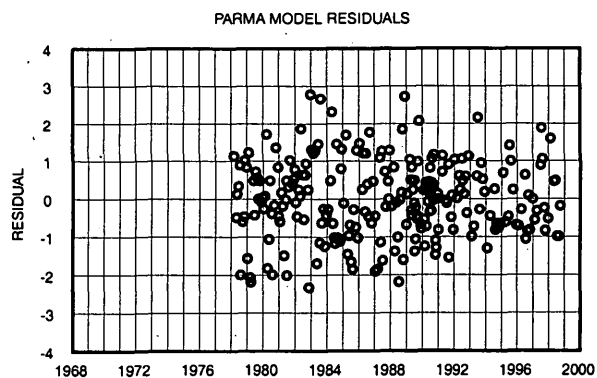
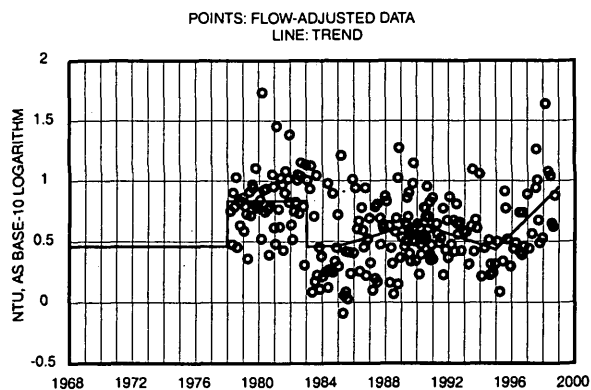
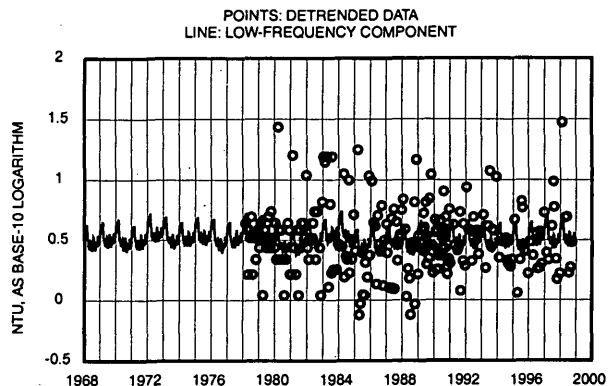
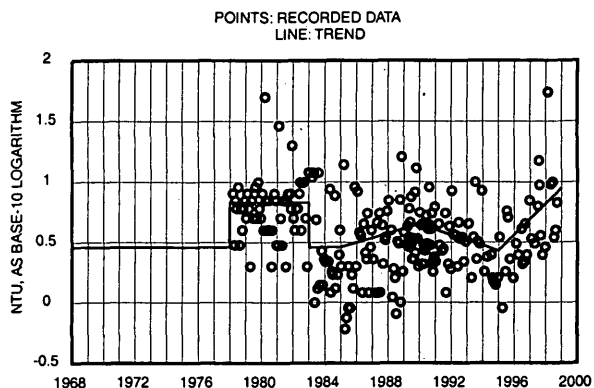
Upper-right graph: Detrended data, $Y - T$, and fitted low-frequency component, $C + A + S$ (see eq. 2);

Lower-left graph: Flow-adjusted data, $Y - A - S$, and fitted trend, $C + T$ (see eq. 2). These graphs correspond to figures 9 through 14 in the report.

Lower-right graph: PARMA model residuals (see Vecchia, 2000, Appendix A).

Quinnipiac River at Wallingford, Connecticut

TURBIDITY, IN NEPHELOMETRIC TURBIDITY UNITS



Appendix 2. Selected trend models of water-quality data for streams in Connecticut by station and constituent—Continued

The water-quality data and selected time-series model output used in this report are shown in this appendix.

The graphs in each of the figures are interpreted as follows:

Upper-left graph: Recorded data, Y , and fitted trend $C + T$ (see eq. 2);

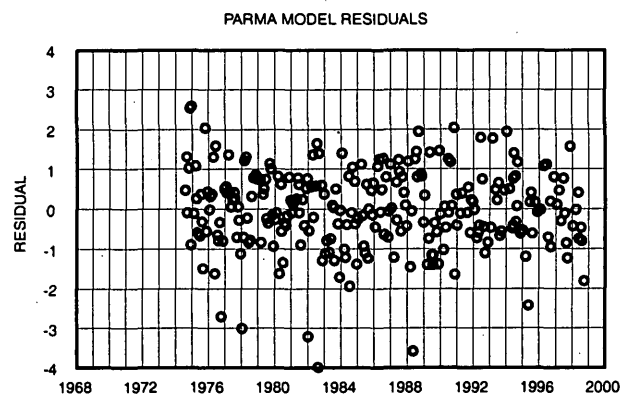
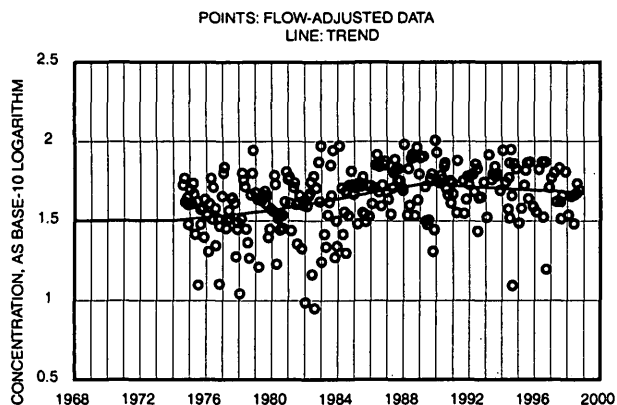
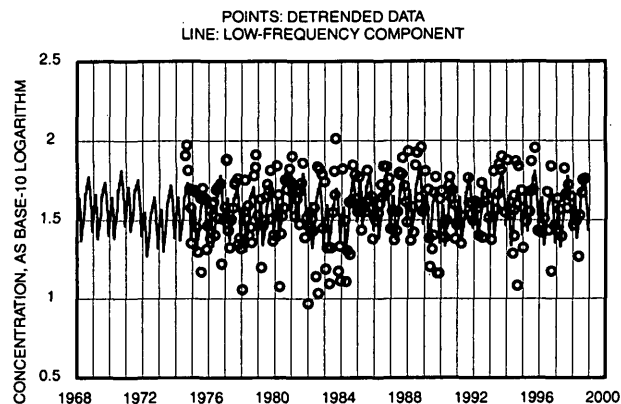
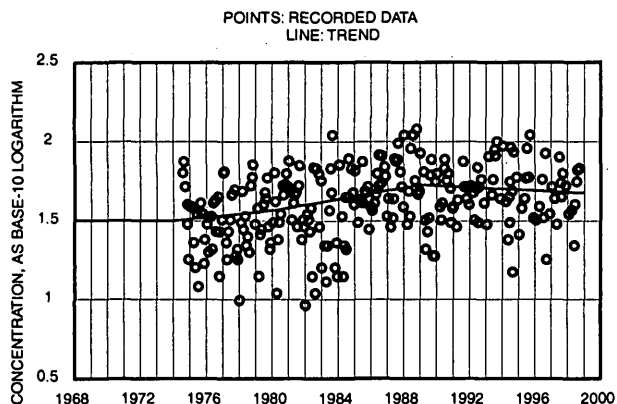
Upper-right graph: Detrended data, $Y - T$, and fitted low-frequency component, $C + A + S$ (see eq. 2);

Lower-left graph: Flow-adjusted data, $Y - A - S$, and fitted trend, $C + T$ (see eq. 2). These graphs correspond to figures 9 through 14 in the report.

Lower-right graph: PARMA model residuals (see Vecchia, 2000, Appendix A).

Naugatuck River at Beacon Falls, Connecticut

DISSOLVED CHLORIDE, IN MILLIGRAMS PER LITER (AS CL)



Appendix 2. Selected trend models of water-quality data for streams in Connecticut by station and constituent—Continued

The water-quality data and selected time-series model output used in this report are shown in this appendix.

The graphs in each of the figures are interpreted as follows:

Upper-left graph: Recorded data, Y , and fitted trend $C + T$ (see eq. 2);

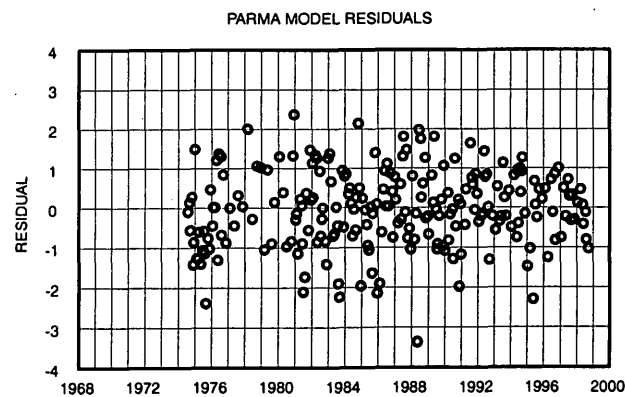
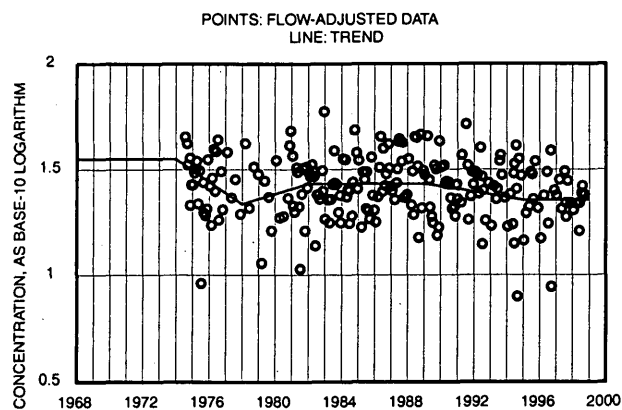
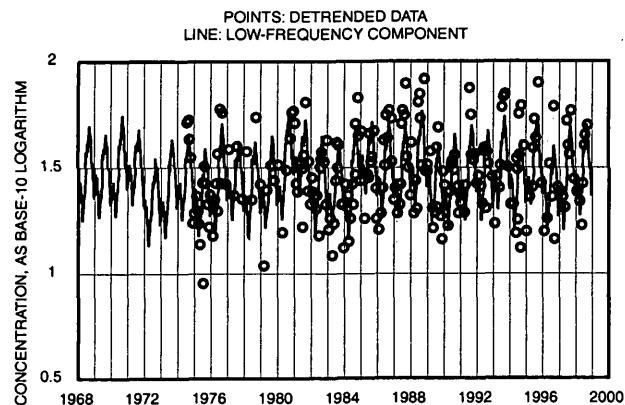
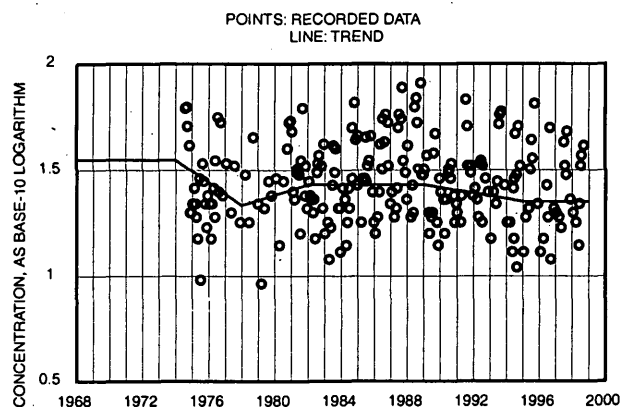
Upper-right graph: Detrended data, $Y - T$, and fitted low-frequency component, $C + A + S$ (see eq. 2);

Lower-left graph: Flow-adjusted data, $Y - A - S$, and fitted trend, $C + T$ (see eq. 2). These graphs correspond to figures 9 through 14 in the report.

Lower-right graph: PARMA model residuals (see Vecchia, 2000, Appendix A).

Naugatuck River at Beacon Falls, Connecticut

DISSOLVED SULFATE, IN MILLIGRAMS PER LITER (AS SO₄)



Appendix 2. Selected trend models of water-quality data for streams in Connecticut by station and constituent—Continued

The water-quality data and selected time-series model output used in this report are shown in this appendix.

The graphs in each of the figures are interpreted as follows:

Upper-left graph: Recorded data, Y , and fitted trend $C + T$ (see eq. 2);

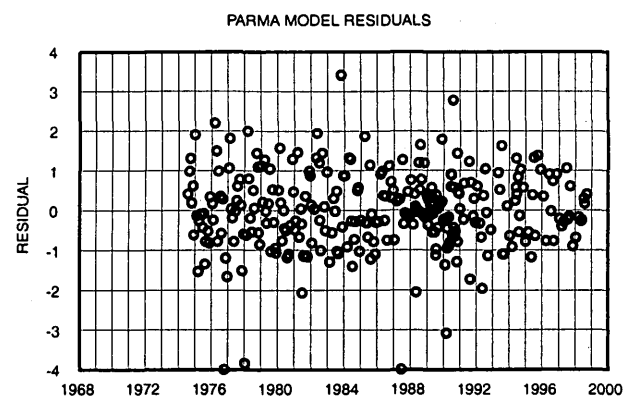
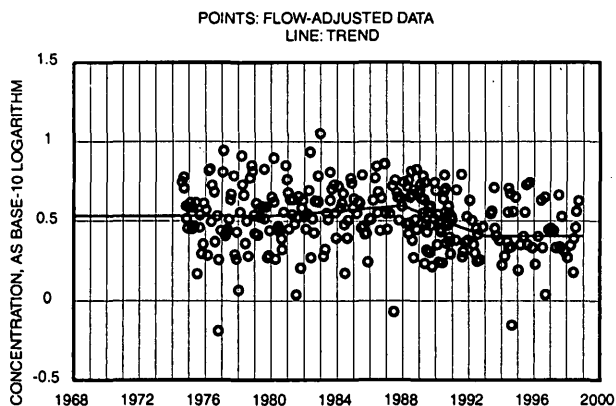
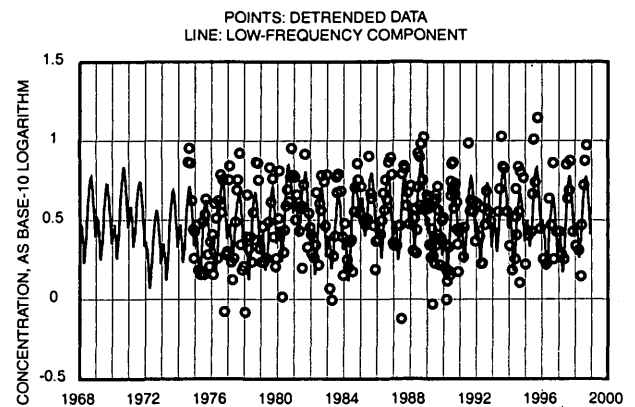
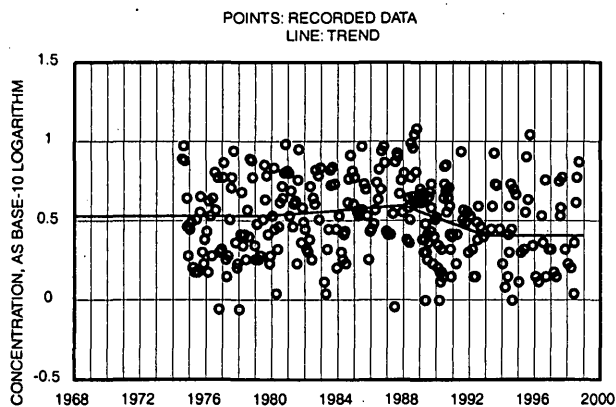
Upper-right graph: Detrended data, $Y - T$, and fitted low-frequency component, $C + A + S$ (see eq. 2);

Lower-left graph: Flow-adjusted data, $Y - A - S$, and fitted trend, $C + T$ (see eq. 2). These graphs correspond to figures 9 through 14 in the report.

Lower-right graph: PARMA model residuals (see Vecchia, 2000, Appendix A).

Naugatuck River at Beacon Falls, Connecticut

TOTAL NITROGEN, IN MILLIGRAMS PER LITER (AS N)



Appendix 2. Selected trend models of water-quality data for streams in Connecticut by station and constituent—Continued

The water-quality data and selected time-series model output used in this report are shown in this appendix.

The graphs in each of the figures are interpreted as follows:

Upper-left graph: Recorded data, Y , and fitted trend $C + T$ (see eq. 2);

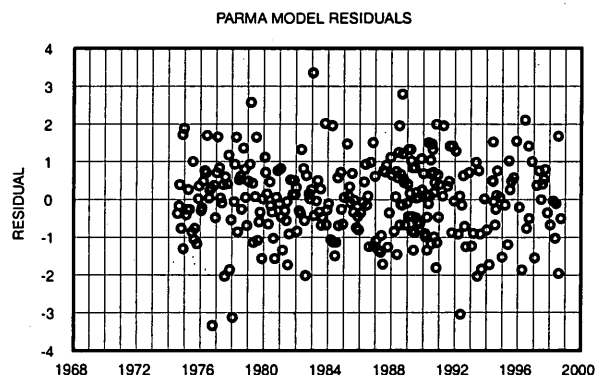
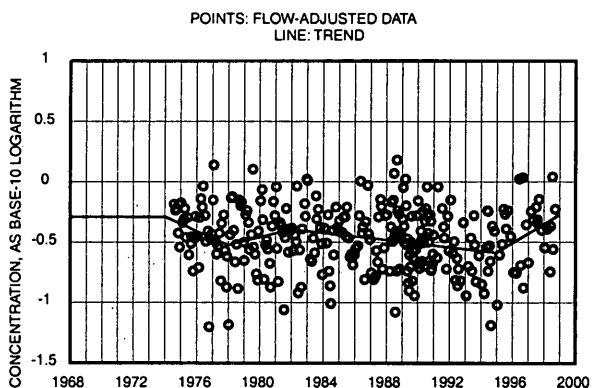
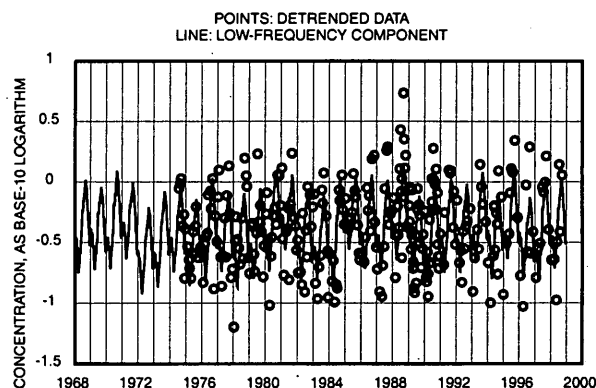
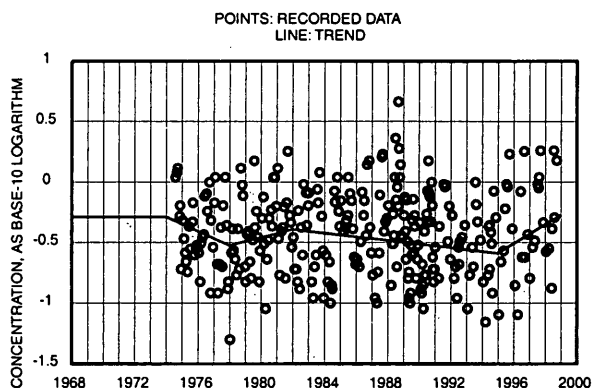
Upper-right graph: Detrended data, $Y - T$, and fitted low-frequency component, $C + A + S$ (see eq. 2);

Lower-left graph: Flow-adjusted data, $Y - A - S$, and fitted trend, $C + T$ (see eq. 2). These graphs correspond to figures 9 through 14 in the report.

Lower-right graph: PARMA model residuals (see Vecchia, 2000, Appendix A).

Naugatuck River at Beacon Falls, Connecticut

TOTAL PHOSPHORUS, IN MILLIGRAMS PER LITER (AS P)



Appendix 2. Selected trend models of water-quality data for streams in Connecticut by station and constituent—Continued

The water-quality data and selected time-series model output used in this report are shown in this appendix.

The graphs in each of the figures are interpreted as follows:

Upper-left graph: Recorded data, Y , and fitted trend $C + T$ (see eq. 2);

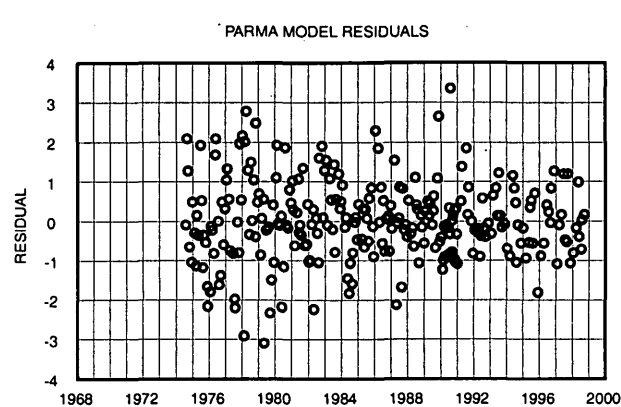
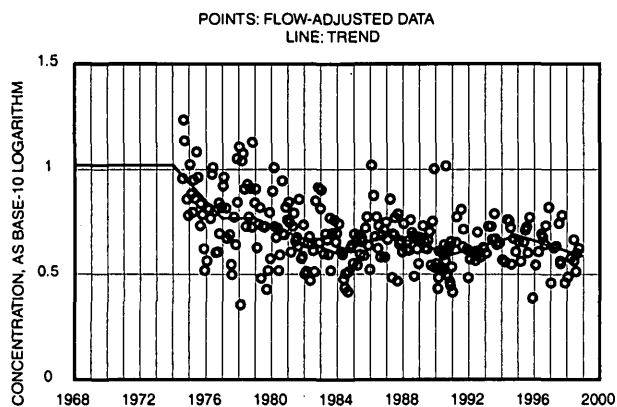
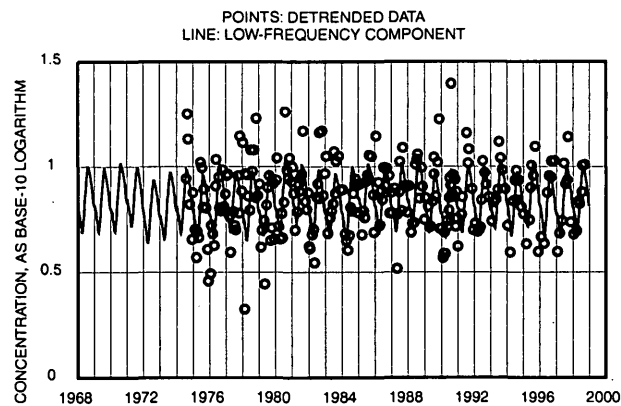
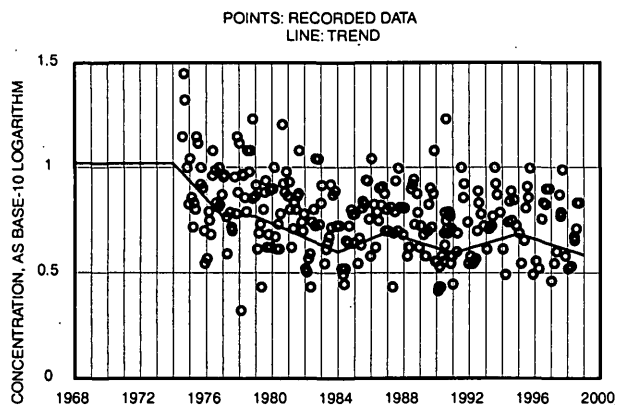
Upper-right graph: Detrended data, $Y - T$, and fitted low-frequency component, $C + A + S$ (see eq. 2);

Lower-left graph: Flow-adjusted data, $Y - A - S$, and fitted trend, $C + T$ (see eq. 2). These graphs correspond to figures 9 through 14 in the report.

Lower-right graph: PARMA model residuals (see Vecchia, 2000, Appendix A).

Naugatuck River at Beacon Falls, Connecticut

TOTAL ORGANIC CARBON, IN MILLIGRAMS PER LITER (AS C)



Appendix 2. Selected trend models of water-quality data for streams in Connecticut by station and constituent—Continued

The water-quality data and selected time-series model output used in this report are shown in this appendix.

The graphs in each of the figures are interpreted as follows:

Upper-left graph: Recorded data, Y , and fitted trend $C + T$ (see eq. 2);

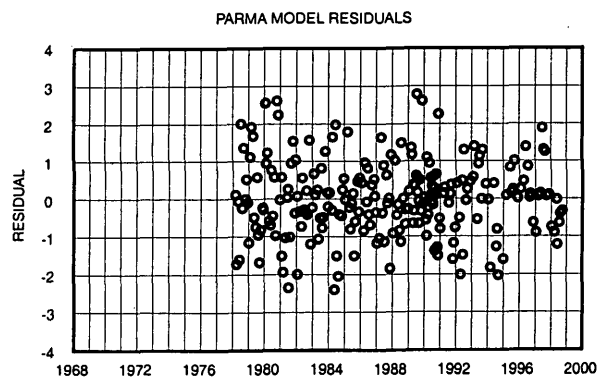
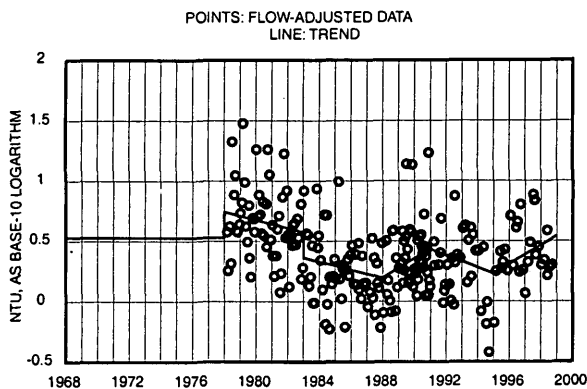
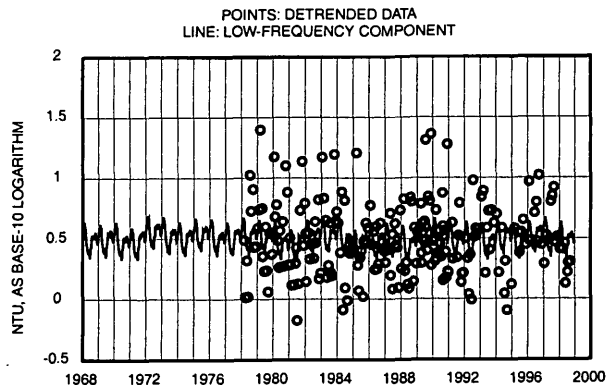
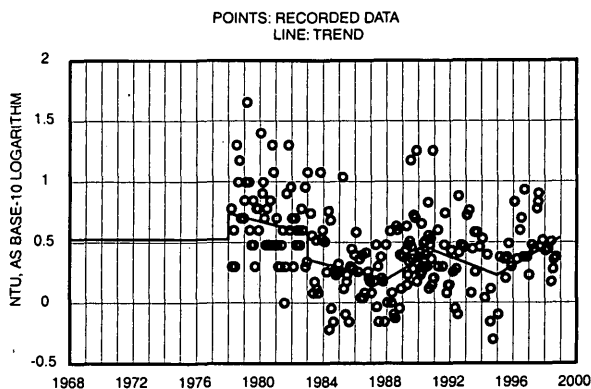
Upper-right graph: Detrended data, $Y - T$, and fitted low-frequency component, $C + A + S$ (see eq. 2);

Lower-left graph: Flow-adjusted data, $Y - A - S$, and fitted trend, $C + T$ (see eq. 2). These graphs correspond to figures 9 through 14 in the report.

Lower-right graph: PARMA model residuals (see Vecchia, 2000, Appendix A).

Naugatuck River at Beacon Falls, Connecticut

TURBIDITY, IN NEPHELOMETRIC TURBIDITY UNITS



Appendix 2. Selected trend models of water-quality data for streams in Connecticut by station and constituent—Continued

The water-quality data and selected time-series model output used in this report are shown in this appendix.

The graphs in each of the figures are interpreted as follows:

Upper-left graph: Recorded data, Y , and fitted trend $C + T$ (see eq. 2);

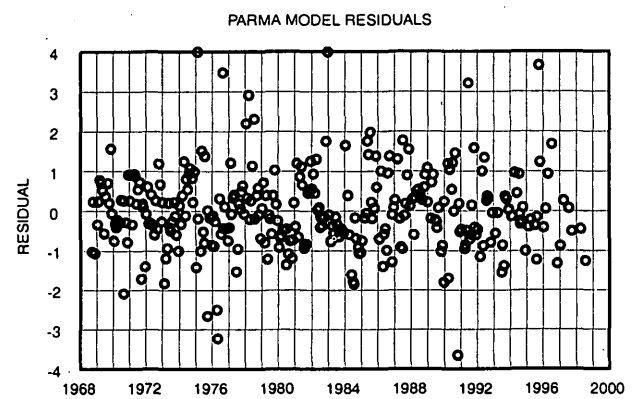
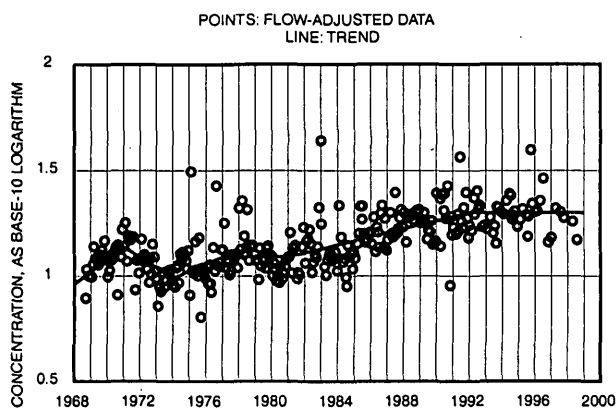
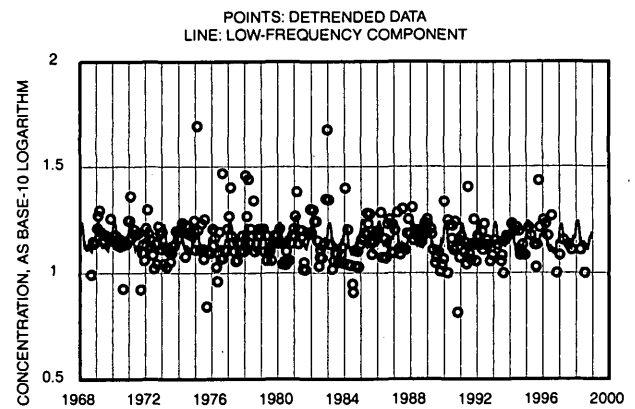
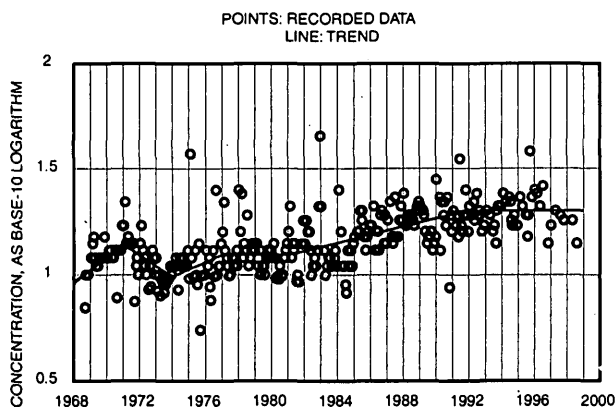
Upper-right graph: Detrended data, $Y - T$, and fitted low-frequency component, $C + A + S$ (see eq. 2);

Lower-left graph: Flow-adjusted data, $Y - A - S$, and fitted trend, $C + T$ (see eq. 2). These graphs correspond to figures 9 through 14 in the report.

Lower-right graph: PARMA model residuals (see Vecchia, 2000, Appendix A).

Saugatuck River near Redding, Connecticut

DISSOLVED CHLORIDE, IN MILLIGRAMS PER LITER (AS CL)



Appendix 2. Selected trend models of water-quality data for streams in Connecticut by station and constituent—Continued

The water-quality data and selected time-series model output used in this report are shown in this appendix.

The graphs in each of the figures are interpreted as follows:

Upper-left graph: Recorded data, Y , and fitted trend $C + T$ (see eq. 2);

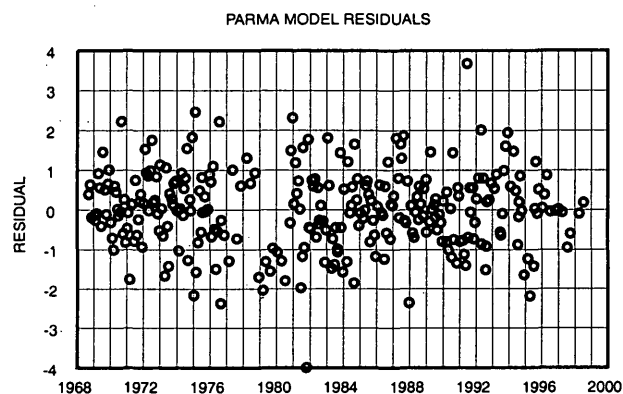
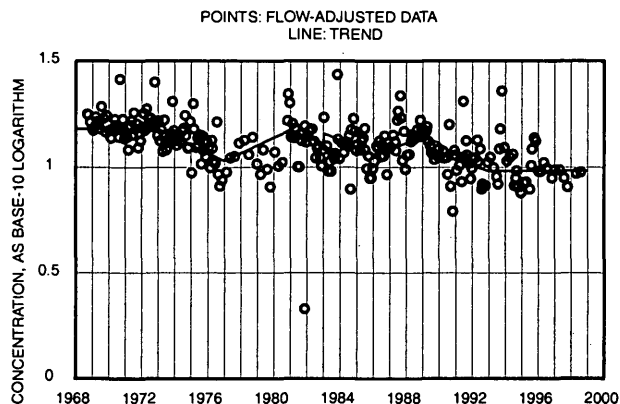
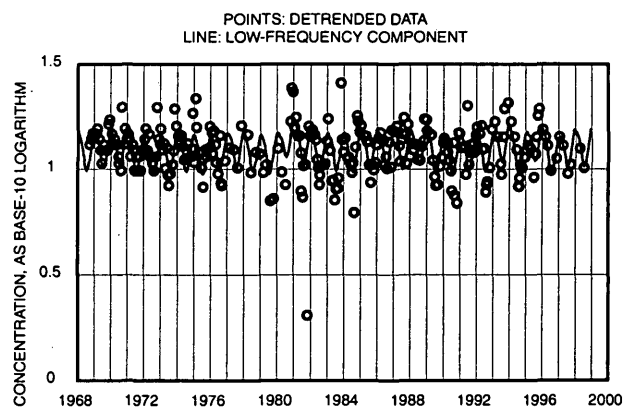
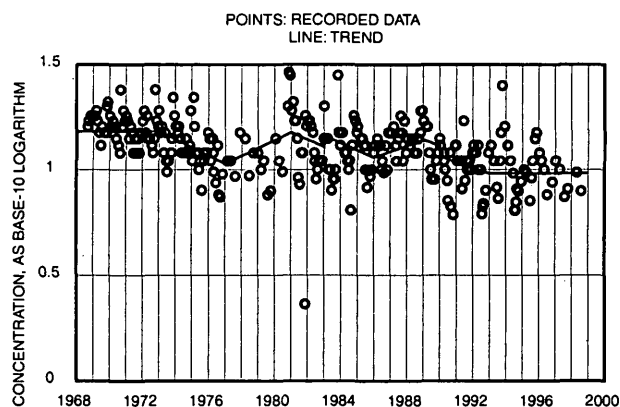
Upper-right graph: Detrended data, $Y - T$, and fitted low-frequency component, $C + A + S$ (see eq. 2);

Lower-left graph: Flow-adjusted data, $Y - A - S$, and fitted trend, $C + T$ (see eq. 2). These graphs correspond to figures 9 through 14 in the report.

Lower-right graph: PARMA model residuals (see Vecchia, 2000, Appendix A).

Saugatuck River near Redding, Connecticut

DISSOLVED SULFATE, IN MILLIGRAMS PER LITER (AS SO_4)



Appendix 2. Selected trend models of water-quality data for streams in Connecticut by station and constituent—Continued

The water-quality data and selected time-series model output used in this report are shown in this appendix.

The graphs in each of the figures are interpreted as follows:

Upper-left graph: Recorded data, Y , and fitted trend $C + T$ (see eq. 2);

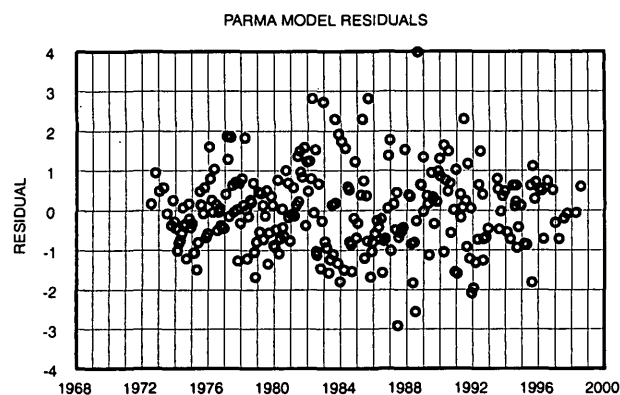
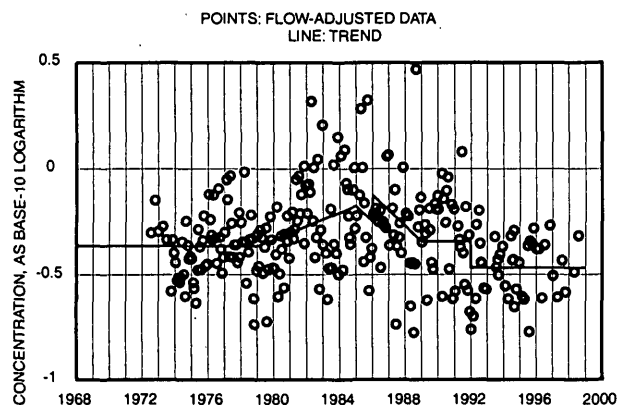
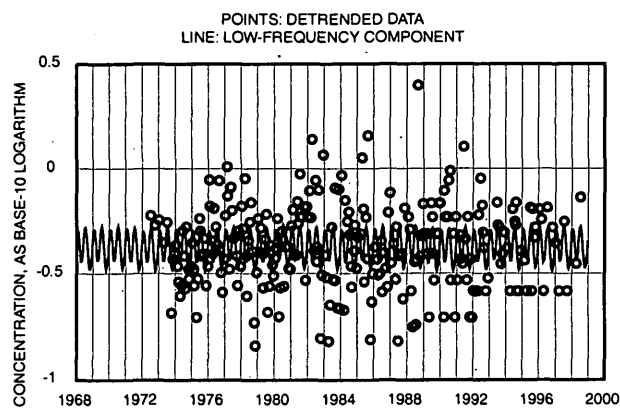
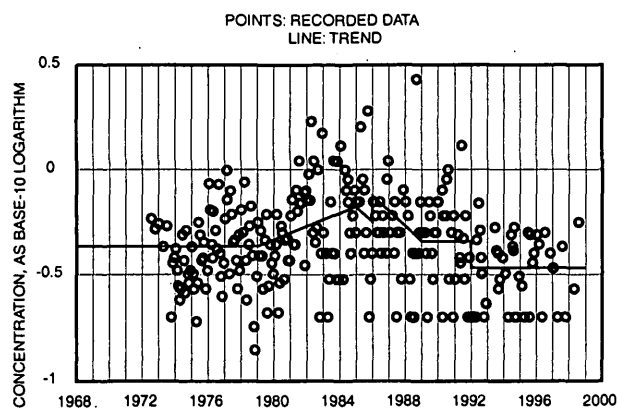
Upper-right graph: Detrended data, $Y - T$, and fitted low-frequency component, $C + A + S$ (see eq. 2);

Lower-left graph: Flow-adjusted data, $Y - A - S$, and fitted trend, $C + T$ (see eq. 2). These graphs correspond to figures 9 through 14 in the report.

Lower-right graph: PARMA model residuals (see Vecchia, 2000, Appendix A).

Saugatuck River near Redding, Connecticut

TOTAL NITROGEN, IN MILLIGRAMS PER LITER (AS N)



Appendix 2. Selected trend models of water-quality data for streams in Connecticut by station and constituent—Continued

The water-quality data and selected time-series model output used in this report are shown in this appendix.

The graphs in each of the figures are interpreted as follows:

Upper-left graph: Recorded data, Y , and fitted trend $C + T$ (see eq. 2);

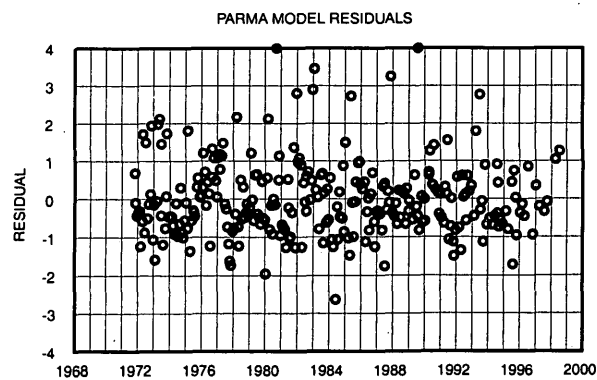
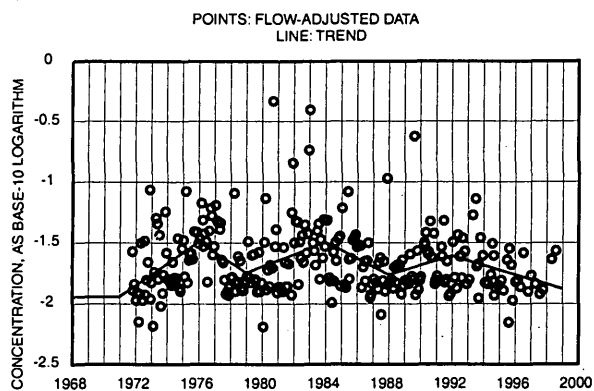
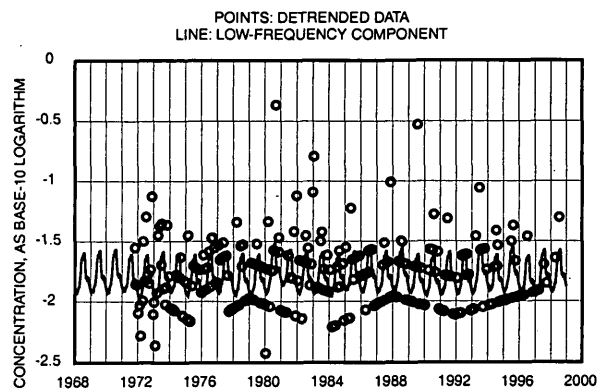
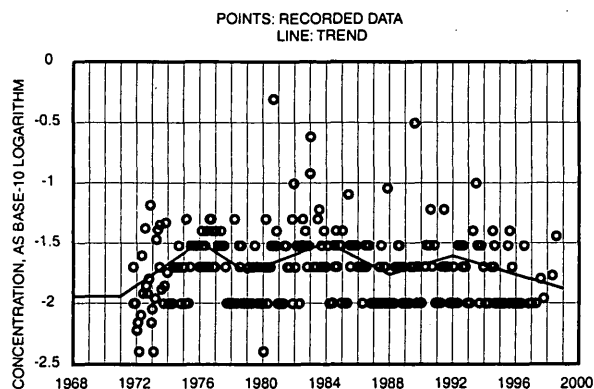
Upper-right graph: Detrended data, $Y - T$, and fitted low-frequency component, $C + A + S$ (see eq. 2);

Lower-left graph: Flow-adjusted data, $Y - A - S$, and fitted trend, $C + T$ (see eq. 2). These graphs correspond to figures 9 through 14 in the report.

Lower-right graph: PARMA model residuals (see Vecchia, 2000, Appendix A).

Saugatuck River near Redding, Connecticut

TOTAL PHOSPHORUS, IN MILLIGRAMS PER LITER (AS P)



Appendix 2. Selected trend models of water-quality data for streams in Connecticut by station and constituent—Continued

The water-quality data and selected time-series model output used in this report are shown in this appendix.

The graphs in each of the figures are interpreted as follows:

Upper-left graph: Recorded data, Y , and fitted trend $C + T$ (see eq. 2);

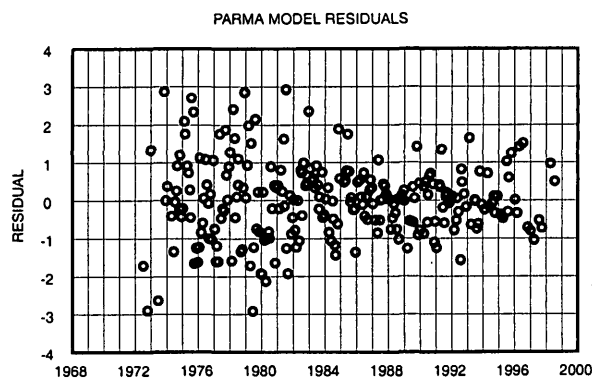
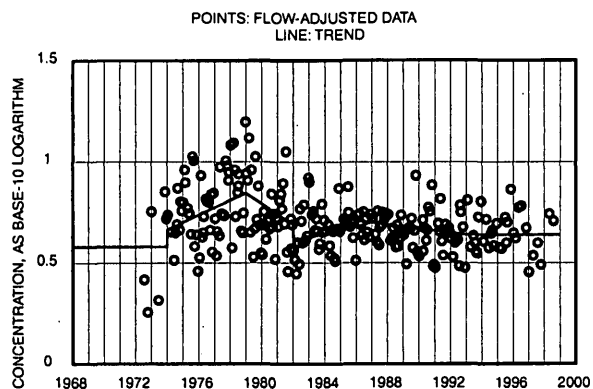
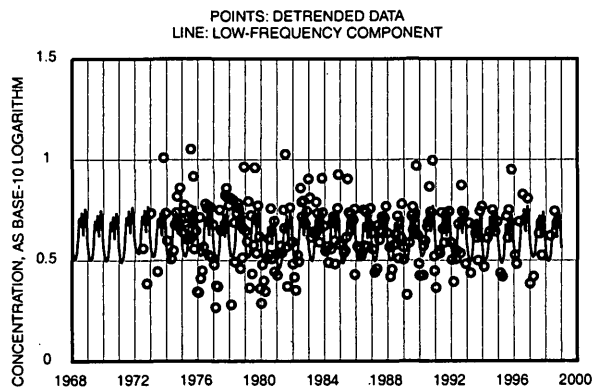
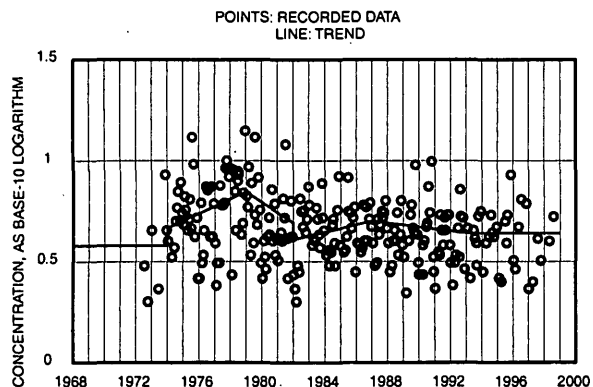
Upper-right graph: Detrended data, $Y - T$, and fitted low-frequency component, $C + A + S$ (see eq. 2);

Lower-left graph: Flow-adjusted data, $Y - A - S$, and fitted trend, $C + T$ (see eq. 2). These graphs correspond to figures 9 through 14 in the report.

Lower-right graph: PARMA model residuals (see Vecchia, 2000, Appendix A).

Saugatuck River near Redding, Connecticut

TOTAL ORGANIC CARBON, IN MILLIGRAMS PER LITER (AS C)



Appendix 2. Selected trend models of water-quality data for streams in Connecticut by station and constituent—Continued

The water-quality data and selected time-series model output used in this report are shown in this appendix.

The graphs in each of the figures are interpreted as follows:

Upper-left graph: Recorded data, Y , and fitted trend $C + T$ (see eq. 2);

Upper-right graph: Detrended data, $Y - T$, and fitted low-frequency component, $C + A + S$ (see eq. 2);

Lower-left graph: Flow-adjusted data, $Y - A - S$, and fitted trend, $C + T$ (see eq. 2). These graphs correspond to figures 9 through 14 in the report.

Lower-right graph: PARMA model residuals (see Vecchia, 2000, Appendix A).

Saugatuck River near Redding, Connecticut

TURBIDITY, IN NEPHELOMETRIC TURBIDITY UNITS

