U.S. Department of the Interior
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Hydrology and Water Quality of Geneva Lake, Walworth County, Wisconsin

Water-Resources Investigations Report 02–4039

Prepared in cooperation with the
Geneva Lake Environmental Agency
Wisconsin Department of Natural Resources

USGS
science for a changing world

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By Dale M. Robertson, Gerald L. Goddard, Elizabeth A. Mergener, William J. Rose, and Paul J. Garrison

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Middleton, Wisconsin
2002

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

Multiply	By	To Obtain
micrometer (µm)	0.00003927	inch
millimeter (mm)	.03937	inch
centimeter (cm)	.3937	inch
meter (m)	3.2808	feet
kilometer (km)	.62137	miles
square kilometer (km^2)	247.105	acres
square kilometer (km^2)	.3861	square miles
hectare (ha)	2.47105	acres
hectare (ha)	.003861	square miles
cubic micrometers (μm^3)	6.056×10^{-14}	cubic inches
cubic meters (m^3)	35.3147	cubic feet
liter (L)	.26417	gallons
gram (g)	.0022	pounds
kilogram (kg)	2.2046	pounds
meters per second (m/s)	2.2369	miles per hour
cubic meters per day (m^3/d)	.000409	cubic feet per second
kilogram per square kilometer (kg/km^2)	.00892	pounds per acre
kilogram per square kilometer (kg/km^2)	5.7099	pounds per square mile

Temperature in degrees Celsius (°C) can be converted to degrees Fahrenheit (°F) by use of the following equation:

$$°F = 1.8\,(°C) + 32.$$

Water year: water year is defined as the period beginning October 1 and ending September 30, designated by the calendar year in which it ends.

Abbreviated water-quality units used in this report: Chemical concentrations and water temperature are given in metric units. Chemical concentration is given in milligrams per liter (mg/L) or micrograms per liter (µg/L). Milligrams per liter is a unit expressing the concentration of chemical constituents in solution as weight (milligrams) of solute per unit volume (liter) of water. One thousand micrograms per liter is equivalent to one milligram per liter. For concentrations less than 7,000 mg/L, the numerical value is the same as for concentrations in parts per million.

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

Other Abbreviations Used in this Report:

GLEA	Geneva Lake Environmental Agency
WDNR	Wisconsin Department of Natural Resources
SEWRPC	Southeastern Wisconsin Regional Planning Commission
P	Phosphorus
USGS	U.S. Geological Survey
N	Nitrogen
TSI_p	Trophic State Index based on phosphorus concentration
TSI_c	Trophic State Index based on chlorophyll a concentration
TSI_{sd}	Trophic State Index based on Secchi depth
WY	Water year
SS	Suspended sediment
EWI	Equal width interval
PAHs	Polycyclic aromatic hydrocarbons
PCBs	Polychlorinated biphenyls
N:P	Nitrogen to phosphorus
$\mu m^3/mL$	cubic micrometers per milliliter
g/kg	grams per kilogram
mg/kg	milligrams per kilogram
ΔS	Change in storage
PPT	Precipitation
SW	Surface water
GW	Ground water
Evap	Evaporation
DLM	Dynamic Lake Model
DYRESM	Dynamic Reservoir Simulation Model
kg/ha	Kilograms per hectare
M	P input from septic systems
E_S	Export coefficient
S_R	Soil retention coefficient
g/d	grams per day
$g/cm^2/yr$	grams per square centimeter per year

ACKNOWLEDGMENTS

Technical Reviewers

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Robert Wakeman, Southeast Region Aquatic Habitat Coordinator, Wisconsin Department of Natural Resources, Milwaukee, Wis.

Hebert S. Garn, Supervisory Hydrologist, U.S. Geological Survey, Middleton, Wis.

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HYDROLOGY AND WATER QUALITY OF GENEVA LAKE, WALWORTH COUNTY, WISCONSIN

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Abstract

As part of continuing efforts to improve the water quality of Geneva Lake, a collaborative effort between the U.S. Geological Survey, the Wisconsin Department of Natural Resources, and the Geneva Lake Environmental Agency was initiated in 1997 to document the present quality of the lake and its sediments, compute detailed hydrologic and nutrient (primarily phosphorus) budgets for the lake, estimate how changes in nutrient loading may affect water quality, and describe changes in the lake over the past 170 years by comparing water quality measured in this study with historical measurements and sediment-core information. This report presents the results of this collaborative study.

Measurements collected during this study (1997–2000) indicate that the trophic status of the lake ranges from mesotrophic to oligotrophic: the mean Secchi depth was 4.8 m (meters), mean surface phosphorus concentration was 9 µg/L (micrograms per liter), mean surface nitrogen concentration was 550 µg/L, and mean surface chlorophyll *a* concentration was 3 µg/L. Surface nitrogen:phosphorus ratios indicated that, if just these nutrients are considered, phosphorus should be the limiting nutrient.

Phosphorus budgets constructed for water years 1998 and 1999 indicate that recent annual phosphorus loads were about 2,000 kg (kilograms) less than that estimated in 1975 (total annual input was about 3,200 kg in 1998 and about 8,500 kg in 1999). The major source of phosphorus to the lake was from its tributaries, which contributed about 84 percent of the total load. The primary difference from the phosphorus load estimates for 1975 was the decrease in loading from the Fontana sewage-treatment plant.

Direct measurements and indirect measurements based on sediment-core analyses indicate that the water quality of Geneva Lake has degraded in the last 170 years, the greatest effects resulting from urbanization. Sedimentation rates were highest between 1900 to 1930, and phosphorus concentrations were highest between the 1930s to early 1980s. As a result of the recent reduction in phosphorus loading, in-lake near-surface phosphorus concentrations decreased from 20–25 µg/L to about 10–15 µg/L and are similar to those estimated for the lake in the early 1900s. Concentrations of other chemical constituents associated with urban areas, however, have continually increased, especially in Williams Bay and Geneva Bay.

INTRODUCTION

Geneva Lake (fig. 1), in Walworth County, has long been considered one of the most important natural resources in southeastern Wisconsin. Formed about 10,000 years ago, the lake is well known for its deep, clear water. The lake originally was called "Kishwauketoe," meaning clear water, by the Pottawatomie Indians. Early European settlers, who first visited the lake around 1830, observed an abundance of fish, game, and edible plants in the area. Since then, much of the area surrounding the lake has changed dramatically. By 1858, the lake had a steamboat for tourists (Jenkins, 1921), and by 1871, a railway was built that connected the City of Lake Geneva with Chicago, Ill. (Jenkins, 1921). With these developments came changes in the watershed; parts became agriculturalized and parts became urbanized. In more recent years, increases in residential and urban areas around Lake Geneva, Fontana, and Williams Bay (fig. 1) have replaced much of the forested and some of the agricultural areas in the watershed.

With these changes in the Geneva Lake watershed came additional stresses on the lake. Early concern by lakeshore property owners led to a State Statute in 1893

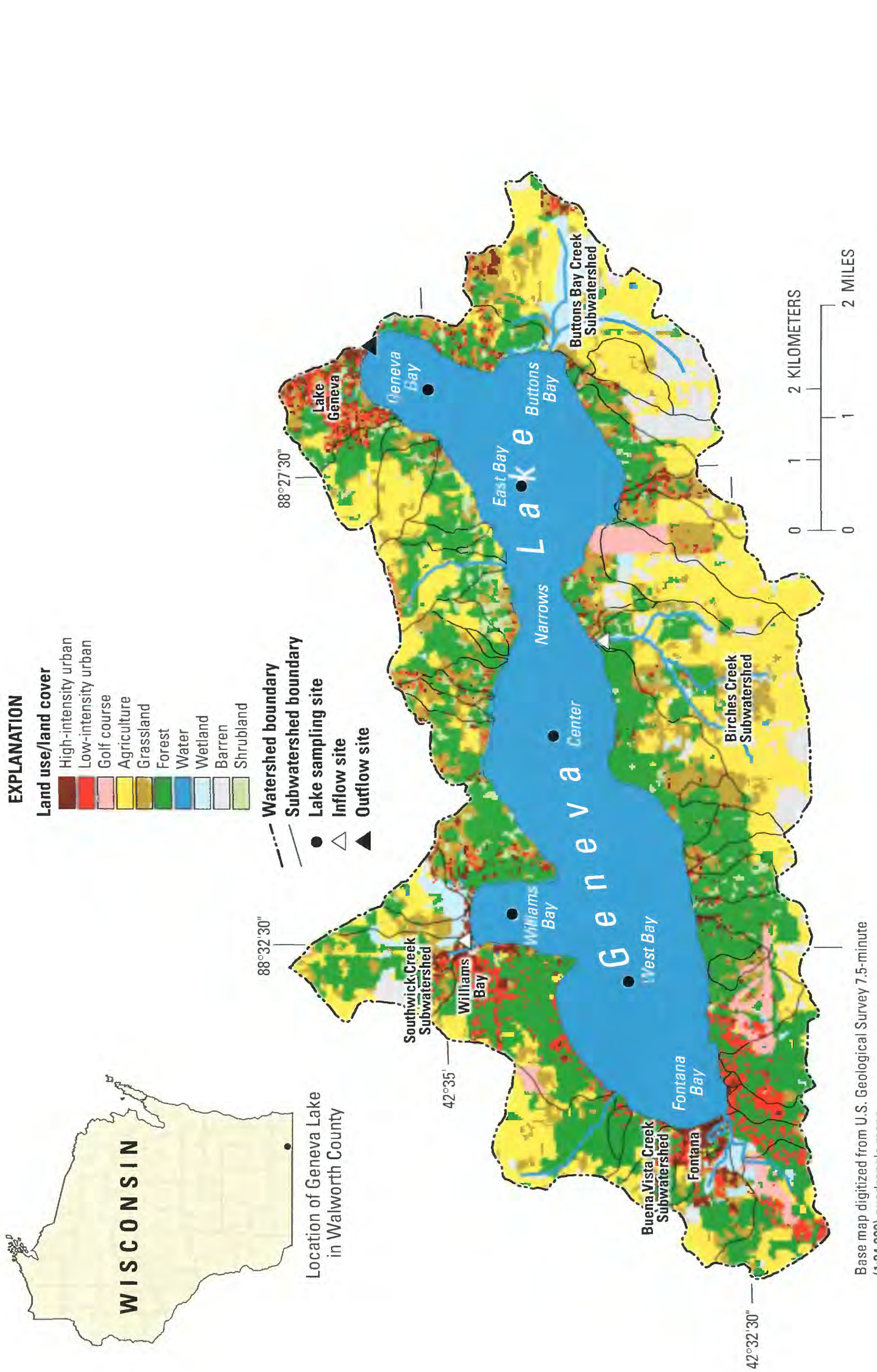


Figure 1. Data-collection sites in Geneva Lake, Wisconsin, and its watershed. (Land use/land cover from WISCLAND (Lillesand and others, 1998.)

prohibiting the discharge of pollutants to the lake (Southeastern Wisconsin Regional Planning Commission, 1985). In 1935, the Lake Association was formed. In 1946, the Linn Sanitary District was established to address sanitary issues within the Town of Linn (approximately 60 percent of the watershed). In 1971, a group of concerned citizens formed the Geneva Lake Environmental Agency (GLEA) to study the physical, chemical, and biological aspects of the lake and its watershed, examine resource related problems, and make recommendations to protect the lake. In 1981, the Geneva Lake Conservancy was formed to encourage conservation easements as a tool for protecting environmentally sensitive lands within the watershed.

To document the water quality of the lake prior to GLEA being established, various sampling efforts were done by scientists from the University of Wisconsin-Madison. Sporadic studies were done from 1901 to 1941 under the supervision of E. Birge and C. Juday (unpublished data, University of Wisconsin–Madison) and in 1972 by R. Stauffer (1974). More systematic studies were done by scientists from the Wisconsin Department of Natural Resources (WDNR) from 1958 through 1977 (R. Lillie, Wisconsin Department of Natural Resources, written commun., 2000).

Because of concern over degrading water quality in the lake, the Southeastern Wisconsin Regional Planning Commission (SEWRPC) and GLEA began a study in 1976 (Southeastern Wisconsin Regional Planning Commission, 1985) to document the water quality, describe and quantify the natural and anthropogenic factors influencing water quality, and develop a water-quality management plan to be used to improve the water quality of the lake. As part of the study, a phosphorus (P) budget for the lake was constructed for 1975 (a typical runoff year), which indicated that major controllable sources included effluent from the Fontana sewage-treatment plant, septic systems in the watershed, and various nonpoint agricultural and urban sources. Because of the study (Southeastern Wisconsin Regional Planning Commission, 1985), the Fontana sewage-treatment plant was discontinued in 1984, and its influent was diverted out of the watershed. Actions were also taken to improve many of the septic systems around the lake. From 1981 to 1995, GLEA maintained a basic water-quality monitoring program on the lake.

In an effort to seek funding to reduce nonpoint loading from the watershed, the WDNR sought to make Geneva Lake Watershed a Nonpoint Watershed Project. As part of this effort, the WDNR collected a year of detailed temperature, dissolved oxygen, and P data (S. Toshner, Wisconsin Department of Natural Resources, written commun., 1997).

As part of continuing efforts to improve the water quality of Geneva Lake, a collaborative effort between the U.S. Geological Survey (USGS), WDNR, and the GLEA was initiated in 1997 to thoroughly describe the present water quality of Geneva Lake and its sediments, and describe if and why changes in water quality have occurred in the lake over the past 170 years. This report presents the results of this collaborative study and will be used to update the management plan for the lake.

GENEVA LAKE AND ITS WATERSHED

Geneva Lake was formed about 10,000 years ago during the late Wisconsinan glaciation. The lake has a surface area of 2,117 ha, a mean depth of 19.1 m, a maximum depth of 44 m, and maximum length of about 12 km. The lake has only one basin with several bays: the East Bay separated from the West Bay by the Narrows, Williams Bay off of the West Bay, and Geneva Bay and Buttons Bay off of the East Bay (fig. 1). The areas and volumes of the entire lake as a function of lake depth are given in table 1.

The 53.1-km^2 watershed of the lake comprises various land uses: agriculture and grassland, 43 percent; forest and wetland, 37 percent; urban, 9 percent; and other uses, 12 percent (fig. 1). In recent years, the communities of Lake Geneva, Williams Bay, and Fontana have increased in size, and much of the area identified as forested land in figure 1 is actually low-density residential areas.

The watershed has loamy soils with moderate infiltration rates and moderately fine to moderately coarse textures (Southeastern Wisconsin Regional Planning Commission, 1985). The topography around the lake is steep with an average soil slope of 4.6 percent. Most of the areas in the watershed with gentler slopes are used for agriculture (fig. 1).

With a watershed area of 53.1 km^2 and lake surface area of 21.2 km^2, Geneva Lake has a low watershed-to-lake area ratio of about 2.5:1. Therefore, the water in the lake has a relatively long residence time of about 15 years, and the water quality of the lake is expected to respond relatively slowly to changes in the watershed.

Table 1. Morphometry of Geneva Lake, Wisconsin

Depth from the surface (meters)	Area (thousands of square meters)	Volume below specified depth (thousands of cubic meters)
0	21,200	403,000
1	20,900	382,000
2	20,300	361,000
3	19,300	342,000
4	18,300	323,000
5	17,400	305,000
6	16,700	288,000
7	16,200	271,000
8	15,500	256,000
9	15,100	240,000
10	14,700	225,000
11	14,300	211,000
12	14,000	197,000
13	13,600	183,000
14	13,200	169,000
15	12,900	156,000
16	12,500	144,000
17	12,000	132,000
18	11,500	120,000
19	10,900	109,000
20	10,300	98,000
21	9,720	88,000
22	9,060	78,600
23	8,630	69,700
24	8,240	61,300
25	7,680	53,300
26	7,240	45,900
27	6,790	38,900
28	6,280	32,300
29	5,700	26,300
30	5,030	21,000
31	4,330	16,300
32	3,720	12,300
33	3,040	8,890
34	2,290	6,230
35	1,790	4,190
36	1,340	2,620
37	896	1,500
38	560	774
39	323	333
40	153	95
41	32	2
44	0	0

METHODS

Lake Sampling

Water-quality monitoring in Geneva Lake by the USGS began in April 1997 and has continued to the present (2002); however, in this report we concentrate on data collected from April 1997 to August 2000 (table 2). The spatial and temporal intensity of the monitoring changed during this period in the process of designing a long-term monitoring program that could be used to describe most changes in water quality and yet would be financially feasible for the community. During the first year (April 1997 through March 1998), sampling was done monthly at five sites (the center of West Bay at 44 m, just west of the Narrows (referred to as the "Center" site) at 28 m, East Bay at 23 m, Williams Bay at 18 m, and Geneva Bay at 15 m; fig. 1). During the second year (April 1998 through March 1999), sampling was done every 2 weeks from April through October and monthly for the remainder of the year at three locations (West Bay, Center, and East Bay). During the third year (April 1999 through May 2000), sampling was done monthly at the West Bay and East Bay. Since May of 2000, sampling has continued only in the West Bay.

Table 2. Summary of sampling efforts done in Geneva Lake, Wisconsin, and its tributaries by the U.S. Geological Survey and Wisconsin Department of Natural Resources

Sampling effort	Dates
Lake Sampling	April, 1997 to August, 2000
Spacial analysis (five sites)	April, 1997 to March, 1998
Temporal analysis (three sites)	April, 1998 to March, 1999
Routine sampling (two sites)	April, 1998 to May, 2000
Routine sampling (one site)	June, 2000 to August, 2000
Tributary Sampling	
Southwick and Birches Creeks	Oct. 1, 1997 to Sept. 30, 1999
Synoptic sampling of tributaries	
Base flow	Nov. 8, 1997 and July 19, 2000
High flow	May 13, 1998
Sediment Sampling	
Sediment-core collection	October, 1995
Surficial-sediment collection	October, 1996 and July, 1997

During each sampling, profiles of water temperature, dissolved oxygen, specific conductance, and pH were collected with a multiparameter meter, and Secchi depths were measured with a standard black and white Secchi disk. At all locations, water samples were collected near the surface (0.5 m) and bottom of the water column (approximately 1 m above the bottom) at times when the water column was well mixed. During periods when stratification was present, water samples were collected near the top and bottom of the epilimnion and top and bottom of the hypolimnion. At the shallower sites (Williams Bay and Geneva Bay), only one hypolimnion sample was collected near the top of the hypolimnion. An additional sample was collected in the middle of the hypolimnion in the West Bay. All samples were collected with a Van Dorn sampler and analyzed for total and dissolved ortho-P, dissolved nitrite plus nitrate (referred to as just nitrate, because nitrite represents only a very small fraction of the total value), dissolved ammonia, and Kjeldahl nitrogen (N). In addition, surface samples were analyzed for chlorophyll *a*. During open-water periods, samples were collected for identification of zooplankton (April 1997–December 1999) and phytoplankton (April 1997–September 1999) at the three deeper stations. (Only plankton data from the West Bay are presented in this report because of the similarity found among sites.) Phytoplankton samples were composited from samples collected at the surface and at depths of 1, 2, and 3 m. Zooplankton samples were collected by vertically towing a closing Wisconsin net (153-µm mesh) through the epilimnion and through the hypolimnion of the lake. Only one zooplankton sample from the entire water column was collected when the lake was well mixed. During spring overturn, the surface and bottom water samples also were analyzed for color, turbidity, hardness, alkalinity, solids, calcium, chloride, iron, magnesium, manganese, potassium, silica, sodium, and sulfate. (These data were published in annual USGS reports (U.S. Geological Survey, Wisconsin District Lake-Studies Team, 1998, 1999, 2000, 2001) and are not presented in this report). All dissolved constituents were passed through a 0.45-µm-pore-diameter cellulose membrane filter in the field. Chlorophyll *a* samples were filtered through a 5.0-µm-pore-diameter standard method (SM) filter.

Historically, GLEA measured chlorophyll *a* in water samples composited from the surface, and at 1, 2, and 3 m. Therefore, to determine whether these data were comparable to data collected by the USGS, a comparability test was done in April through September 1997. During this test, additional upper epilimnion samples (surface, 1m, 2m, and 3m) were collected in the West Bay with a Van Dorn sampler, composited using a churn splitter, and analyzed for chlorophyll *a*. A paired Student t-test was used to determine whether differences in chlorophyll *a* concentrations between the two methods of sample collection were significant.

All chemical analyses of water samples were done by the Wisconsin State Laboratory of Hygiene in accordance with standard analytical procedures described in the "Manual of Analytical Methods, Inorganic Chemistry Unit" (Wisconsin State Laboratory of Hygiene, 1993). Phytoplankton and zooplankton samples were identified to family and species by a private laboratory (Aquatic Analysts, Wilsonville, Oregon).

One method of classifying the water quality of a lake is by computing water-quality or Trophic State Index (TSI) values based on surface total P and chlorophyll *a* concentrations, and Secchi depths, as developed by Carlson (1977) and modified for Wisconsin lakes by Lillie and others (1993). TSI values based on P concentrations (TSI_P), chlorophyll *a* concentrations (TSI_C), and Secchi depths (TSI_{SD}) were computed for each sampling by means of equations 1–3 and were used to compute summer average TSI values:

$$TSI_P = 28.24 + 17.81\ [\log_{10} \text{P (in micrograms per liter)}] \qquad (1)$$

$$TSI_C = 34.82 + 17.41\ [\log_{10} \text{chlorophyll (in micrograms per liter)}] \qquad (2)$$

$$TSI_{SD} = 60.0 - 33.2\ [\log_{10} \text{Secchi depth (in meters)}] \qquad (3)$$

Tributary Sampling

To describe the flow, water quality, and nutrient loading from the watershed, two tributaries to the lake (Southwick Creek at Williams Bay and Birches Creek at Lackey Lane) were examined (fig. 1). These two tributaries were chosen because they were thought to best represent the conditions in the watershed: Southwick Creek has a subwatershed with mixed land use, and Birches Creek (the largest subwatershed) has a subwatershed dominated by agriculture. To describe the flow, water quality, nutrients, and sediments leaving the lake, the White River (lake outlet) was sampled just downstream of the lake. Water levels at these three sites were measured continuously (every 5 minutes at the White River and every 15 minutes at Birches and Southwick Creeks) and used to estimate flow by use of stage-discharge relations. These data then were used to compute daily average flows for each site for water year (WY) 1998 (10/1/97–9/30/98) and WY1999 (10/1/98–9/30/99) (table 2). The daily average data were published in USGS data reports (Holmstrom and others, 1999, 2000). These data were used to estimate the total flow to and from the lake, described in the section "Hydrologic Budget" of Geneva Lake.

Water samples were collected from each of these three sites and analyzed for P and suspended sediment (SS) concentrations. Storm-runoff samples were collected at the tributaries by means of stage-activated, refrigerated, automatic samplers. A few additional samples were collected manually by use of the equal-width-increment (EWI) method described by Edwards and Glysson (1999). Some EWI samples, collected at times when automated samples also were collected, were used to evaluate the representativeness of the automated samples. EWI and grab samples also were collected at low flow at these sites to estimate loading during these conditions. At the outlet, EWI and grab samples were collected approximately monthly, plus during about 12 storm-runoff events per year. Daily P and SS loads then were computed for each site for WY1998 and WY1999 by use of the integration method described by Porterfield (1972). All water-quality data and computed loads (except those estimated from ungaged areas) were published in annual USGS data reports (Holmstrom and others, 1999, 2000) and stored in the National Water Information System database (U.S. Geological Survey, 1998).

Flow was measured and EWI water samples were collected (and analyzed for P) at 24 additional sites to describe the variability in the water quality in the other tributaries and to refine estimates of the total water and P loading to the lake. Most sites were sampled during two base-flow periods (November 8, 1997, and July 19, 2000) and one high-flow event (13 May, 1998). One of these sites (Buttons Bay Creek, the stream with the second largest subwatershed in the study area) was sampled more often (17 times during the 2-year period), but not as frequently as the two intensive sites.

All tributary water samples were analyzed for total P concentration by the Wisconsin State Laboratory of Hygiene (Wisconsin State Laboratory of Hygiene, 1993) and for SS concentration by the USGS Iowa Sediment Laboratory in accordance with standard analytical procedures described by Fishman and Friedman (1988).

Lake-Sediment Sampling

Surficial Sediments

Surficial-sediment samples were collected from a depth of about 40 m in the deep hole of the West Bay in October 1996 and about 20 m in Williams Bay and Geneva Bay in October 1996 and in July 1997 (fig. 1; table 2). The two bay locations were chosen because they were thought to represent sites with the most potential for observing accumulated contamination. The samples, which consisted of material from approximately the top 5-cm layer of sediments, were collected by means of a gravity corer in 1996 and an Ekman dredge in 1997. The core samples from 1996 were cut into 1-cm-thick slices and sent to the Wisconsin State Laboratory of Hygiene for determination of sedimentation rates, water content, organic matter, and calcium carbonate, as well as concentrations of aluminum, arsenic, N, P, and zinc. In addition to these constituents, concentrations of calcium, copper, iron, manganese, and potassium were determined in samples from the West Bay. The concentrations from the top five 1-cm depths in the 1996 sample were averaged to obtain an average concentration in the surficial deposits. Analytical methods for these constituents followed approved protocols of the U.S. Environmental Protection Agency (1983). The samples from 1997 were chilled in ice and sent to the USGS National Water-Quality Laboratory for determination of total organic carbon, polycyclic aromatic hydrocarbons (PAHs), chlorinated hydrocarbons (including DDT and its derivatives), polychlorinated biphenyls (PCBs), and some trace metals. Analytical methods for these constituents followed approved USGS protocols (Fishman and others, 1994).

Sediment-Core Analyses

Sediment cores were collected from Geneva Lake on October, 22 1995 by means of a gravity corer with a 6.5-cm plastic liner. Cores were extracted from the deep hole of the West Bay in about 40 m of water, at the mouth of Williams Bay in 29 m of water, and in Geneva Bay in 14 m of water. In the laboratory, the upper 40 to 50 cm of each core were sectioned into 1- or 2-cm slices.

Dry weight, organic matter, and calcium carbonate were determined by measuring weight loss after drying, ashing, or combusting. Sediment samples were sent to the Wisconsin State Laboratory of Hygiene for determination of total aluminum, calcium, potassium, iron, manganese, titanium, copper, zinc, arsenic, P, and N. For each core, sediment age and bulk sediment accumulation rates were determined for each sediment slice by means of the constant rate of supply model (Appleby and Oldfield, 1978). Diatom populations were counted in each sediment slice. Historical spring P concentrations were then inferred from the diatom community in each slice using a weighted-averaging technique of the various P optima of the measured diatom taxa. A more complete description of the methods used for sediment-core analysis is described in Garrison (2000).

DESIGN OF A LONG-TERM SAMPLING STRATEGY

To design a long-term sampling strategy that would describe most changes in water quality in the lake and yet would be financially feasible for the community, the spatial and temporal intensity of the monitoring was varied throughout the period of sampling. During the first year of the study (April 1997 to March 1998), five sites (West Bay, Center site, East Bay, Williams Bay, and Geneva Bay; fig. 1) were sampled to describe areal differences in water quality and to determine whether more than one sampling site was needed to describe the water quality in the lake. To demonstrate the differences or lack of differences, the water-quality measurements at the surface at all of the sites were compared with that measured in the West Bay (table 3).

In general, water-quality characteristics measured at the surface (water clarity, and chlorophyll *a* and nutrient concentrations) were horizontally uniform throughout the lake. The average differences in Secchi depths from that measured in the West Bay were 0.2 m or less; average differences in chlorophyll *a* concentrations were 0.85 μg/L or less, and differences in total P concentrations were 2 μg/L or less. In general, the median differences were even less than these values (table 3). None of these differences were statistically significant at $p < 0.1$. The differences in the N species were a little larger, with highest concentrations being measured in the West Bay, possibly caused by a relative absence of consumption of N in the pelagic areas of the lake. Concentrations of Kjeldahl N usually were lower in Williams Bay and the East Bay than in the West Bay (statistically less than in the West Bay at $p < 0.05$). Concentrations at the Center site and Geneva Bay were statistically less than in the West Bay ($p < 0.1$). Concentrations of nitrate usually were lower at the East Bay and Geneva Bay than at the West Bay, but concentrations were not statistically different among sites.

Although very little horizontal difference in temperature was found in the lake, consistent horizontal seasonal differences in oxygen were observed below the thermocline (fig. 2). Dissolved oxygen was more slowly consumed in the middle of the hypolimnion in the West Bay than in other locations, and dissolved oxygen was more quickly consumed just above the bottom sediments in shallower bays than at similar depths in other deeper locations. Nutrient concentrations also were slightly higher just above the bottom in some shallower areas of the lake, especially in areas where the dissolved oxygen concentrations were low. For example, concentrations of P at 28 m at the Center site in August of 1997 (96 μg/L) were higher than those measured at that depth in the West Bay (3 μg/L). These differences just above the sediments were observed only in late summer.

During the first summer of the study (April–September 1997), the method of collecting samples for chlorophyll *a* analysis also was evaluated. Samples composited from the surface, and at depths of 1, 2, and 3 m (GLEA technique) were compared with just near-surface samples. The average concentrations from composited and near-surface samples (based on six monthly samples collected from April through September) were 4.5 and 4.6 μg/L, respectively, and the maximum difference in concentrations was only 0.6 μg/L. These differences were not statistically significant, even at $p < 0.2$.

Results from the first year of the study showed very little horizontal variation in water quality throughout the lake. The differences that were found probably were caused by the physical differences in water depth and possibly local inputs of nutrients or lack of consumption of nutrients. It was felt that very little additional information was obtained by sampling in Geneva Bay and Williams Bay; therefore, sampling at these two sites was discontinued after the first year of study. In addition, chlorophyll *a* concentrations subsequently were measured by use of near-surface samples only.

During the second year of the study (April 1998–March 1999), sampling was done every 2 weeks from April through October and monthly for the remainder of the year at three sites (West Bay, Center site, and East Bay) to determine whether increasing the sampling frequency improved the description of changes in water quality or provided statistically significant differences in average seasonal values. (Only the samples from the West Bay are described here; however, similar results were found for the Center site and East Bay.) A comparison of the thermal structure of the West Bay for 1998 based on all of the data collected and based just on monthly samples is shown in figure 3. In general, the increased number of temperature and dissolved oxygen profiles only slightly improved the description of the onset and breakdown of stratification. Additional sam-

Table 3. Spatial variability in water quality in Geneva Lake, Wisconsin, April 1997–March 1998

[μg/L, micrograms per liter]

Month	West	Center	East	Geneva Bay	Williams Bay	Difference from that measured in the West Bay			
						Center	East	Geneva Bay	Williams Bay
				Secchi depth (meters)					
April	3.0	3.2	2.5	2.8	2.9	0.2	-0.5	-0.2	-0.1
May	3.5	3.0	2.9	2.6	3.3	-.5	-.6	-.9	-.2
June	4.3	4.4	4.0	3.4	3.7	.1	-.3	-.9	-.6
July	2.7	2.4	3.0	3.0	3.7	-.3	.3	.3	1.0
August	3.0	3.2	3.4	3.8	2.9	.2	.4	.8	-.1
September	3.4	2.7	3.4	3.8	2.7	-.7	.0	.4	-.7
October	4.9	5.6	5.8	5.9	4.6	.7	.9	1.0	-.3
November	4.4	4.7	5.0	4.7	4.4	.3	.6	.3	.0
March	5.0	5.6	4.4	6.1	5.8	.6	-.6	1.1	.8
Average	**3.8**	**3.9**	**3.8**	**4.0**	**3.8**	**.1**	**.0**	**.2**	**.0**
Median	**3.5**	**3.2**	**3.4**	**3.8**	**3.7**	**.2**	**.0**	**.3**	**-.1**
				Chlorophyll *a* (μg/L)					
April	5.29	5.17	5.16	5.12	4.60	-.12	-.13	-.17	-.69
May	6.20	6.90	7.00	7.70	5.70	.70	.80	1.50	-.50
June	.98	1.19	1.17	1.19	1.20	.21	.19	.21	.22
July	4.85	4.77	5.26	4.39	4.71	-.08	.41	-.46	-.14
August	5.61	5.89	4.02	3.51	5.00	.28	-1.59	-2.10	-.61
September	4.61	3.47	2.72	1.80	2.93	-1.14	-1.89	-2.81	-1.68
October	5.57	4.64	3.35	2.58	3.18	-.93	-2.22	-2.99	-2.39
November	3.83	3.78	3.99	3.03	4.01	-.05	.16	-.80	.18
March	3.00	3.00	3.00	3.00	3.00	.00	.00	.00	.00
Average	**4.44**	**4.31**	**3.96**	**3.59**	**3.81**	**-.13**	**-.47**	**-.85**	**-.62**
Median	**4.85**	**4.64**	**3.99**	**3.03**	**4.01**	**-.05**	**.00**	**-.46**	**-.50**
				Surface total phosphorus (μg/L)					
April	16	16	17	16	15	0	1	0	-1
May	16	13	23	12	14	-3	7	-4	-2
June	5	7	3	3	8	2	-2	-2	3
July	14	13	10	10	15	-1	-4	-4	1
August	3	3	3	11	14	0	0	8	11
September	6	3	3	3	3	-3	-3	-3	-3
October	7	8	8	9	18	1	1	2	11
November	5	5	3	3	9	0	-2	-2	4
March	7	6	7	3	3	-1	0	-4	-4
Average	**9**	**8**	**9**	**8**	**11**	**1**	**0**	**-1**	**2**
Median	**7**	**7**	**7**	**9**	**14**	**0**	**0**	**-2**	**1**

Table 3. Spatial variability in water quality in Geneva Lake, Wisconsin, April 1997–March 1998—Continued

[µg/L, micrograms per liter]

Month	West	Center	East	Geneva Bay	Williams Bay	Difference from that measured in the West Bay			
						Center	East	Geneva Bay	Williams Bay
				Kjeldahl nitrogen (µg/L)					
April	600	500	400	400	400	-100	-200	-200	-200
May	400	400	400	400	400	0	0	0	0
June	470	640	480	540	450	170	10	70	-20
July	500	490	400	400	300	-10	-100	-100	-200
August	800	300	400	400	400	-500	-400	-400	-400
September	500	400	400	600	400	-100	-100	100	-100
October	500	500	400	400	200	0	-100	-100	-300
November	400	300	200	400	600	-100	-200	0	200
March	500	400	400	500	400	-100	-100	0	-100
Average	**519**	**437**	**387**	**449**	**394**	**-82**	**-132**	**-70**	**-124**
Median	**500**	**400**	**400**	**400**	**400**	**-100**	**-100**	**0**	**-100**
				Nitrate (µg/L)					
April	53	64	31	36	58	11	-22	-17	5
May	49	43	22	22	40	-6	-27	-27	-9
June	17	5	5	5	5	-12	-12	-12	-12
July	5	5	5	5	5	0	0	0	0
August	5	5	5	5	5	0	0	0	0
September	5	5	5	5	5	0	0	0	0
October	5	5	5	5	12	0	0	0	7
November	48	37	28	29	44	-11	-20	-19	-4
March	84	57	50	51	60	-27	-34	-33	-24
Average	**30**	**25**	**17**	**18**	**26**	**-5**	**-13**	**-12**	**-4**
Median	**17**	**5**	**5**	**5**	**12**	**0**	**-12**	**-12**	**0**

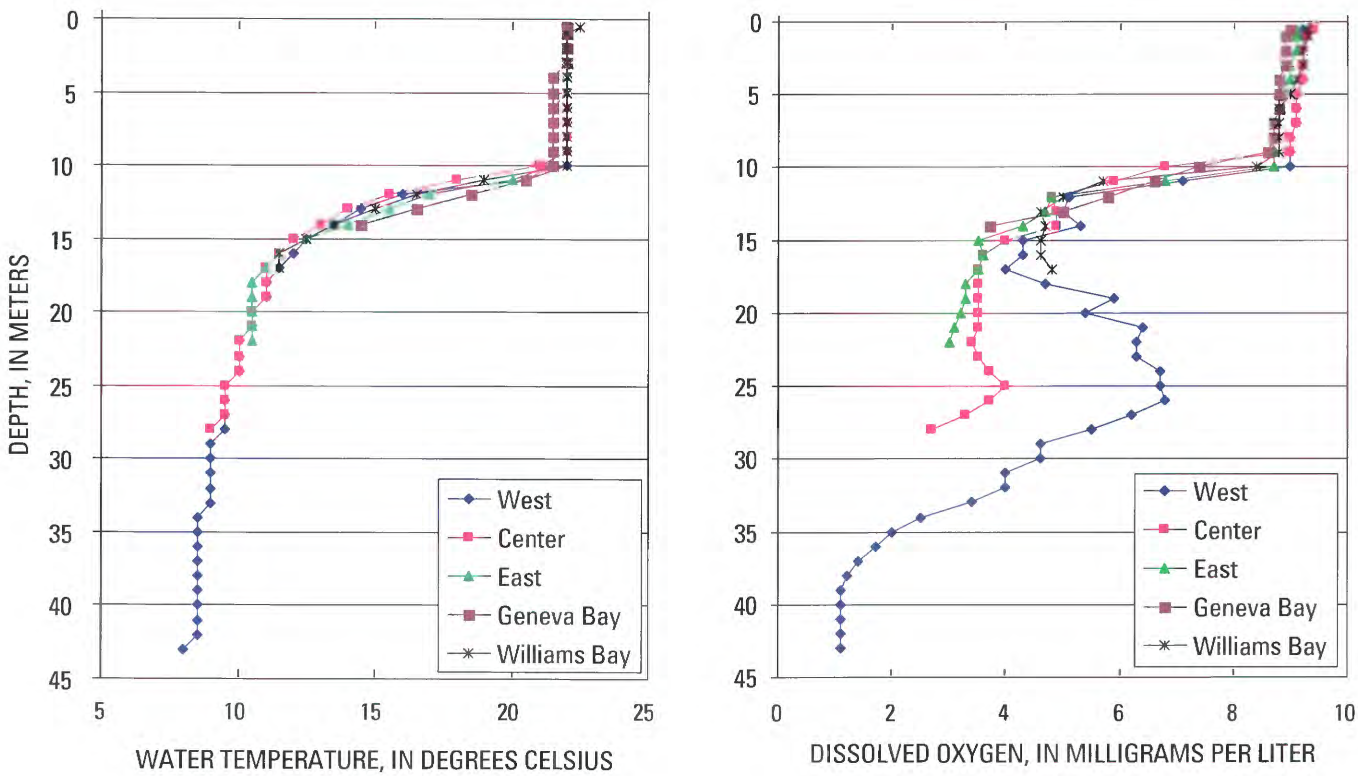


Figure 2. Temperature and dissolved oxygen profiles throughout Geneva Lake, Wisconsin, August 19, 1997.

pling added very little information describing water temperatures any depth or the depth of stratification during summer.

The data from the West Bay were evaluated to determine whether the additional data collected every 2 weeks provided different seasonal average water-quality values from those computed using just monthly data (table 4). The difference in average Secchi from that using monthly data was 0.5 m, difference in near-surface chlorophyll *a* concentrations was about 0.2 µg/L, and difference in P concentrations was less than 0.1 µg/L. The differences in N species concentrations were somewhat larger. The difference in average Kjeldahl N concentrations was 48 µg/L and difference in nitrate concentrations was about 6 µg/L. The difference in bottom P concentrations was about 4 µg/L, difference in bottom Kjeldahl N concentrations was 74 µg/L, and difference in bottom nitrate concentrations was 26 µg/L. None of these differences were statistically significant at $p < 0.35$. On the basis of these results, it was felt that very little additional information was obtained by sampling every 2 weeks, except for possibly during the onset and breakdown of stratification. Therefore, sampling every 2 weeks was discontinued except during late October. An additional sample in late October would provide a better description of the timing of the breakdown in stratification and possibly a better description of the highest concentrations of nutrients above the bottom sediments.

During the first 2 years of the study, only a limited amount of additional information was gained by sampling in the Center site; therefore, this site near the Narrows was discontinued after the second year of the study.

A long-term strategy for sampling Geneva Lake that minimizes sampling effort, but is adequate to define the changes in water quality in the lake, consists of monthly sampling (except for October) in the center of the West and East Bays. During October, twice-per-month sampling would allow better definition of when stratification breaks down. Discontinuing sampling of the East Bay also may be considered if there are financial constraints, because the only significant differences from the West Bay were found in N concentrations and dissolved oxygen concentrations below the thermocline.

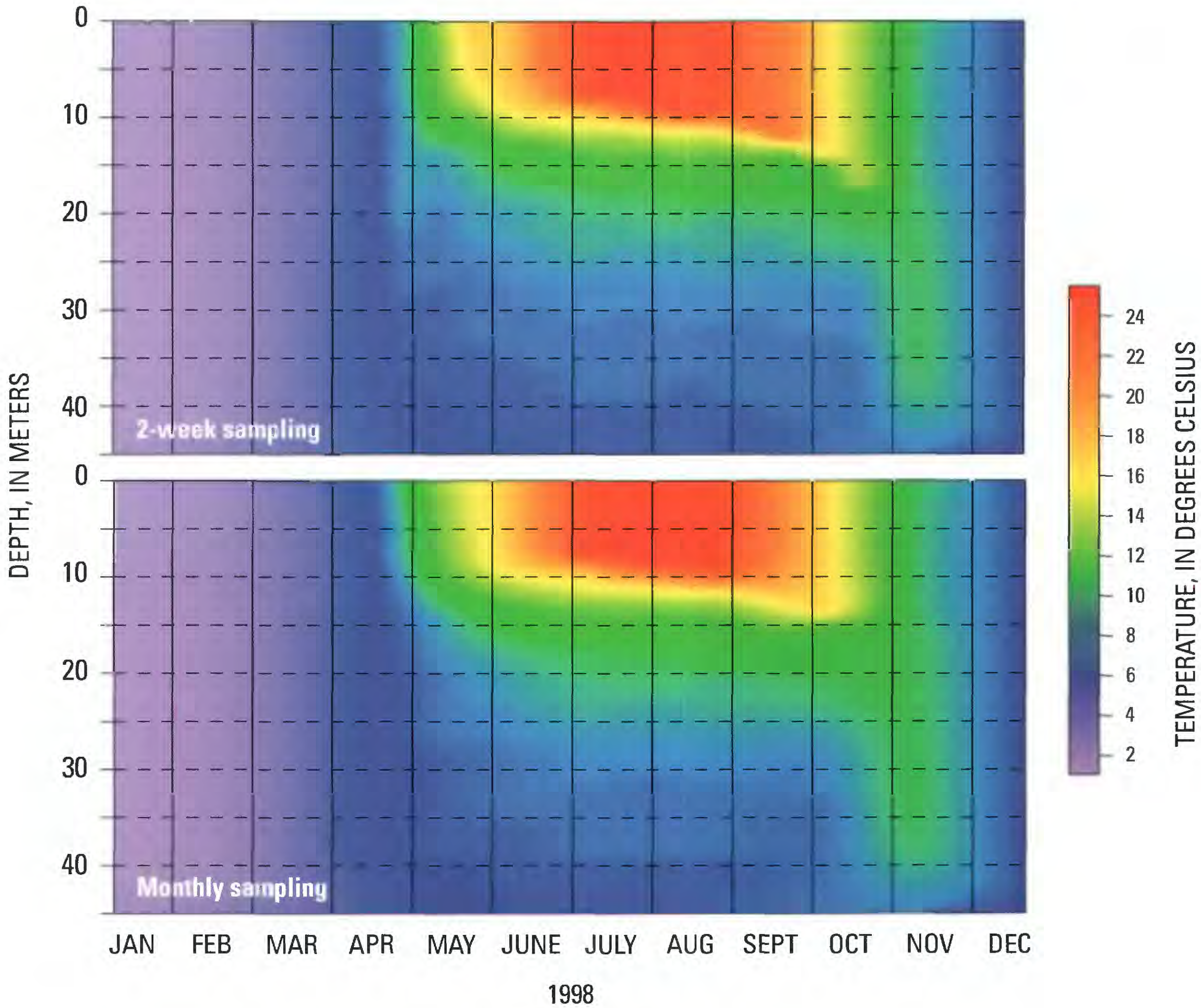


Figure 3. Water temperature distribution in Geneva Lake, Wisconsin, in 1998 based on 2-week and monthly sampling.

CLIMATIC CONDITIONS DURING THE STUDY PERIOD

The climatic conditions during the study period from 1997 through 2000 generally were higher than normal temperatures and about normal precipitation based on data from a National Weather Service station in Lake Geneva, Wis. (table 5 and fig. 4). Average annual air temperatures ranged from 8.5°C in 1997 to 11.1°C in 1998 compared to the normal for the period during 1961-90 of 8.9°C. Air temperatures were normal or slightly above normal during the summer (June–August); the warmest summer (23.2°C in 1999) was about 1.4°C above normal; however, air temperatures were above normal during the winter (January–March), ranging from 0.7°C above normal in 1997 to 4.5°C above normal in 1998.

Annual precipitation ranged from 826 mm in WY1997 to 1,072 mm in WY1999 compared to the normal precipitation of 938 mm (table 5). During the years in which the hydrologic budget was computed, annual precipitation was 885 mm in WY1998 and 1,072 in WY1999. During both years, precipitation was much above normal in May and June, especially in 1999 when precipitation was 173 mm above normal for these months (fig. 4).

WATER QUALITY OF GENEVA LAKE

Water quality throughout Geneva Lake has been shown to be relatively uniform horizontally; therefore, the thermal structure and water quality of the lake are described on the basis of measurements from the West Bay.

Table 4. Differences in water quality between 2-week and monthly sampling in the West Bay of Geneva Lake, Wisconsin, April–October 1998
[µg/L, micrograms per liter; NM, not measured]

		Surface					Bottom		
Month	Sample number	Secchi depth (meters)	Chlorophyll *a* (µg/L)	Total phosphorus (µg/L)	Kjeldahl nitrogen (µg/L)	Nitrate, nitrogen (µg/L)	Total phosphorus (µg/L)	Kjeldahl nitrogen (µg/L)	Nitrate, nitrogen (µg/L)
2-week sampling									
April	1	3.4	4.18	10	510	74	12	540	73
April	2	2.1	7.45	10	430	5	11	380	73
May	1	1.7	6.60	10	480	5	16	490	63
May	2	2.4	3.62	10	680	5	13	1,230	61
June	1	3.0	5.52	12	540	5	22	680	62
June	2	6.9	1.47	7	530	12	22	650	5
July	1	2.8	3.38	10	1,510	5	60	740	205
July	2	3.5	2.92	3	650	5	39	480	309
August	1	3.8	2.85	7	350	5	63	530	332
August	2	3.4	2.14	10	620	5	101	880	5
September	1	2.9	2.54	9	530	5	106	1,040	5
September	2	4.6	4.94	13	510	5	106	1.150	5
September	3	3.7	5.85	11	550	5	92	990	5
October	1	4.1	NM	5	570	25	61	1,100	5
October	2	5.5	1.97	9	440	5	67	1,200	5
Average		**3.6**	**3.96**	**9**	**593**	**11**	**53**	**805**	**81**
Monthly sampling									
April	1	3.4	4.18	10	510	74	12	540	73
May	1	1.7	6.60	10	480	5	16	490	63
June	1	3.0	5.52	12	540	5	22	680	62
July	1	2.8	3.38	10	1,510	5	60	740	205
August	1	3.8	2.85	7	350	5	63	530	332
September	1	2.9	2.54	9	530	5	106	1,040	5
October	1	4.1	NM	5	570	25	61	1,100	5
Average		**3.1**	**4.18**	**9**	**641**	**18**	**49**	**731**	**106**
Difference between average values		**-.5**	**.22**	**0**	**48**	**6**	**-4**	**-74**	**26**
p value for test of difference		**.38**	**.80**	**.95**	**.74**	**.50**	**.80**	**.57**	**.63**

Table 5. Climatic conditions at Lake Geneva, Wisconsin during calendar year 1975 and water years 1997–2000

[°C, degrees Celsius; mm, millimeter; --, not applicable]

	Normal[1]	1975	1997	1998	1999	2000
			Air temperature, in °C			
Annual	8.9	9.4	8.5	11.1	10.9	10.7
Summer[2]	21.8	22.1	21.5	22.6	23.2	21.4
Winter[3]	-3.3	-3.3	-2.6	1.2	-1.3	.9
			Departure from normal air temperature, in °C			
Annual	--	0.5	-0.4	2.2	2.0	1.8
Summer	--	.3	-.3	.8	1.4	-.3
Winter	--	.0	.7	4.5	2.0	4.2
			Normal precipitation, in mm			
Annual	938	915	826	885	1,072	1,017
Summer	307	388	270	269	311	371
Winter	154	196	153	210	205	115
			Departure from normal precipitation, in mm			
Annual	--	-23	-111	-53	135	80
Summer	--	81	-37	-38	3	64
Winter	--	42	-2	56	51	-40

[1]Based on the period 1961–90.

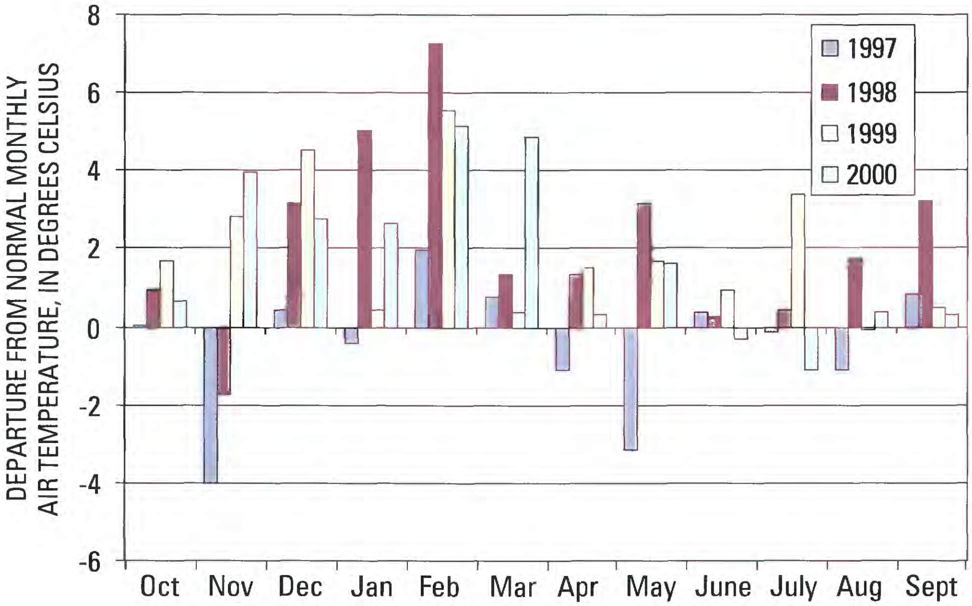
[2]June–August.

[3]January–March.

Water Temperature and Ice Cover

Geneva Lake is a dimictic lake that strongly stratifies during summer and usually freezes during winter. The lake gradually freezes completely, over several days to a month; ice formation begins in the bays and gradually spreads out to the middle of the West Bay. Usually, the center of the West Bay freezes over completely in 1 or 2 days. The ice cover on the lake usually breaks up very quickly; over just a 2–3 days. The lake typically is completely frozen over from early January through mid-March. However, in the winter of 1997–98, freeze-over was incomplete for the first time in the 139-year record, dating back to 1862, and in the winter of 1998–99, the lake was completely frozen for only about 29 days. Average freeze dates have not changed dramatically during the entire period. The average freeze date for the entire period of record (1862–present, appendix) was January 1, that for the last 20 years was January 4, and that for the past 5 years (1996–97 to 2000–2001, not including the year not freezing) was January 8. Average breakup dates have changed dramatically. The average breakup date for the entire period of record was March 30; however, that for the last 20 years was March 20, and that for the past 5 years (not including the year not freezing) was March 10. The average duration of ice cover for the entire period of record was about 90 days; however, for the past 5 years, the average number of days with ice cover was only 49 days. Two of the shortest periods of ice cover during the entire 139-year record took place during the study period.

After the breakup of ice cover, thermal stratification generally begins to develop in mid-May and is fully developed in early June (fig. 5). The average depth of the thermocline is fairly stable or deepens slightly from June through August. The average thermocline depth ranged from about 10 to 10.5 m (fig. 5 and table 6). From 1997 to 1999, the average surface temperature was about 23°C (ranging from 22.5°C in 1997 to 23.2°C in 1999), very similar to summer air temperatures. The average temperature of the deeper part of the hypolimnion was about 7°C (ranging from 6.5°C in 1998 to 8.5°C in 1997; fig. 5 and table 7). Temperatures throughout the hypolimnion are determined when the lake stratifies, and hypolimnetic water temperatures change very little as summer progresses. Thermal stratification generally begins to break down and the thermocline deepens as cooler fall air temperatures cause the thermocline to gradually erode. Generally, stratification does not completely deteriorate until late November to early December.



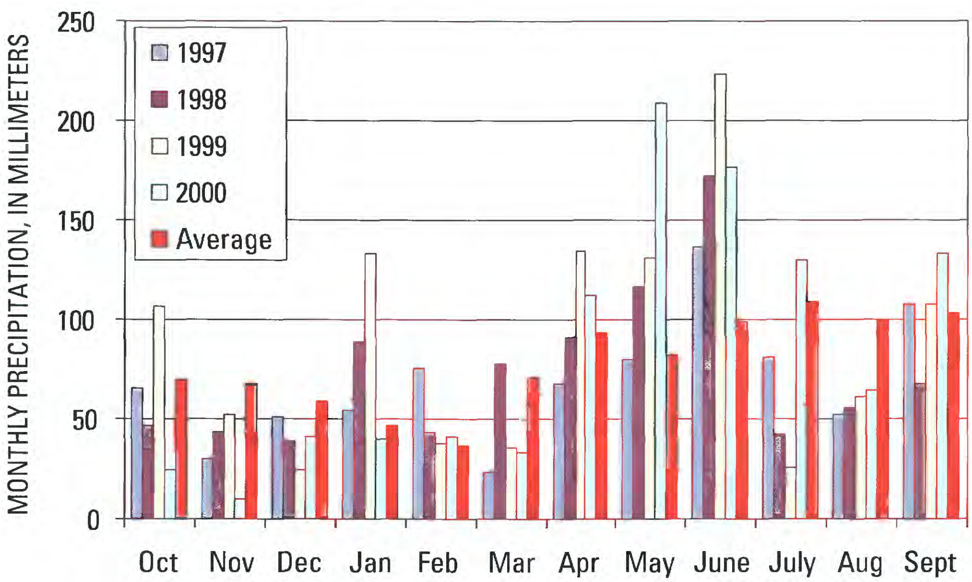


Figure 4. Monthly air temperature (departure from 30-year normal, 1961–90) and precipitation for Lake Geneva, Wisconsin, water years 1997–2000.

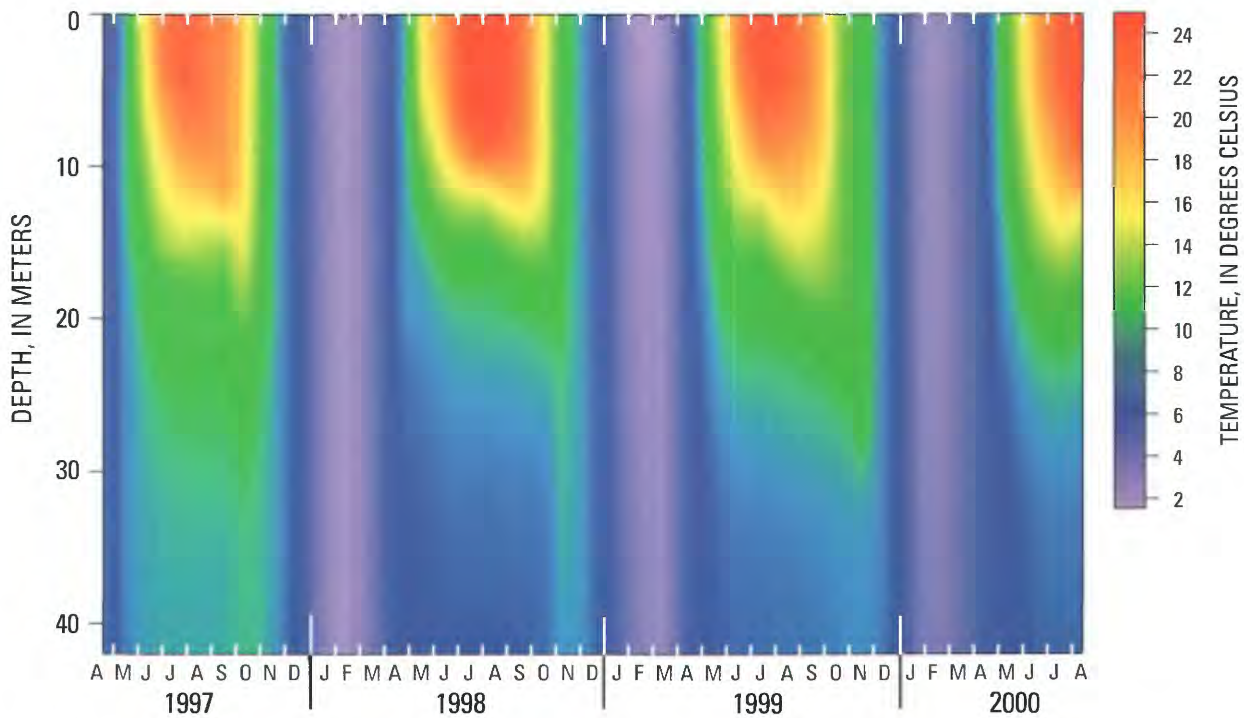


Figure 5. Water temperature distribution from April 1997 through August 2000, Geneva Lake, Wisconsin.

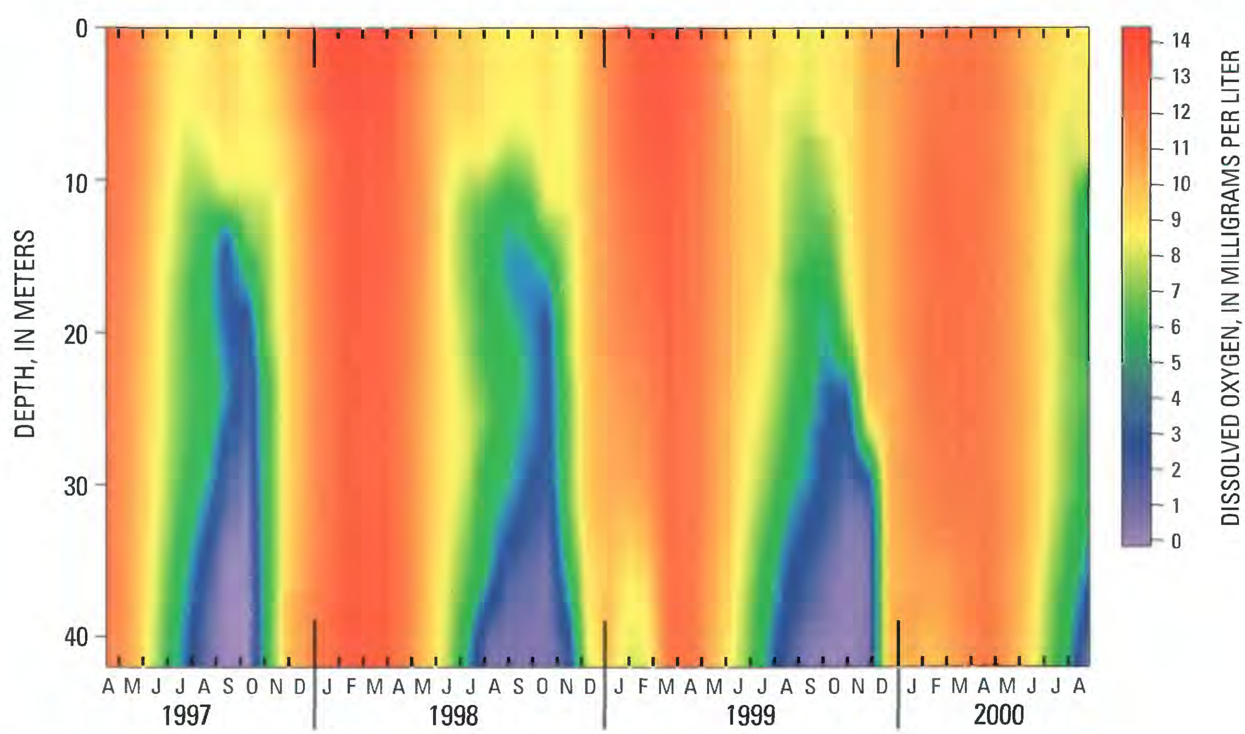


Figure 6. Dissolved oxygen distribution from April 1997 through August 2000, Geneva Lake, Wisconsin.

Table 6. Seasonal summaries of near-surface water quality in the West Bay of Geneva Lake, Wisconsin, 1997–2000

[P, phosphorus; N, nitrogen; TSI_P, TSI_C, and TSI_{SD}, trophic state index values based on phosphorus, chlorophyll *a* and secchi depth, respectively; °C, degrees Celsius; mg/L, milligrams per liter; m, meter; µg/L, micrograms per liter; --, not applicable]

		Temperature (°C)	Dissolved oxygen (mg/L)	Thermocline depth (m)	Secchi depth (m)	Total P (µg/L)	Dissolved P (µg/L)	Percent dissolved P	Kjeldahl N (µg/L)	Nitrate, N (µg/L)	Total N (µg/L)	Percent nitrate	N:P ratio	Chlorophyll *a* (µg/L)	Secchi depth (m)	TSI_P	TSI_C	TSI_{SD}
Spring turnover	1997	5.0	13.2	--	3.0	16	3.0	19	600	53	653	8	41	5.3	3.0	50	47	44
	1998	5.8	13.9	--	3.4	10	1.5	15	435	40	475	8	47	5.8	2.8	46	48	46
	1999	6.6	13.4	--	6.7	6	1.0	17	450	83	533	16	89	1.3	6.7	42	37	33
	2000	5.2	12.2	--	5.6	13	1.0	8	920	70	990	7	76	2.2	5.6	48	41	35
Summer	1997	22.5	9.3	10.7	3.3	7	1.3	23	557	7	563	1	131	3.8	3.3	42	43	43
	1998	23.1	9.3	10.1	3.9	8	2.0	32	700	6	706	1	101	3.0	3.9	44	43	41
	1999	23.2	9.0	9.7	4.9	6	2.3	46	540	27	567	4	107	1.7	4.9	41	37	37
	2000	21.9	9.1	10.7	4.3	12	1.3	12	430	22	458	5	40	1.7	4.3	47	39	39
Year average	1997	--	--	--	3.7	9	1.8	25	509	23	531	4	89	4.6	3.7	44	46	42
	1998	--	--	--	4.2	9	1.9	25	544	29	573	6	74	3.5	4.2	45	43	40
	1999	--	--	--	5.7	7	1.7	29	482	47	529	9	91	1.9	5.7	42	38	35
	2000	--	--	--	5.6	11	1.5	15	520	37	580	5	51	1.5	5.6	47	37	36
Study period annual average		**--**	**--**	**--**	**4.8**	**8.6**	**1.8**	**24**	**513**	**35**	**551**	**7**	**78**	**2.9**	**4.8**	**44**	**41**	**38**
Study period summer average		**22.7**	**9.2**	**10.3**	**4.1**	**8.3**	**1.8**	**28**	**557**	**15**	**574**	**3**	**95**	**2.6**	**4.1**	**44**	**40**	**40**

Table 7. Seasonal summaries of near-bottom water quality in the West Bay, Center, and East Bay of Geneva Lake, Wisconsin,1997–2000

[°C, degrees Celsius; mg/L, milligrams per liter; m, meters; µg/L, micrograms per liter; P, phosphorus; N, nitrogen; --, not applicable or not measured]

		West Bay								Center	East Bay
		Temper-ature (°C)	Dissolved oxygen (mg/L)	Total P (µg/L)	Dissolved P (µg/L)	Percent dissolved P	Kjeldahl N (µg/L)	Nitrate, N (µg/L)	Total N (µg/L)	Total P (µg/L)	Total P (µg/L)
Spring turnover	1997	4.5	11.6	20	2.0	10	500	66	566	15	15
	1998	5.0	12.8	12	1.5	13	460	73	533	12	10
	1999	5.1	12.1	10	1.0	10	490	79	569	--	8
	2000	4.9	12.0	12	1.0	8	510	68	578	--	12
Summer	1997	8.5	4.0	73	58.7	72	820	64	884	39	11
	1998	6.5	3.9	67	52.7	74	660	155	815	14	9
	1999	7.1	4.5	28	19.3	67	820	94	914		8
	2000	6.9	5.0	31							
Year average	1997	--	--	51	39.0	52	708	46	754	24	11
	1998	--	--	44	29.5	47	684	68	752	12	10
	1999	--	--	26	17.7	47	665	68	734	--	9
	2000	--	--	23	11.8	33	567	115	540	--	10
Study period annual average		**--**	**--**	**36**	**25**	**46**	**673**	**68**	**721**	**17**	**10**
Study period summer average		**7.3**	**4.4**	**50**	**44**	**71**	**767**	**104**	**871**	**26**	**9**

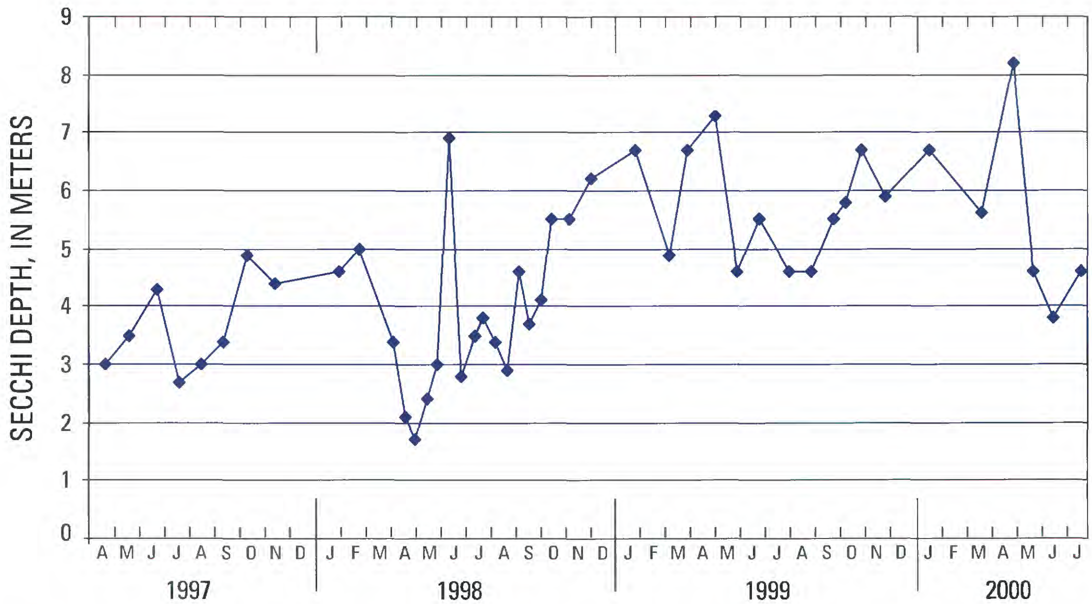


Figure 7. Secchi depth measurements from Geneva Lake, Wisconsin (West Bay), April 1997 through July 2000.

Dissolved Oxygen

Dissolved oxygen concentrations in Geneva Lake usually are very near saturation throughout the lake during well-mixed periods and throughout the epilimnion during stratified periods (fig. 6). The average concentration of dissolved oxygen in the epilimnion during summer was about 9 mg/L (table 6). As summer stratification develops and the hypolimnion is isolated from most surface mixing, dissolved oxygen in the hypolimnion is consumed. Within the hypolimnion, oxygen is consumed most quickly near the bottom and immediately below the thermocline. This consumption often results in a zone with higher dissolved oxygen concentrations around 25 m (fig. 6). This zone with higher dissolved oxygen concentrations was observed to a limited extent at the Center site, but not in other shallower areas of the lake. At 40 m, dissolved oxygen concentrations decreased to about 1 mg/L by August, and dissolved oxygen was consumed completely at this depth by September (fig. 6). Anaerobic conditions (concentrations less than 0.5 mg/L) can extend to about 30 m from the surface by late October or early November.

Water Clarity

During the study period, water clarity in the lake ranged from a Secchi depth of just less than 2 m in May of 1998 to greater than 8 m in May of 2000 (fig. 7). The mean Secchi depth during the study period was 4.8 m, and the mean during summer (June to August) was 4.1 m (table 6). During the study period, the water clarity increased from an annual mean of 3.7 m in 1997, to 4.2 m in 1998, to 5.7 and 5.6 m in 1999 and 2000, respectively. During this period, summer averages increased from 3.3 m in 1997 to 4.9 m in 1999 before decreasing to 4.3 m in 2000. Similar differences were observed at all of the sites examined.

Nutrient Concentrations

Phosphorus and N are essential nutrients for plant and algal growth; however, P is often the nutrient that limits biotic growth in Midwestern lakes. High P concentrations can cause dense algal populations (blooms) and can therefore be a major cause of accelerated eutrophication (increased algal and macrophyte productivity and accelerated aging) in lakes. Near-surface total P concentrations ranged from 3 to 16 μg/L in the West Bay (fig. 8a), although a concentration of 23 μg/L was measured once in the East Bay. The mean concentration in the West Bay during the study period was 8.6 μg/L, and the mean for summer periods was 8.3 μg/L (table 6). During the study period, there was no consistent trend in total P concentrations, and no consistent differences among sites. Near-surface dissolved P con-

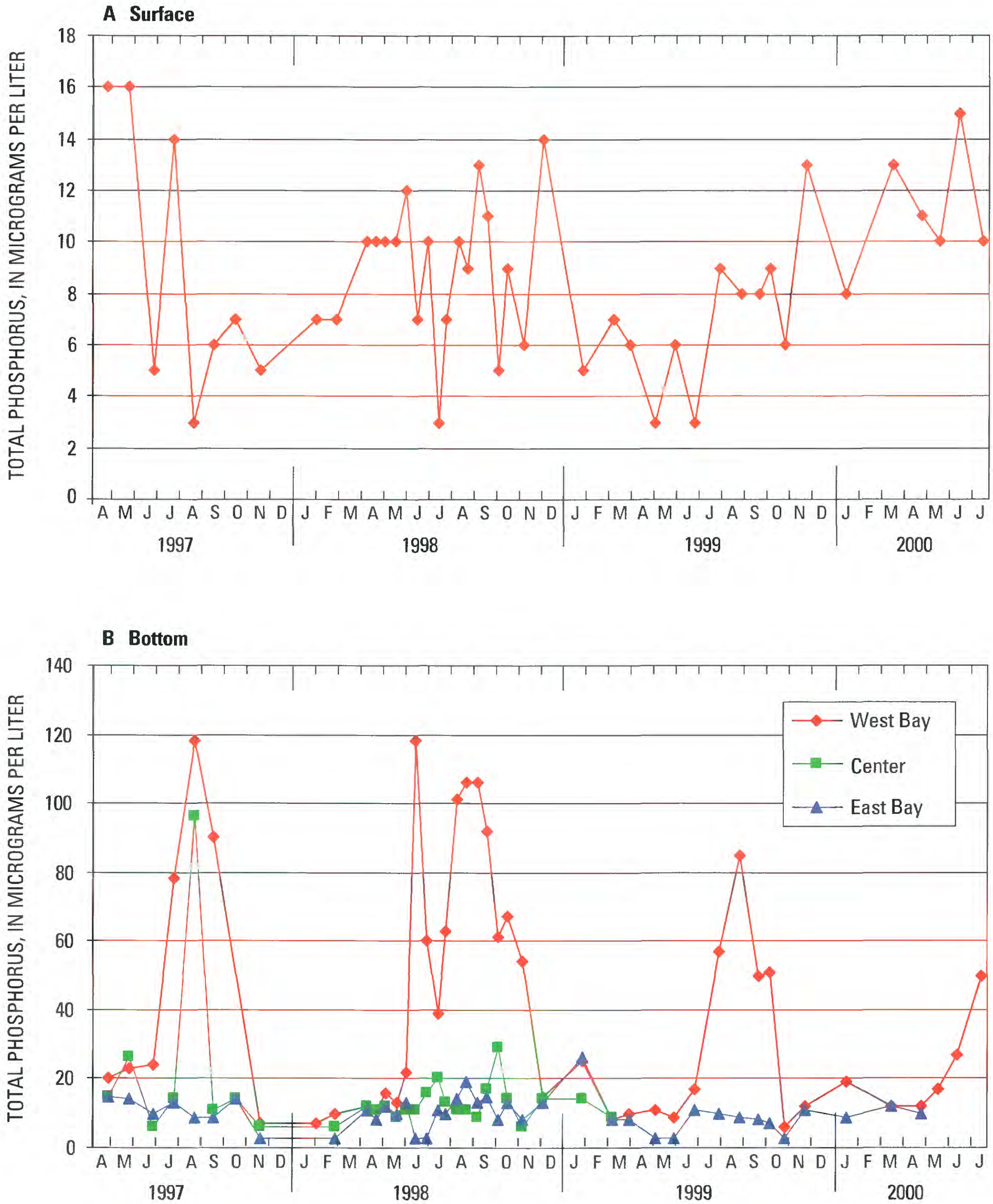


Figure 8. Total phosphorus concentrations in Geneva Lake, Wisconsin, April 1997 through July 2000: (A) surface of West Bay, and (B) bottom of West Bay, Center site, and East Bay.

centrations usually were low, ranging from 4 µg/L to less than the detection limit (1 µg/L).

Concentrations of total P throughout the water column, except just above the bottom, were similar to those measured near the surface. Near-bottom P concentrations in the West Bay generally increased from June through September, coinciding with the loss of dissolved oxygen (fig. 8b). The maximum near-bottom P concentration was 118 µg/L, occurring on two occasions during summer. After reaching peaks between late June and September, near-bottom concentrations decreased but usually remained elevated throughout October. Most near-bottom P was in dissolved forms from July to October, whereas most near-bottom P was in particulate forms the remainder of the year. During the study period, average-summer, near-bottom P concentration gradually decreased from 73 µg/L in 1997 to about 30 µg/L in 1999 and 2000 (table 7). The maximum near-bottom P concentration decreased from 118 µg/L in 1997 to 61 µg/L in 2000 (occurring just after the study period in October 2000). High near-bottom P concentrations were observed only once at the other sampling locations (during August 1997 at the Center site, fig. 8b).

Near-surface total N concentrations (computed as the sum of Kjeldahl N and dissolved nitrite plus nitrate) in the West Bay ranged from 355 µg/L to about 1,500 µg/L; only one concentration exceeded 1,000 µg/L (fig. 9a). During the study period, the mean concentration in the West Bay was 551 µg/L, and the mean for the summer periods was 574 µg/L (table 6). During this period, no consistent trend in total N concentrations was evident. Concentrations usually were higher in the West Bay than at other sites; therefore, N concentrations in other parts of the lake may be slightly lower than shown in figure 9a. Most near-surface N was in particulate forms (Kjeldahl N, 93 percent), especially in summer periods (97 percent) (table 6 and fig. 9a). The few relatively high Kjeldahl N concentrations were measured during summer. During summer, dissolved nitrate concentrations commonly were below the detection limit, whereas during fall through spring dissolved nitrate reached maximums of 80 µg/L (fig. 9a,c).

Near-bottom total N concentrations ranged from just under 400 µg/L to about 1,300 µg/L (fig. 9b). Most of the N was in the form of Kjeldahl N, similar to that at the surface. Near-bottom Kjeldahl N concentrations generally were around 400 µg/L and increased every summer to about 1,000 µg/L in early September to October (fig. 9b). Near-bottom dissolved nitrate concentrations also increased as summer progressed (fig. 9c); however, the concentrations dropped quickly when dissolved oxygen was depleted, usually in mid-August.

The ratio of concentrations of total N to total P (N:P ratio) often is used to determine the potential limiting nutrient in a lake. A N:P ratio greater than about 7:1 by weight (equivalent to an atomic ratio of 16:1) usually indicates that P is potentially the limiting nutrient, whereas a N:P ratio significantly less than 7:1 by weight indicates that N should be the limiting nutrient (Redfield, 1958; Correll, 1998). The specific value of when this ratio determines which nutrient potentially is limiting, however, has been shown to vary widely under various conditions (Correll, 1998). The N:P ratios (by weight) for near-surface water were always greater than 28:1, and the average during summer periods was almost 100:1 (table 6). Therefore, if just N and P are considered, P is the nutrient in shortest supply and should almost always be the nutrient limiting algal growth in Geneva Lake.

Chlorophyll *a* Concentrations

Chlorophyll *a* is a photosynthetic pigment found in algae and other green plants. Its concentration, therefore, is commonly used as a measure of the density of the algal population of a lake. During the study period, near-surface chlorophyll *a* concentrations ranged from 7.5 µg/L in April 1998 to about 0.5 µg/L on two occasions (fig. 10). The mean concentration during the study period was 2.9 µg/L and during just summer periods was 2.6 µg/L (table 6). During the study period, concentrations gradually decreased from an annual mean of 4.6 µg/L in 1997 to 1.5 µg/L in 2000. During this period, summer average concentrations decreased from 3.8 to 1.7 µg/L.

Trophic State Indices

One method of classifying the water quality or productivity of a lake is by computing TSI values. These indices, based on near-surface concentrations of P and chlorophyll *a*, and on Secchi depths, were developed to place these three characteristics on similar scales. Oligotrophic lakes (TSI values less than 40) have a limited supply of nutrients, typically are clear with low algal populations and P concentrations, and typically contain

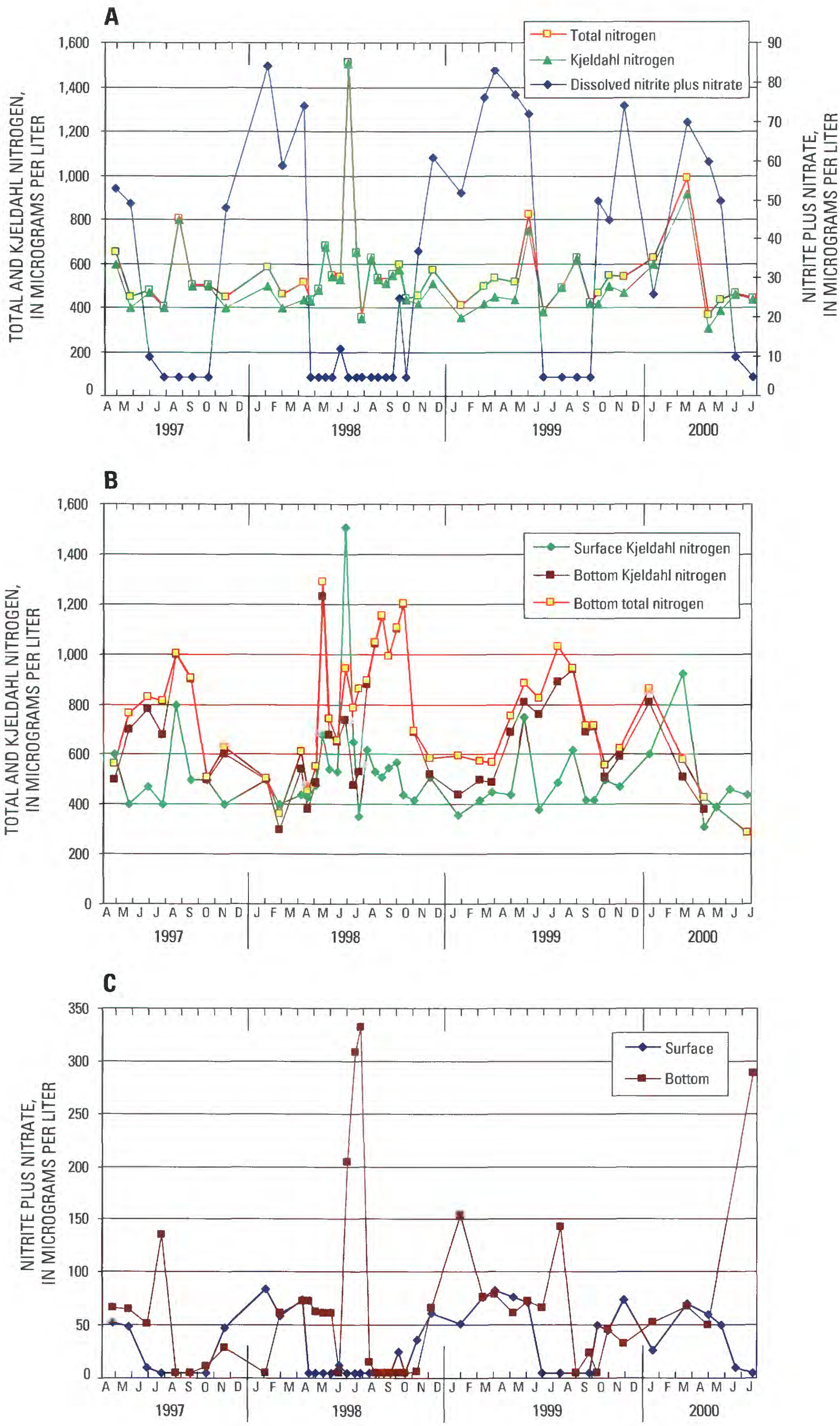


Figure 9. Nitrogen concentrations in Geneva Lake, Wisconsin (West Bay), April 1997 through July 2000: (A) nitrogen species at the surface, (B) Kjeldahl nitrogen at the surface and bottom, and (C) dissolved nitrite plus nitrate at the surface and bottom.

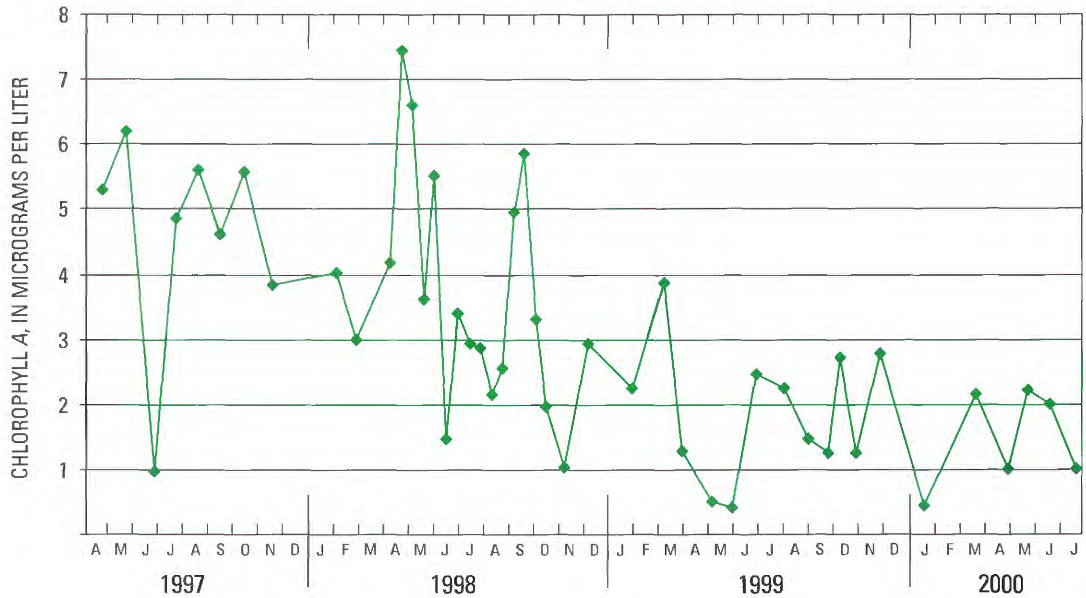


Figure 10. Chlorophyll *a* concentrations at the surface in Geneva Lake, Wisconsin (West Bay), April 1997 through July 2000.

oxygen throughout the year in their deepest zones. Mesotrophic lakes (TSI values between 40 and 50) have a moderate supply of nutrients, are not especially clear, and are prone to moderate algal blooms; occasional oxygen depletions in the deepest zones of the lake are possible. Eutrophic lakes (TSI values greater than 50) are nutrient rich with correspondingly severe water-quality problems, such as frequent seasonal algal blooms; oxygen depletion is common in the deeper zones of the lake, and clarity is poor.

All three indices generally indicate that Geneva Lake fluctuated between mesotrophic and oligotrophic (fig. 11). In 1997–98, all three indices indicate that the lake was generally mesotrophic (fig. 11 and table 6). By 1999, because of the trends toward increased clarity and decreased chlorophyll *a* concentrations, TSI values based on these characteristics indicated that the lake was generally oligotrophic. However, TSI values based on P concentrations still classified the lake as mesotrophic. The decreased chlorophyll *a* concentrations and increased clarity with little change in P concentrations may indicate that something other than nutrients was limiting the lake's response, possibly a change in the foodweb associated with zebra mussels or a change in the fish community. Zebra mussels were first observed in the lake in the fall of 1995, and their populations recently have increased dramatically, especially after 1998 (T. Peters, Geneva Lake Environmental Agency, written commun., 2000).

Plankton

Phytoplankton

To describe the phytoplankton species composition and populations, phytoplankton species were divided into six groups: bacillariophyta (diatoms), chlorophyta (green algae), cryptophyta (cryptomonads), chrysophyta (golden-brown algae), cyanophyta (blue-green algae), and pyrrhophyta (dinoflagellates). The total biovolume of each group was computed, and these biovolumes are compared in figure 12.

In general, phytoplankton in Geneva Lake were dominated by diatoms (various *Fragilaria* and *Cyclotella* species) except during some summer periods when cryptomonads (primarily *Cryptomonas erosa* and *Rhodomonas minuta*) and dinoflagellates (primarily *Ceratium hirundinella*) were abundant (fig. 12). The total biovolume of phytoplankton demonstrated distinct seasonality, generally peaking in April or May and reaching lowest levels in June and July. The highest biovolume was 1,370,000 μm^3/mL in May 1997, and lowest biovolume of 50,000 μm^3/mL in June 1999. This seasonality in total biovolume directly corresponded with changes in diatom populations. Green and blue-

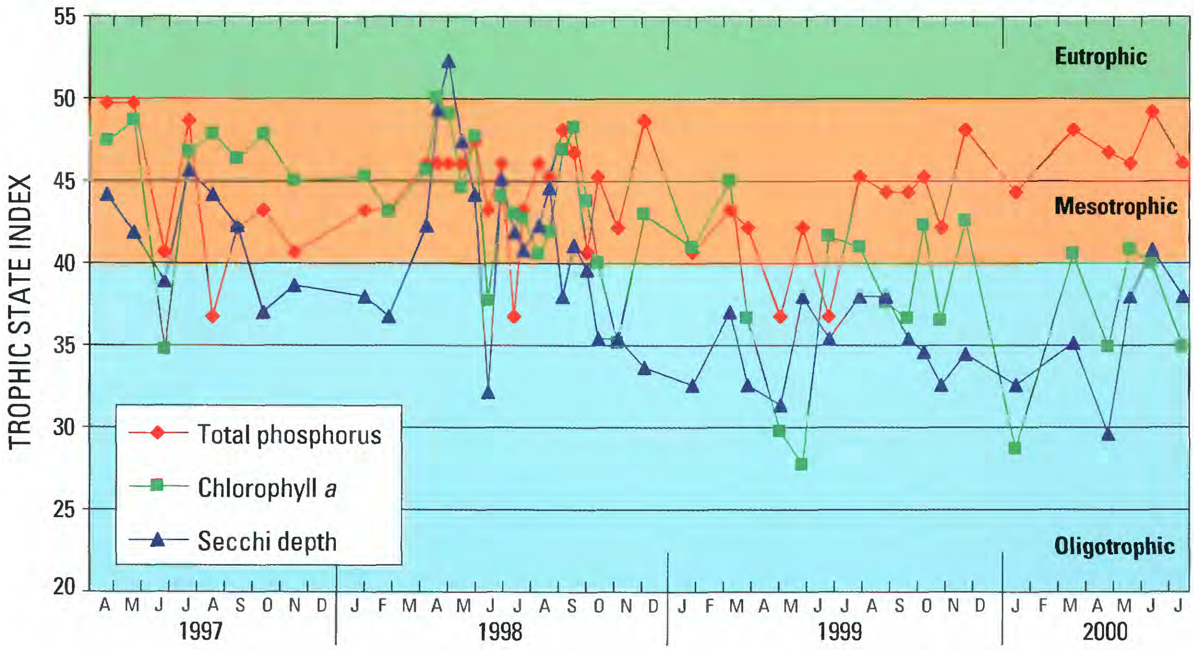


Figure 11. Trophic State Index values for Geneva Lake, Wisconsin (West Bay), April 1997 through July 2000.

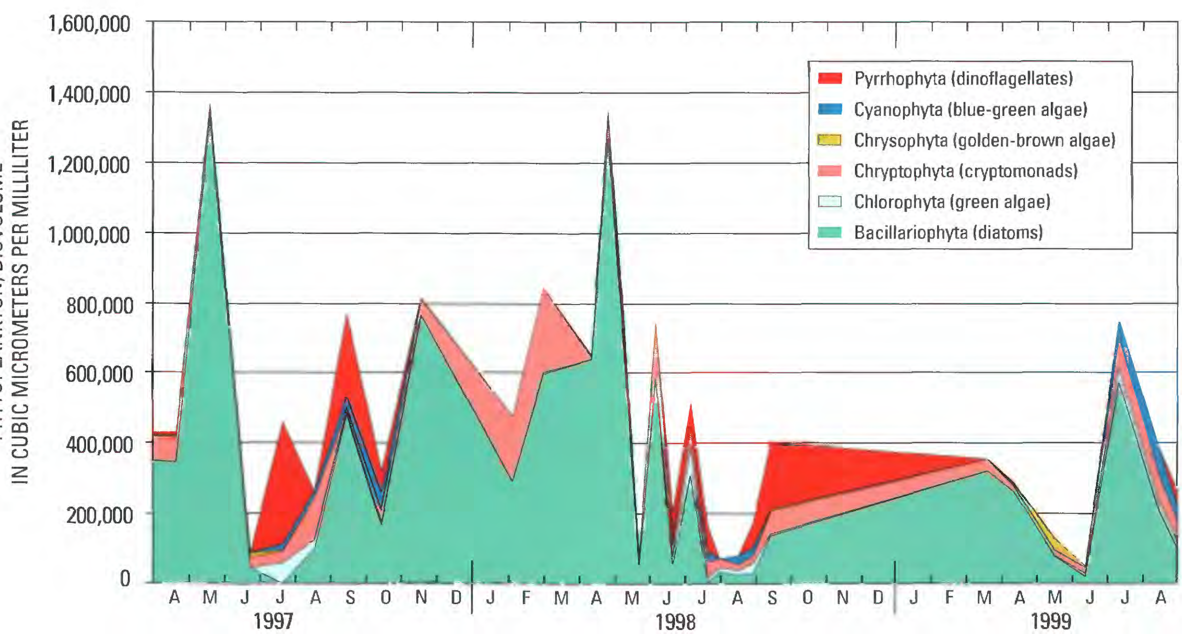


Figure 12. Phytoplankton populations in the surface water of Geneva Lake, Wisconsin (West Bay), April 1997 through August 1999.

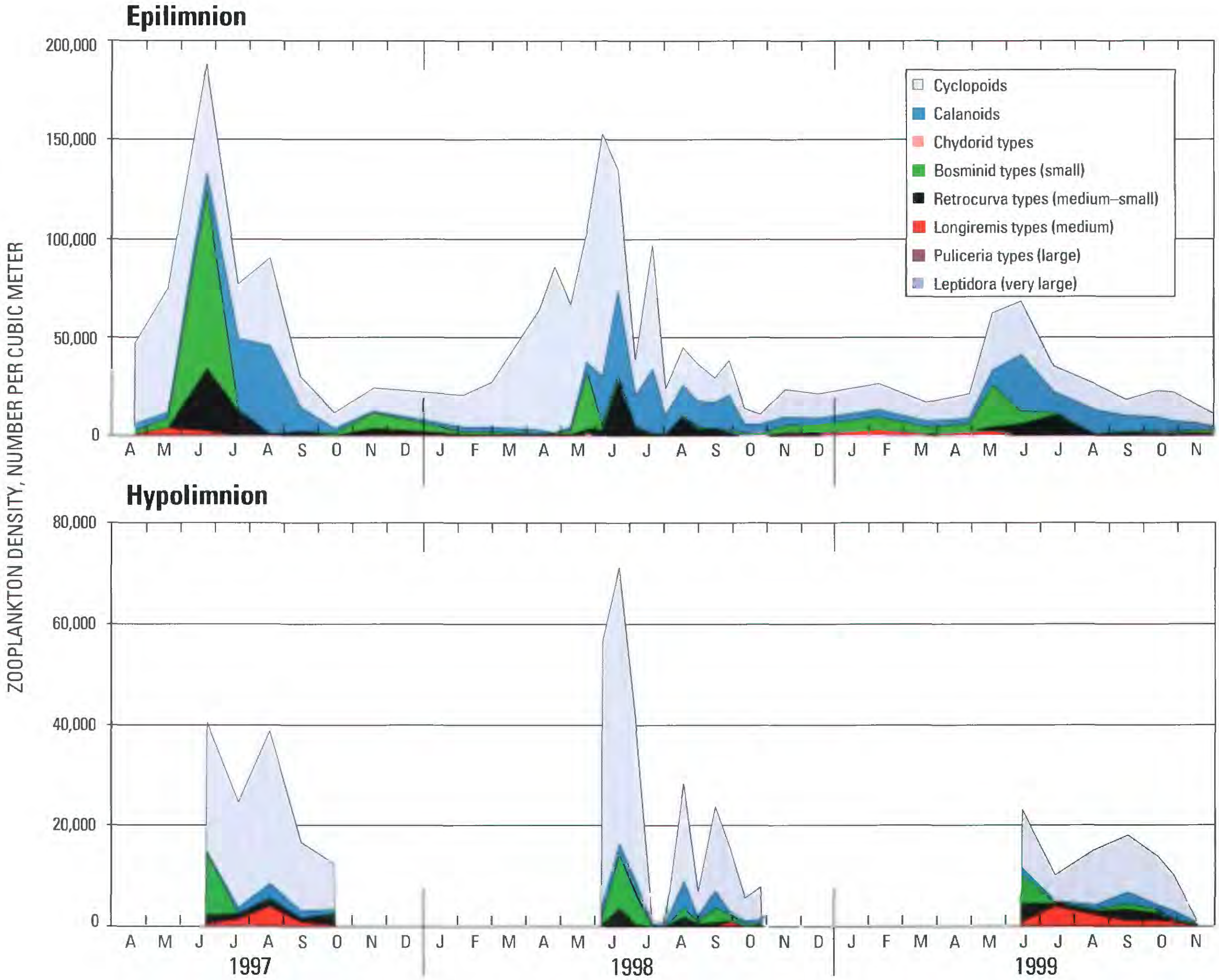


Figure 13. Zooplankton populations in the epilimnion and hypolimnion of Geneva Lake, Wisconsin (West Bay), April 1997 through November 1999.

green algae never were very important to the total biovolume of phytoplankton.

Zooplankton

To describe changes in zooplankton species composition and populations, the zooplankton species were divided into cladocerans and copepods. The cladocerans were further subdivided into six groups based primarily on size: Leptidora (very large species, *Leptidora kindti*), puliceria types (large species, none were found during the study period), longiremis types (medium-sized species, primarily *Daphnia longiremis*), retrocurva types (medium-to-small sized species, primarily *Daphnia retrocurva* and *Diaphanosoma brachyurum)*, bosminid types (small species, including *Bosmina* and *Eubosmina* species), and chydorid types (associated with blue-green algae, primarily *Chydorid sphaericus*). Copepods were divided into calanoids and cyclopoids. Total populations of each group were computed for the epilimnion and beneath the thermocline when it was present, and these populations are compared in figure 13.

Zooplankton in Geneva Lake were dominated by small to medium-sized species (fig. 13). Usually the most abundant type of zooplankton was cyclopoid copepods; however, often during summer, calanoid copepods and small bosminid types also were abundant. Larger species of daphnids were rare in the lake, except for medium-to-small sized retrocurva types in July.

During stratified periods, zooplankton populations were subdivided into epilimnetic and hypolimnetic populations. The population distributions were similar in these two areas, but population densities were about twice as high in the epilimnion as in the hypolimnion. No large daphnia were found during the study period; however, a few very large Leptidora were found in the epilimnion during summer.

The density of zooplankton demonstrated strong seasonality, with highest densities from May through August, usually peaking in June (fig. 13). These high densities were caused by increases in the cyclopoid copepods, calanoid copepods, bosminids, and retrocurva types. Lowest densities of zooplankton were observed from November through March.

CHEMICAL COMPOSITION OF THE SURFICIAL SEDIMENT

The chemical composition of the surficial sediment of the lake previously was described in a USGS Fact Sheet entitled "Chemical composition of surficial sediment of Geneva Lake, Wisconsin" by Elder and others (2000); therefore, only a summary of the findings is presented here.

The characteristics of the sediment and concentrations of nutrients, metals, and major ions measured in the sediments of the West Bay, Williams Bay, and Geneva Bay are summarized in table 8. Total P concentrations in the surficial sediments ranged from 0.78 g/kg in the West Bay to about 1.5 g/kg in the shallower bays. Total N concentrations also were higher in the shallower bays (about 15 g/kg) than in the West Bay (8.1 g/kg). Of the values in table 8, only arsenic (ranging from about 17 to 35 g/kg) and zinc concentrations (ranging from 158 to 232 mg/kg) were above the "Low Range" of effects based on toxicity data for the Great Lakes, 13 g/kg and 110 mg/kg, respectively (Ingersoll and others, 1996). A summary of the concentrations of synthetic organic compounds (primarily aromatic hydrocarbons, PAHs, and DDT derivatives) in the sediments of Williams Bay and Geneva Bay are given in table 9. Of these compounds, the concentrations of benzo(a)anthracene, dibenzo(a,h)anthracene, acenaphthalene, fluoranthene, chrysene, phenanthrene, pyrene, benzo(a)pyrene, DDD, and DDE were above the Interim Sediment Quality Guidelines of the Canada (Canadian Council of Ministers of the Environment, 1999), but all were below the Probable Effects Levels, except DDE. The sediments also were analyzed for a number of commonly used pesticides (dieldrin, endrin, heptachlor epoxide, aldrin, endosulfan, lindane, and mirex), but none were detected.

Analyses of the surficial sediments of Geneva Lake indicated that agricultural and municipal drainage from the watershed has produced elevated (above background levels), and potentially problematic, concentrations of some elements and compounds in the lake sediments. Moreover, continued development in the watershed could increase loading and the accumulation of these compounds. Although the concentrations of various potentially toxic elements and compounds were found to exceed published criteria and guidelines for sediments, the actual effects on organisms are complicated by many interacting environmental factors, most of which have not yet been investigated in the Geneva Lake system. Because concentrations were elevated above background levels indicates little about actual toxicological effects on any biological species, but it serves as a signal that future investigators of contaminants in the lake may want to focus on such details as bioavailability, cycling within the ecosystem, and toxicology of these compounds.

WATER QUALITY OF AND LOADING FROM TRIBUTARIES TO GENEVA LAKE

Two tributaries (Southwick Creek at Williams Bay and Birches Creek at Lackey Lane; fig. 1) were used primarily to describe the flow, water quality, and nutrient loading from the watershed. Flow and concentrations of P and SS in Southwick Creek and Birches Creek were substantially different (figs. 14 and 15). Southwick Creek had a relatively steady base flow of approximately 3,000 m^3/d throughout the year with many moderate-sized runoff events (flow exceeding 5,000 m^3/d), typical of an urbanizing area. The highest daily flow was about 39,000 m^3/d in June 1999. In comparison, Birches Creek had a base flow of less than 1,000 m^3/d during late summer and winter, but base flow increased to about 4,000–6,000 m^3/d during March through June. Birches Creek had fewer, but much larger runoff events than did Southwick Creek. The highest daily average flow was greater than 350,000 m^3/d during a storm in June 1999; the second highest was about 100,000 m^3/d in April 1999.

Water temperatures in both streams closely followed air temperatures (fig. 16). Water temperatures reached about 20°C in both streams in July, similar to the daily average air temperature. Water temperatures in

Table 8. Characteristics and concentrations of nutrients, metals, and major ions measured in the sediments of the West Bay, Williams Bay, and Geneva Bay of Geneva Lake, Wisconsin

[All element concentrations based on dry weight; %, percent; g/kg, grams per kilogram; mg/kg, milligrams per kilogram; --, no data]

Characteristic variable and unit	West Bay	Williams Bay	Geneva Bay
Water content, %	89.3	80.8	84.2
Loss on ignition, %[1]	13.3	13.3	14.0
Calcium carbonate, %[1]	25.1	23.4	26.9
Calcium, %[1]	22.2	--	--
Total phosphorus, g/kg	0.78	1.61	1.49
Total nitrogen, g/kg	8.1	15.8	13.4
Aluminum, g/kg	7.9	17.6	--
Potassium, g/kg	12.2	--	--
Iron, g/kg	.9	--	--
Manganese, g/kg	1.12	--	--
Arsenic, g/kg	35.4	16.8	19.3
Copper, mg/kg	44	--	--
Zinc, mg/kg	158	232	241

[1]Percentage on the basis of dried sediment.

Table 9. Aromatic hydrocarbons and chlorinated hydrocarbons detected in sediments of Geneva Lake, Wisconsin, sampled July 17, 1997

[All concentrations in nanograms per gram, dry weight; e, estimate; <, less than; --, no data]

Compound	Williams Bay	Geneva Bay
Anthracene	14 e	14 e
Benzo(*a*)anthracene	45 e	44 e
Di-Benzo(a,h)anthracene	<50	12
Acenaphthylene	9 e	<50
Fluoranthene	130	110
Benzo(b,k)fluoranthene	103	110
Chrysene	43 e	41 e
Phenanthrene	56	47 e
Pyrene	110	91
Benzo(*a*)pyrene	53	52
Benzo(*ghi*)perylene	45 e	47
Phenol	15 e	17
DDD [1](*)	4–7	6–8
DDE [1](*)	9–13	12–15
2,6-Dimethylnaphthalene	58	74
4-HCY Penphenanthrene	17 e	<50
9,10-Anthraquinone	37 e	39 e
9H-Fluorene	3 e	<50
Carbazole	13 e	<50
p-Cresol	8 e	<50
Indeno(1,2,3-cd)pyrene	40 e	47 e
Mesitol	23 e	--
1-Methylpyrene	15 e	14 e

[1](*) sum of o,p' and p,p' isomers

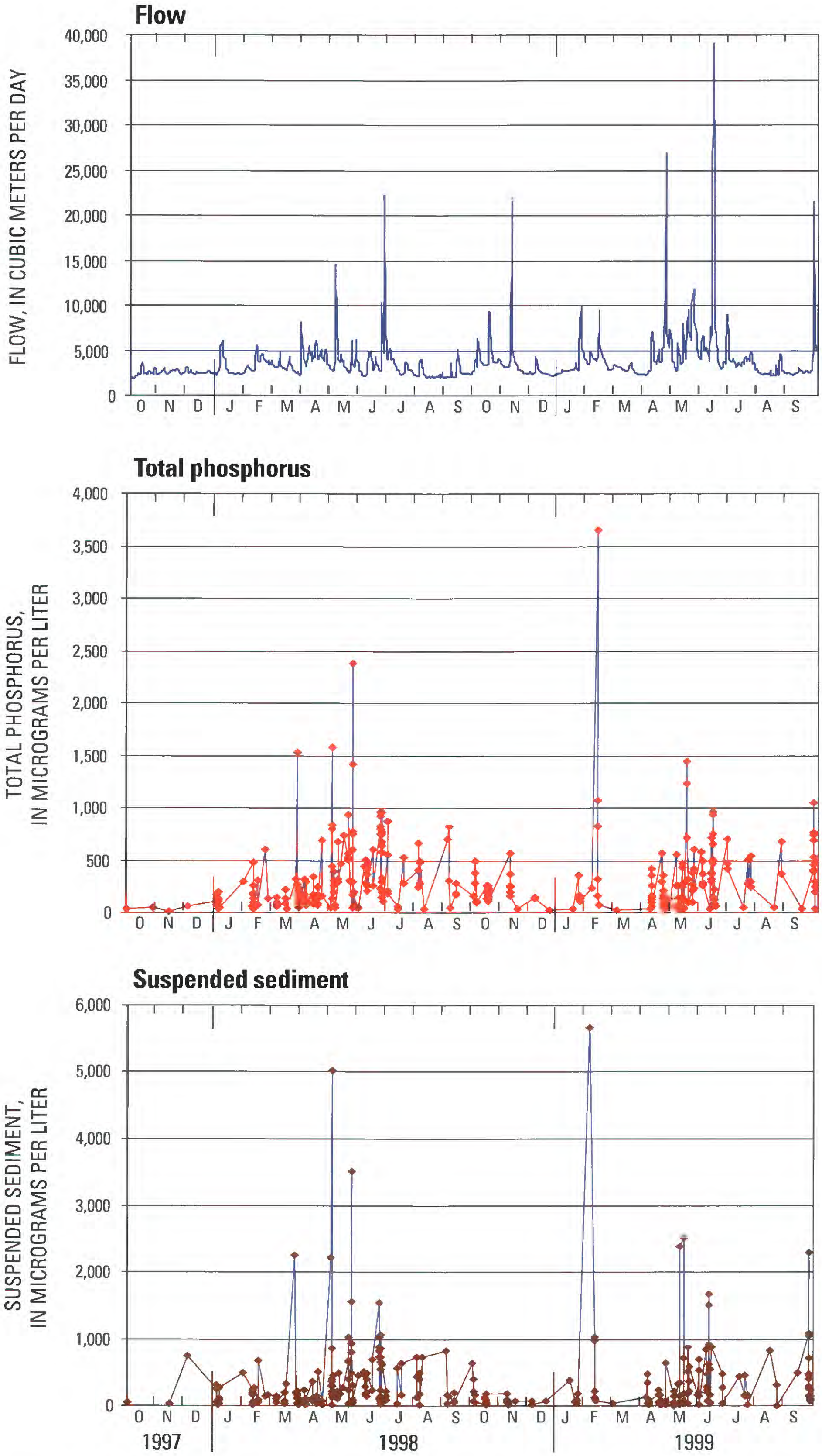


Figure 14. Daily-average flow, and total phosphorus and suspended sediment concentrations in Southwick Creek, tributary to Geneva Lake, Wisconsin, October 1997 through September 1999.

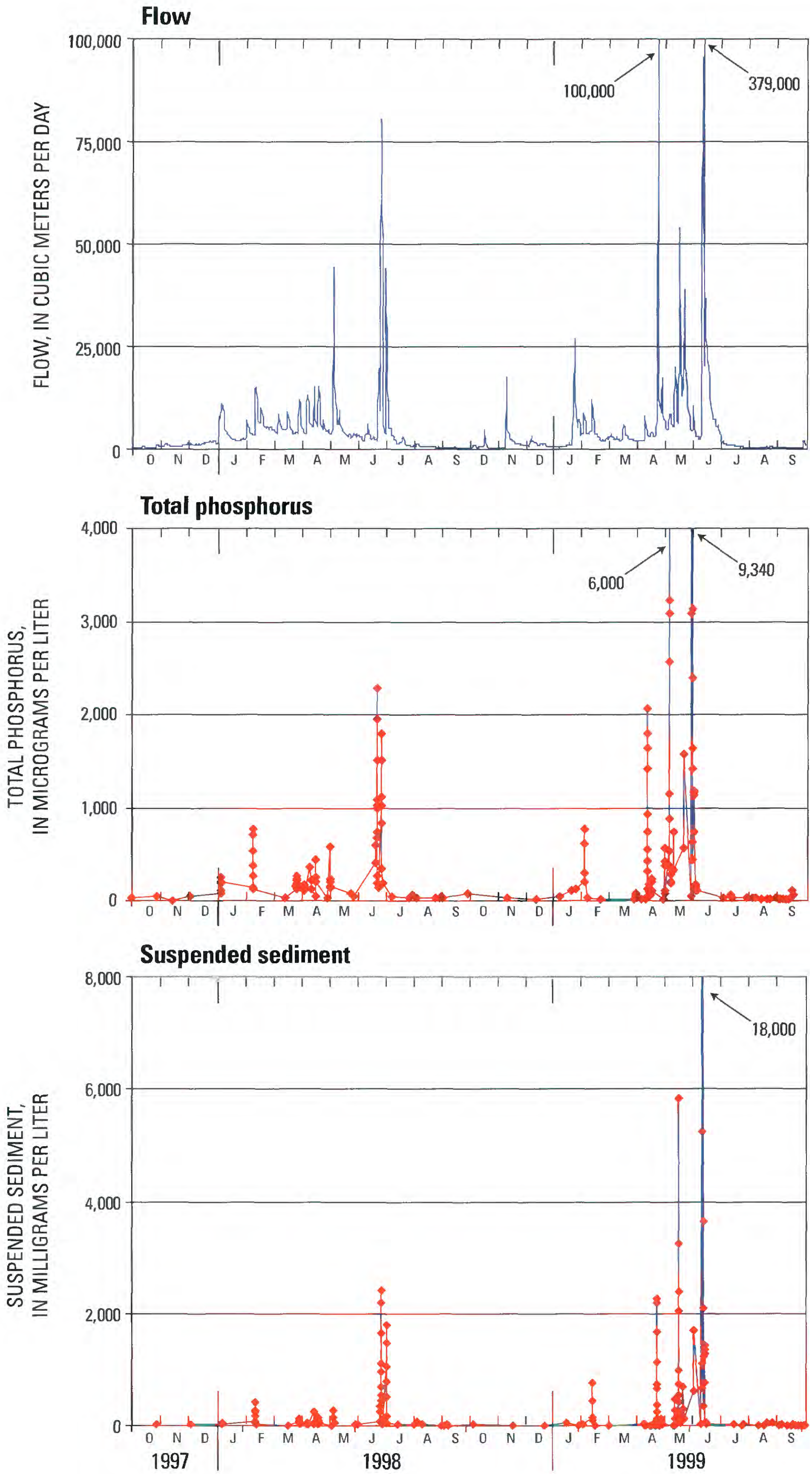


Figure 15. Daily-average flow, and total phosphorus and suspended sediment concentrations in Birches Creek, tributary to Geneva Lake, Wisconsin, October 1997 through September 1999.

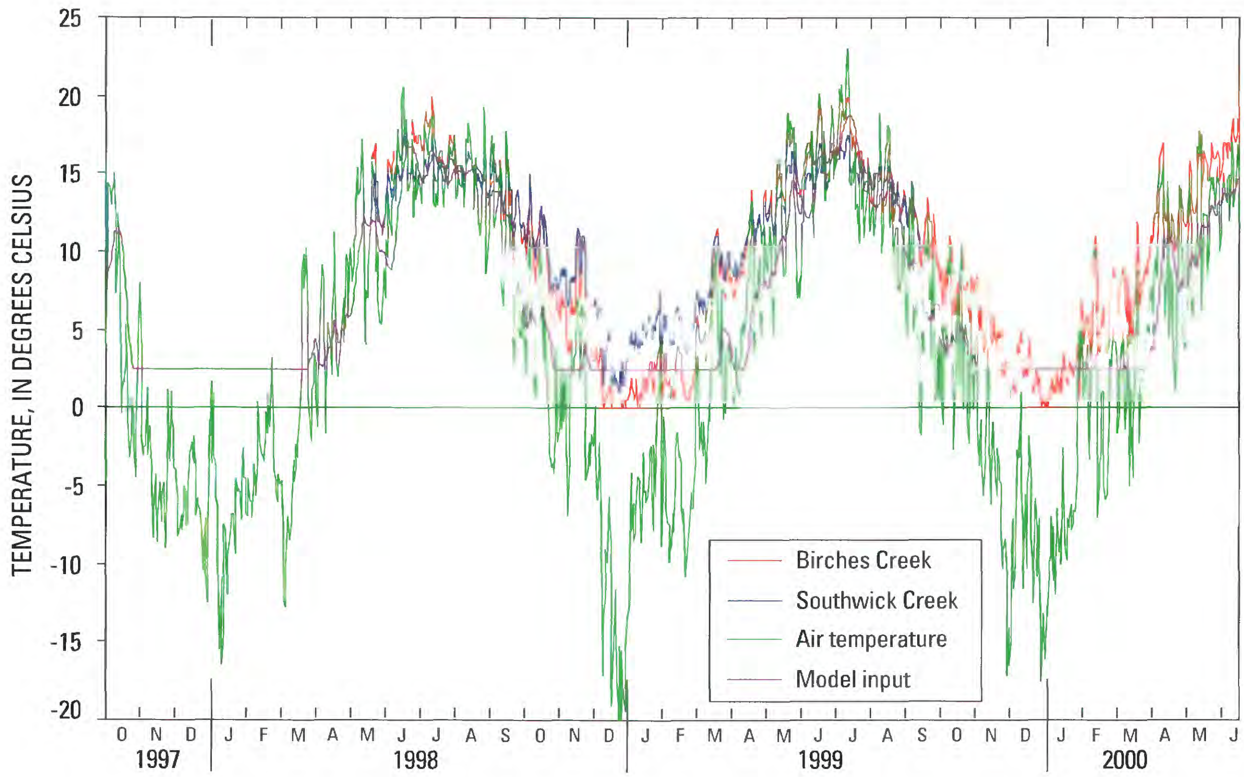


Figure 16. Water temperatures measured in Southwick and Birches Creeks and air temperatures measured at Lake Geneva, Wisconsin, October 1997 through June 2000. (Water temperatures used in model simulations also are shown.)

Birches Creek resembled air temperatures more closely than those in Southwick Creek, which were slightly less. In fall, water temperatures in both streams were slightly higher than air temperatures. In winter, water temperatures decreased to about 0°C in Birches Creek and about 2.5°C or slightly higher in Southwick Creek. In spring, water temperatures also were slightly higher than daily average air temperatures but became similar to air temperatures by early June. The daily range in water temperatures (not shown in fig. 16) was large. The maximum daily range in temperatures in both streams was about from 9 to 10°C.

Phosphorus and SS concentrations in both streams were strongly related to flow. During base flow, P concentrations generally were below 70 µg/L in both streams, and SS concentrations were below 100 mg/L (figs. 14 and 15). During flows higher than base flow, however, concentrations increased dramatically. During higher flows in Southwick Creek, P concentrations commonly were about 500 µg/L but were as high as 3,600 µg/L, and SS concentrations commonly were about 1,000 mg/L but were as high as 5,600 mg/L. During higher flows in Birches Creek, P concentrations commonly were about 500 µg/L but were as high as 9,000 µg/L, and SS concentrations commonly were about 500 mg/L and lower than in Southwick Creek but were as high as 18,000 mg/L.

Phosphorus concentrations also were examined in some detail in Buttons Bay Creek (a site with extensive wetlands in the basin, fig. 1). During the sampling period, concentrations in this stream were more associated with seasonality than with flow. Concentrations were relatively low from October to April (70 to 100 µg/L), increased in early summer, peaked around 1,200 to 1,500 µg/L in July, and then decreased in late summer. To demonstrate this seasonal pattern, data from three different years were all plotted on the same graph (fig. 17). This pattern in P concentrations is similar to what was observed in Delavan Lake Inlet, a site just north of Geneva Lake that also has extensive wetlands in its basin (Robertson and others, 1996).

In addition to Southwick, Birches, and Buttons Bay Creeks, flow was measured and water samples were collected and analyzed for total P concentrations less frequently at 23 additional sites (table 10) to describe the variability in flow and water quality in the other trib-

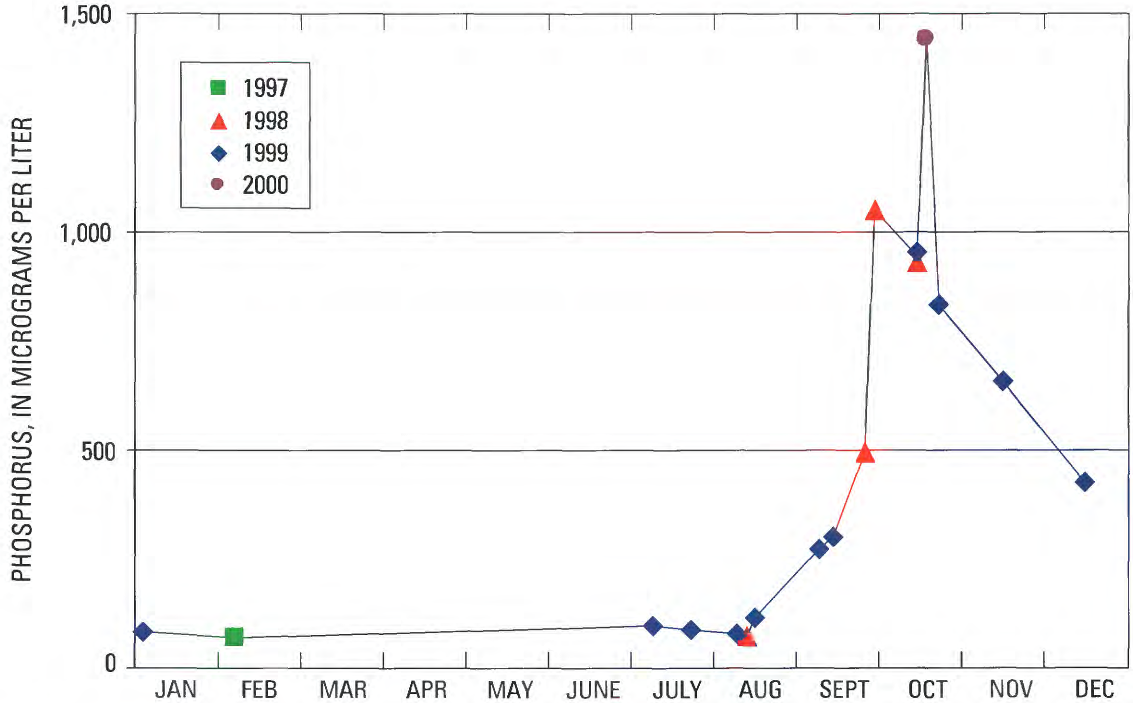


Figure 17. Phosphorus concentrations measured in Buttons Bay Creek, tributary to Geneva Lake, Wisconsin, 1997 through 2000.

utaries during base-flow (fig. 18, shown for only July 19, 2000) and high-flow periods (fig. 19). During base-flow measurements made on July 19, 2000, P concentrations ranged from 17 to 1,440 µg/L. In general, P concentrations were less than 75 µg/L, except in Buttons Bay Creek (1,440 µg/L), Trinke Creek (Subwatershed 20, 150 µg/L), and Subwatershed 56 (98 µg/L). The Buttons Bay subwatershed has extensive areas of agriculture and wetlands. Southwick and Birches Creeks had P concentrations of 46 and 43 µg/L, respectively, which were typical of most base-flow periods during the 2 years they were sampled. Therefore, the differences among tributaries found during this one synoptic period should represent the typical variability throughout the watershed during most base-flow periods.

During the high-flow measurements made on May 13, 1998, P concentrations ranged from 26 to 137 µg/L (table 10 and fig. 19). Southwick and Birches Creeks had concentrations of 110 and 122 µg/L, respectively. The highest P concentrations were measured in a tributary to Geneva Bay (Subwatershed 3, 137 µg/L), Birches Creek, Southwick Creek, and Trinke Creek (Subwatershed 20, 106 µg/L). Concentrations of P in Southwick and Birches Creeks varied considerably during other high-flow periods during the 2 years they were sampled and often were much different and much higher than shown in figure 19. Therefore, the concentrations measured during this particular high-flow period are not expected to represent all high-flow conditions throughout the watershed.

Prior to 1984, Buena Vista Creek (Subwatershed 23) received effluent from the Fontana sewage-treatment plant and was found to have P concentrations routinely as high as 1,500 µg/L during low and high flows (T. Peters, Geneva Lake Environmental Agency, written commun., 2000). However, with the removal of Fontana sewage-treatment plant effluent from the creek, water quality in Buena Vista Creek appreciably improved, and concentrations above 50 µg/L were not observed in this study.

The 15-minute-interval flow values and water-quality data from Southwick and Birches Creeks were used to compute the total loadings of P and SS from these tributaries to Geneva Lake for WY1998 and WY1999 (table 11). Most of the loading from both streams, especially Birches Creek, occurred during short-term (usually less than 1 day), high-flow events, as demonstrated for P in figure 20. During 6 days of high flows, more than 55 kg of P and 75,000 kg of SS

Table 10. Flow and water quality in selected tributaries to Geneva Lake, Wisconsin

[Location of sites are shown in figures 18 and 19. P, phosphorus; NS, site not sampled; NM, not measured, NF, no flow; km^2, square kilometers; m^3/s, cubic meters per second; $m^3/s/km^2$, cubic meters per second per square kilometer; µg/L, micrograms per liter; kg/d, kilograms per day; kg/d/km^2, kilograms per day per square kilometer]

Subwatershed number (name)	Area (km^2)	11/8/97 - Base flow: Flow (m^3/s)	11/8/97: Flow yield ($m^3/s/km^2$)	11/8/97: Total P (µg/L)	11/8/97: Total P load (kg/d)	11/8/97: Total P yield (kg/d/km^2)	7/19/00 - Base flow: Flow (m^3/s)	7/19/00: Flow yield ($m^3/s/km^2$)	7/19/00: Total P (µg/L)	7/19/00: Total P load (kg/d)	7/19/00: Total P yield (kg/d/km^2)	5/13/98 - High flow: Flow (m^3/s)	5/13/98: Flow yield ($m^3/s/km^2$)	5/13/98: Total P (µg/L)	5/13/98: Total P load (kg/d)	5/13/98: Total P yield (kg/d/km^2)
3	0.959	NS	NS	NS	NS	NS	0.001	0.001	57	0.003	0.003	0.003	0.003	137	0.034	0.035
6	.871	NS	NS	NS	NS	NS	.000	NF	NF	NF	NF	.016	.018	58	.080	.092
7	.327	NS	NS	NS	NS	NS	.001	.004	64	.007	.020	.007	.020	68	.039	.119
9	.347	NS	NS	NS	NS	NS	.000	NF	NF	NF	NF	NM	NM	64	NM	NM
14	2.030	0.021	0.010	22	0.041	0.020	.035	.017	22	.068	.034	.019	.009	29	.049	.024
15	1.644	.006	.003	36	.018	.011	.006	.004	45	.023	.014	.006	.003	71	.035	.021
17	.166	.008	.051	NM	NM	NM	.009	.056	17	.014	.083	.005	.031	31	.014	.083
20 (Trinke)	1.818	.009	.005	52	.040	.022	.016	.009	150	.208	.114	.037	.020	106	.341	.188
22	.539	.004	.007	NM	NM	NM	.005	.009	35	.015	.029	.006	.010	61	.030	.056
23 (Buena Vista)	.646	.008	.012	28	.019	.029	.007	.012	29	.019	.029	.009	.014	48	.038	.059
24	1.201	.048	.040	21	.088	.074	.045	.038	21	.083	.069	.054	.045	31	.146	.121
26	.472	.000	.000	NM	NM	NM	.001	.001	72	.003	.007	.004	.009	55	.020	.043
27	1.789	.065	.036	20	.114	.064	.101	.056	18	.159	.089	.074	.041	29	.187	.104
34	.262	.000	.000	NM	NM	NM	.001	.002	63	.003	.012	.003	.011	55	.014	.052
36	.508	.003	.006	25	.006	.012	.008	.016	24	.017	.034	.010	.021	35	.032	.063
39	.895	NS	NS	NS	NS	NS	.000	NF	NF	NF	NF	.006	.006	78	.039	.043
40	1.211	.001	.001	NM	NM	NM	.003	.002	63	.014	.012	.019	.015	53	.087	.072
41	.718	.012	.017	33	.034	.048	.007	.010	19	.012	.016	.019	.026	26	.043	.060
54	1.066	.004	.004	41	.015	.014	.001	.001	45	.003	.003	.014	.013	43	.052	.049
56	1.188	NS	NS	NS	NS	NS	.001	.001	98	.009	.008	.003	.002	57	.014	.012
59	1.141	.006	.005	41	.020	.018	.002	.001	31	.004	.004	.012	.010	41	.043	.037
61	.036	.001	.039	NM	NM	NM	.000	NF	NF	NF	NF	NF	NF	NF	NF	NF
63	.340	.004	.012	NM	NM	NM	.001	.002	73	.004	.011	NS	NS	NS	NS	NS
47 (Buttons Bay)	5.082	.009	.002	68	.054	.011	.022	.004	1,440	2.717	.535	.034	.007	75	.223	.044
32 (Birches)	5.368	.013	.002	42	.049	.009	.016	.003	43	.058	.011	.110	.021	122	1.178	.220
28 (Southwick)	2.256	.027	.012	26	.062	.028	.038	.017	46	.155	.069	.054	.024	110	.518	.229

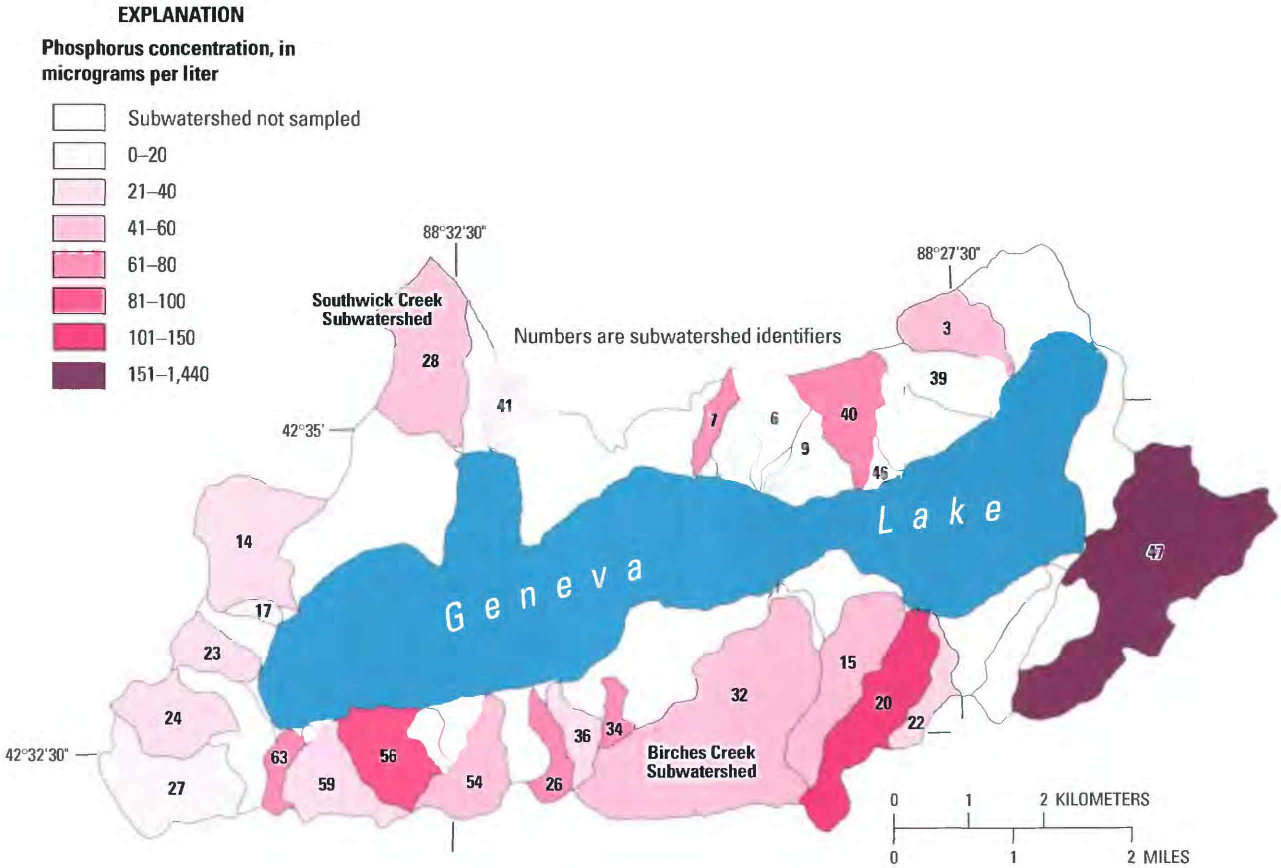


Figure 18. Phosphorus concentrations measured in the tributaries to Geneva Lake, Wisconsin, during base flow, July 19, 2000. (Data for subwatersheds are given in table 10.)

(approximately 25 and 40 percent of the total 2-year loads, respectively) were delivered from Southwick Creek. The total annual loads of P from Southwick Creek were 98 kg (43 kg/km^2) for WY1998 and 121 kg (54 kg/km^2) for WY1999. The total annual loads of SS were 75,000 kg (33,000 kg/km^2) for WY1998 and 110,000 kg (49,000 kg/km^2) for WY1999. During 5 days of high flows, 1,080 kg of P and 1,300,000 kg of SS (approximately 70 and 81 percent, respectively, of the 2-year loads) were delivered from Birches Creek. The total annual loads of P from Birches Creek were 283 kg (53 kg/km^2) for WY1998 and 1,273 kg (238 kg/km^2) for WY1999, respectively. The total annual loads of SS were 181,000 kg (34,000 kg/km^2) for WY1998 and 1,408,000 kg (263,000 kg/km^2) for WY1999.

White River (lake outlet) was sampled just downstream from the lake to determine its water quality and the amount of nutrients removed from the lake. Flow in White River was seasonal in nature with little water leaving the lake from early July to early January (fig. 21). From January to June, outflow generally varied from 100,000 to 300,000 m^3/d. The highest outflow from the lake (over 800,000 m^3/d) occurred in mid-June 1999. The P concentrations in the White River generally were below 20 µg/L, except for three measurements collected during very low flow, the highest of which was 275 µg/L on September 28, 1999 (fig. 21). These generally low concentrations reflect the low P concentrations near the lake surface. Suspended sediment concentrations always were low, the highest of which was 16 mg/L. The 5-minute-interval flow values and water-quality data from the White River were used to compute the total loadings of P and SS leaving Geneva Lake (table 11). The total annual P loads from the lake were 178 kg in WY1998 and 225 kg in WY1999. The total annual loads of SS from the lake were about 35,000 kg in WY1998 and 65,000 kg in WY1999.

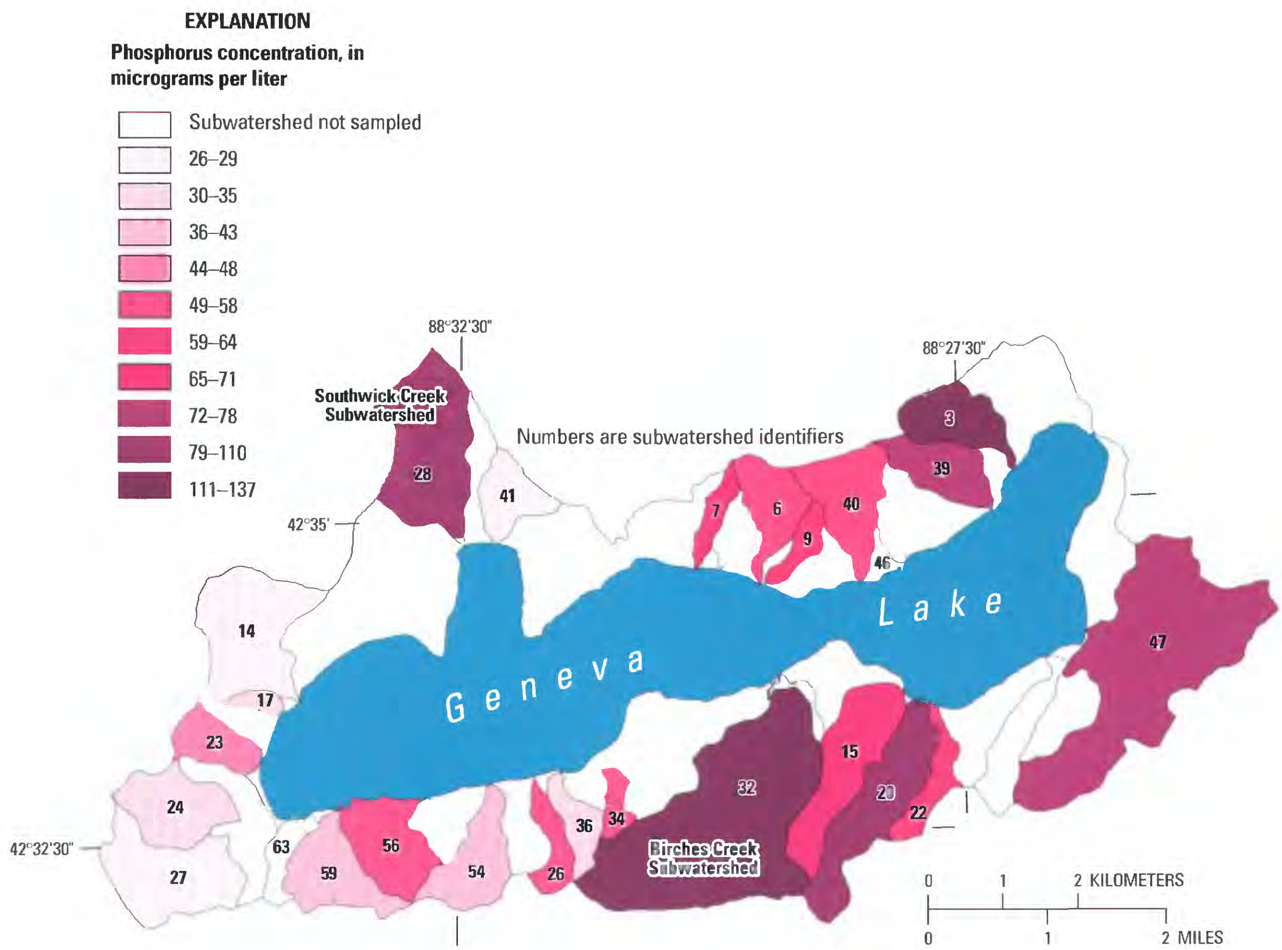


Figure 19. Phosphorus concentrations measured in the tributaries to Geneva Lake, Wisconsin, during high flow, May 13, 1998. (Data for subwatersheds are given in table 10.)

HYDROLOGIC BUDGET

The hydrology of Geneva Lake can be described in terms of components of its water budget (fig. 22) represented by

$$\Delta S = (PPT + SW_{In} + GW_{In}) - (Evap + SW_{Out} + GW_{Out}), \quad (1)$$

where ΔS is the change in the volume of water stored in the lake during the period of interest and is equal to the sum of the volumes of water entering the lake minus the sum of the volumes of water leaving the lake. Water enters the lake as precipitation (PPT), surface-water inflow (SW_{In}), and ground-water inflow (GW_{In}). Water leaves the lake through evaporation (Evap), surface-water outflow (SW_{Out}), and ground-water outflow (GW_{Out}). Each term in the water budget was computed on a daily basis for the 2-year period from October 1997 to September 1999 (WY1998 and WY1999).

Change in Storage

Changes in lake volume were determined from water elevations measured every 5 minutes at the outlet dam (fig. 23) and the surface area of the lake (table 1). Lake stage fluctuated from a minimum elevation of 0.62 m (relative to an arbitrary datum) to a maximum elevation of 0.96 m. Lake stage varied seasonally and generally was highest during mid-April through early July and lowest during the winter. The lake stage at the end of each 1-year period was similar to that at the beginning of the period.

Precipitation

During the 2-year period, total daily precipitation was measured at three sites around the lake: the Delavan Lake Sanitary District (5 km north of the lake), at

Table 11. Monthly flow volumes and phosphorus and suspended sediment loads from Southwick and Birches Creeks and the White River, water years 1998 and 1999

[m^3, cubic meters; kg, kilograms; WY, water year]

		Southwick Creek			Birches Creek			White River		
Calendar year	Month	Flow volume (m^3)	Total phosphorus (kg)	Suspended sediment (kg)	Flow volume (m^3)	Total phosphorus (kg)	Suspended sediment (kg)	Flow volume (m^3)	Total phosphorus (kg)	Suspended sediment (kg)
1997	10	73,400	3.8	3,800	13,200	0.5	336	135,000	0.9	535
	11	80,100	2.8	3,910	25,000	.4	490	12,400	.3	0
	12	76,600	3.9	2,430	44,500	1.6	708	1,840,000	11.6	1,810
1998	1	93,600	4.7	3,070	122,000	10.6	2,010	2,660,000	13.4	3,930
	2	101,000	5.0	3,430	175,000	18.9	5,260	3,080,000	15.4	3,750
	3	10,200	5.3	4,100	166,000	8.7	1,930	169,000	1.0	236
	4	130,000	8.7	5,130	208,000	23.6	6,100	3,060,000	34.4	3,940
	5	130,000	16.3	15,300	205,000	38.3	14,400	4,540,000	53.4	10,600
	6	132,000	29.9	22,700	216,000	116.0	98,000	3,250,000	22.1	6,110
	7	98,500	8.7	4,850	137,000	63.5	51,000	2,520,000	24.3	3,760
	8	73,200	4.2	2,510	22,400	.8	463	30,600	.6	64
	9	75,000	4.3	3,330	9,930	.4	73	45,600	.8	118
	10	130,000	8.2	4,220	22,900	1.4	472	362,000	4.1	463
	11	113,000	9.1	4,030	62,900	14.5	8,400	688,000	5.7	1,320
	12	78,500	2.5	1,570	36,600	.8	118	1,310,000	8.5	2,710
1999	1	11,300	6.1	3,060	110,000	22.0	15,700	2,990,000	20.7	5,070
	2	115,000	8.7	8.820	118,000	10.6	4,960	4,400,000	25.6	3,800
	3	86,500	2.8	1,760	86,800	1.3	1,230	2,040,000	14.3	1,030
	4	160,000	14.9	22,500	257,000	122.2	104,000	2,390,000	21.4	14,000
	5	179,000	14.9	12,100	401,000	225.0	163,000	6,050,000	42.0	11,900
	6	192,000	33.4	37,300	763,000	873.0	1,110,000	9,340,000	65.7	21,400
	7	111,000	5.8	4,270	27,500	1.3	653	1,210,000	14.4	3,420
	8	78,800	4.3	2,060	9,640	.3	200	37,900	.5	181
	9	108,000	10.8	8,410	13,800	.7	91	16,900	1.7	64
WY 1998		**1,165,000**	**97.5**	**74,500**	**1,343,000**	**283.0**	**181,000**	**21,350,000**	**178.3**	**34,880**
WY 1999		**1,465,000**	**121.4**	**110,000**	**1,909,000**	**1,273.0**	**1,408,000**	**30,820,000**	**224.7**	**65,320**

Birches Creek, and in Lake Geneva (by a National Weather Service observer). Data from these three sites were used to estimate the total daily precipitation on the lake surface on the basis of a Thiessen polygon method (Linsley and other, 1958) (fig. 24). The annual precipitation on the lake surface was 840 mm (17,900,000 m^3) in WY1998 and 1,096 mm (23,300,000 m^3) in WY1999 (table 12). Direct precipitation represented approximately 48 percent of the total input of water to the lake in both years.

Surface-Water Inflow

Daily inflow data from Southwick and Birches Creeks (figs. 14 and 15) were used to estimate the total surface-water inflow to the lake. Total daily surface-water inflows to the lake were quantified by estimating contributions from base flow and contributions from surface runoff. Total daily flows from both tributaries were divided into base-flow and surface-runoff components. Base-flow/surface-runoff separation was done by a visual hydrograph-separation approach, as described by Gupta (1995, p. 302–306). The base-flow and surface-runoff components for Southwick and Birches Creeks are shown in figure 25.

Base Flow

In order that base flow from Southwick and Birches Creeks could be extended to estimate base flow from all tributaries to the lake, two base-flow synoptic studies

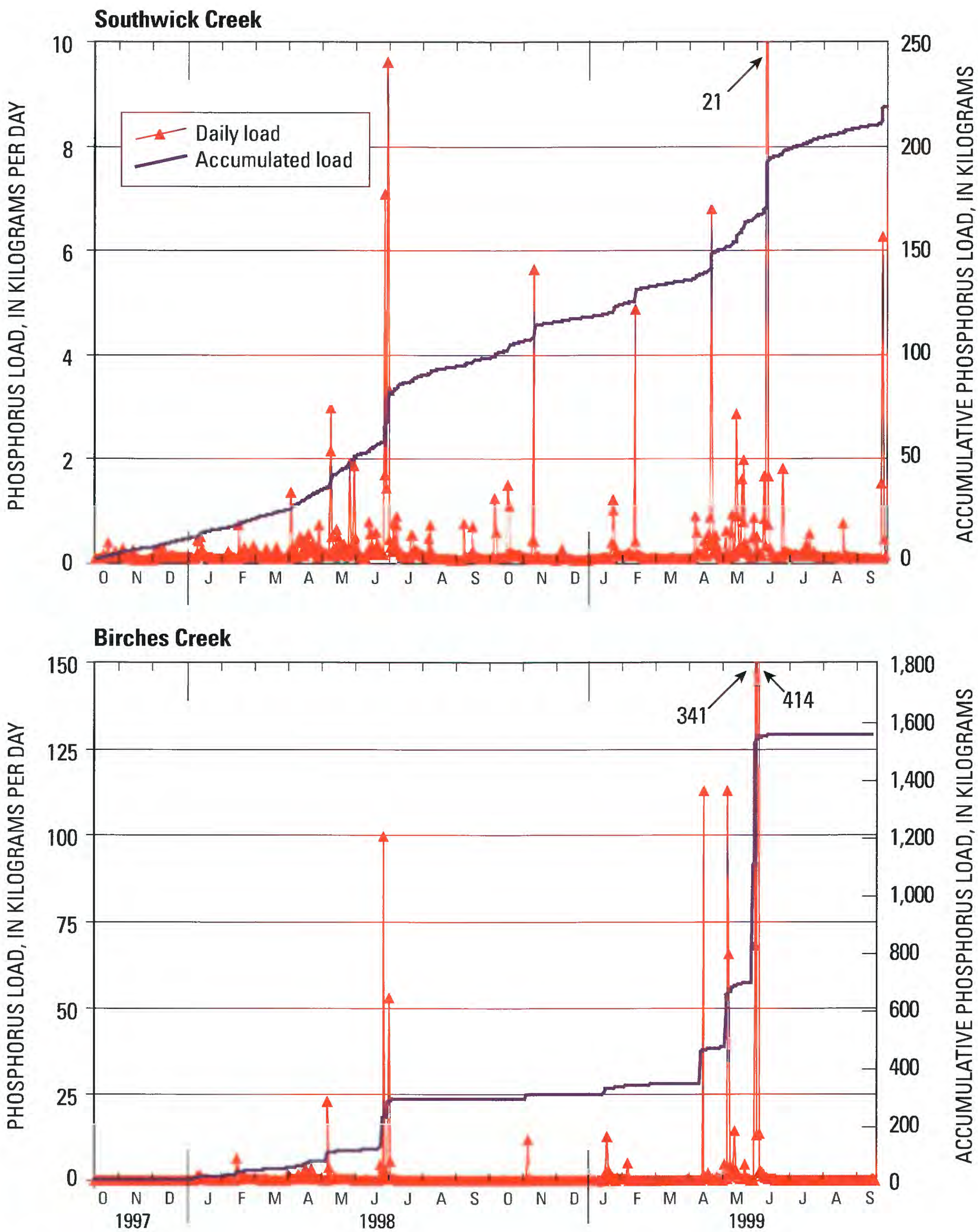


Figure 20. Daily phosphorus loads and accumulative phosphorus loads in Southwick and Birches Creeks, tributaries to Geneva Lake, Wisconsin, October 1997 through September 1999.

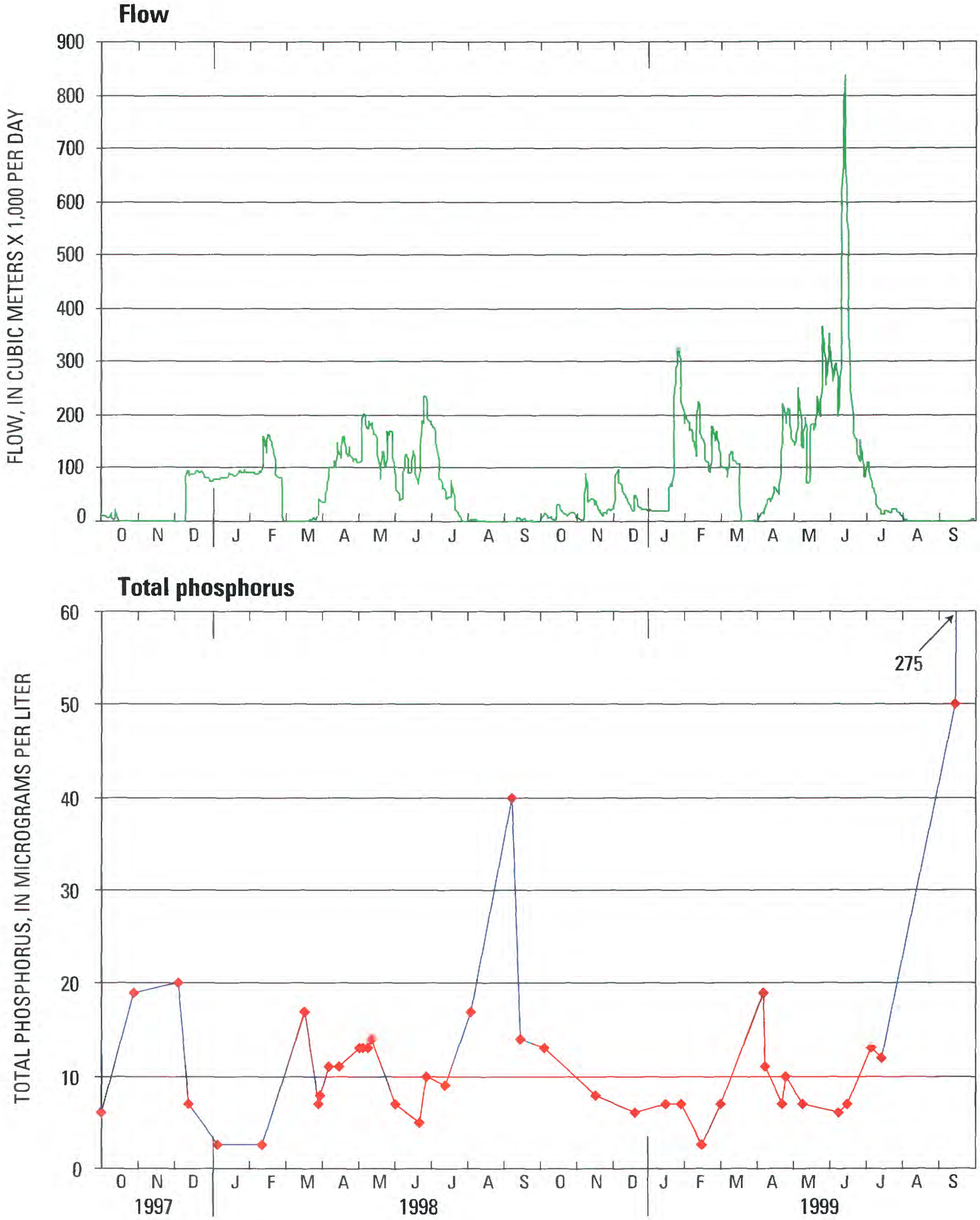


Figure 21. Flow and total phosphorus concentrations in the White River, outlet from Geneva Lake, Wisconsin, October 1997 through September 1999.

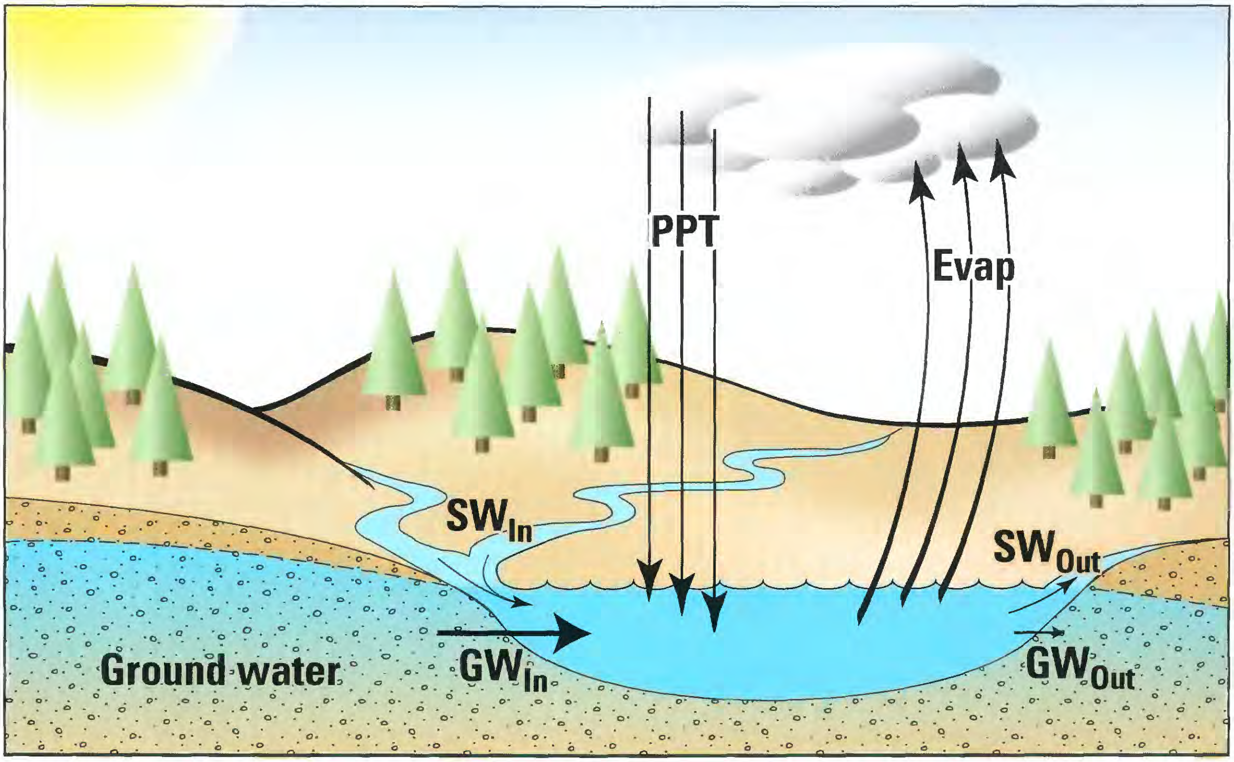


Figure 22. Schematic of the hydrologic budget of Geneva Lake, Wisconsin. (Abbreviations are defined in the text.)

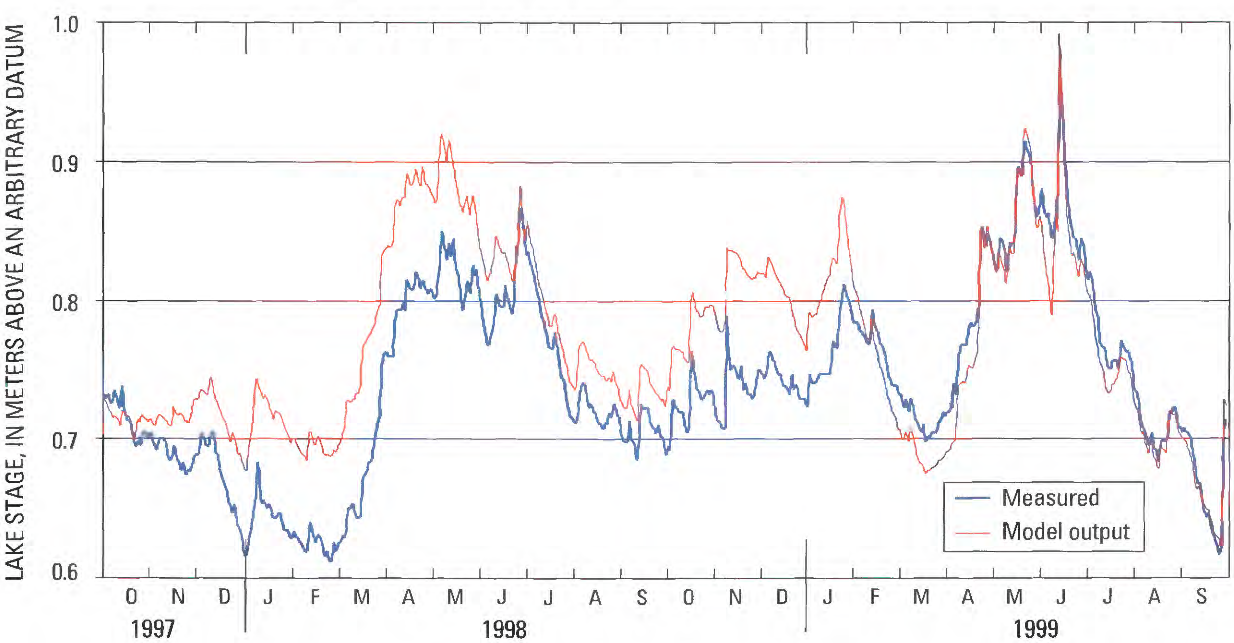


Figure 23. Daily average lake stages (above an arbitrary datum) of Geneva Lake, Wisconsin, October 1997 through September 1999. (Model output from the Dynamic Lake Model also is shown.)

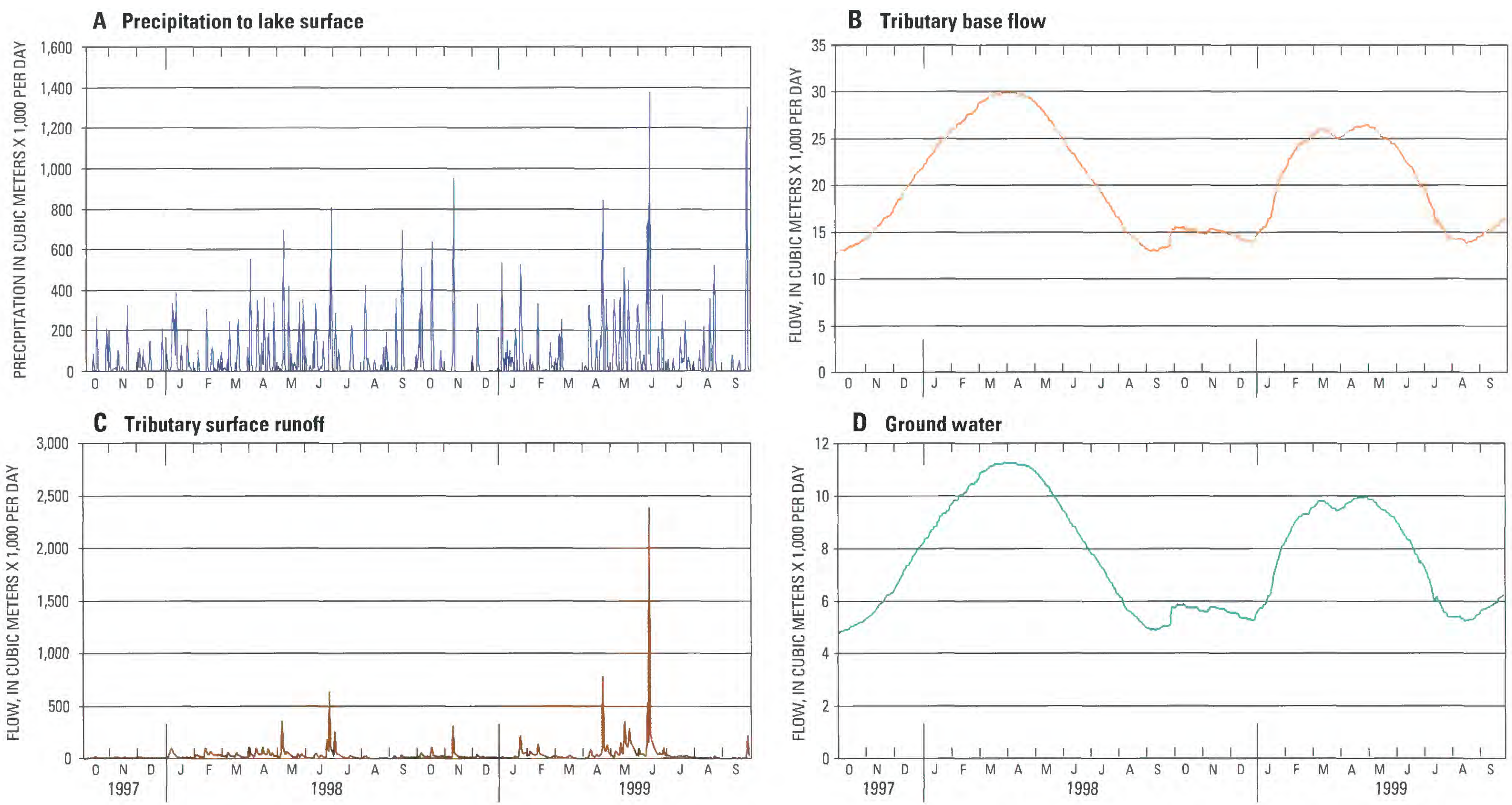


Figure 24. Daily water inputs to Geneva Lake, Wisconsin, from October 1997 through September 1999: (A) precipitation to lake surface, (B) tributary base flow, (C) tributary surface runoff, and (D) ground water.

Table 12. Hydrologic budget for Geneva Lake, Wisconsin, for water years 1998 and 1999
[All data in thousands of cubic meters or percent of total]

Budget component	1998	1998 (percent)	1999	1999 (percent)	Total	Total (percent)
			Inputs to lake			
Precipitation	17,900	48	23,300	48	41,300	48
Surface water	16,600	44	22,600	47	39,200	46
Base flow	7,780	21	7,010	14	14,800	17
Storm flow	8,850	24	15,600	32	24,400	28
Ground water[1]	2,940	8	2,650	5	5,580	6
Total input	**37,500**		**48,600**		**86,000**	
			Outputs from lake			
Evaporation	16,200	43	17,900	37	34,100	40
Surface water	21,400	57	30,800	63	52,200	60
Ground water[1]	0	0	0	0	0	0
Total output	**37,600**		**48,700**		**86,300**	

[1]It was assumed that there was no ground water leaving the lake.

were done. For each synoptic, an attempt was made to measure all flowing tributaries to the lake. During these two studies, the combined base flow in Southwick and Birches Creeks represented 16.1 and 16.6 percent of the total inflow to the lake. Therefore, the total daily base flow to the lake (fig. 24) was computed by summing the daily base flow from Southwick and Birches Creek and multiplying the sum by 6.12 (the reciprocal of 16.3 percent). The annual base flow to the lake was 7,780,000 m^3 in WY1998 and 7,010,000 m^3 in WY1999 (table 12). Base flow represented about 17 percent of the total input of water to the lake.

Surface Runoff

Surface runoff from Geneva Lake's ungaged watershed area was estimated on the basis of the surface-runoff portions of the continuously gaged record from Southwick and Birches Creeks. Daily surface-runoff yields (runoff per unit area) computed for Southwick and Birches Creeks were assumed to each represent the yields from 50 percent of the lake's ungaged watershed. Therefore, the total daily surface runoff to the lake (fig. 24) was determined as the sum of daily runoff from Southwick Creek multiplied by 9.95 (the sum of 50 percent of the ungaged watershed area plus the subwatershed area of Southwick Creek divided by the subwatershed area of Southwick Creek) and the daily runoff from Birches Creek multiplied by 4.76 (the sum of 50 percent of the ungaged watershed area plus the subwatershed area of Birches Creek divided by the subwatershed area of Birches Creek). The annual surface runoff to the lake was 8,850,000 m^3 in WY1998 and 15,600,000 m^3 in WY1999 (table 12). Surface runoff represented about 28 percent of the total input of water to the lake.

Surface-Water Outflow

All surface-water outflow from the Geneva Lake is through the White River (fig. 1). Flow from the lake is regulated by gate settings in the dam. Daily average flow during the 2-year period ranged from 49 m^3/d to 837,000 m^3/d (fig. 21). The annual outflow from the lake was 21,400,000 m^3 in WY1998 and 30,800,000 m^3 in WY1999 (table 12). Surface-water outflow represented about 60 percent of the total water leaving the lake.

Evaporation and Ground-Water Inflow and Outflow

Geneva Lake remains ice free well after air temperatures drop below 0°C. Water temperatures in the lake increase more slowly in spring and decrease more slowly in fall than in other, smaller lakes; therefore, estimating evaporation from nearby evaporation-pan data may provide misleading information. As an alternative,

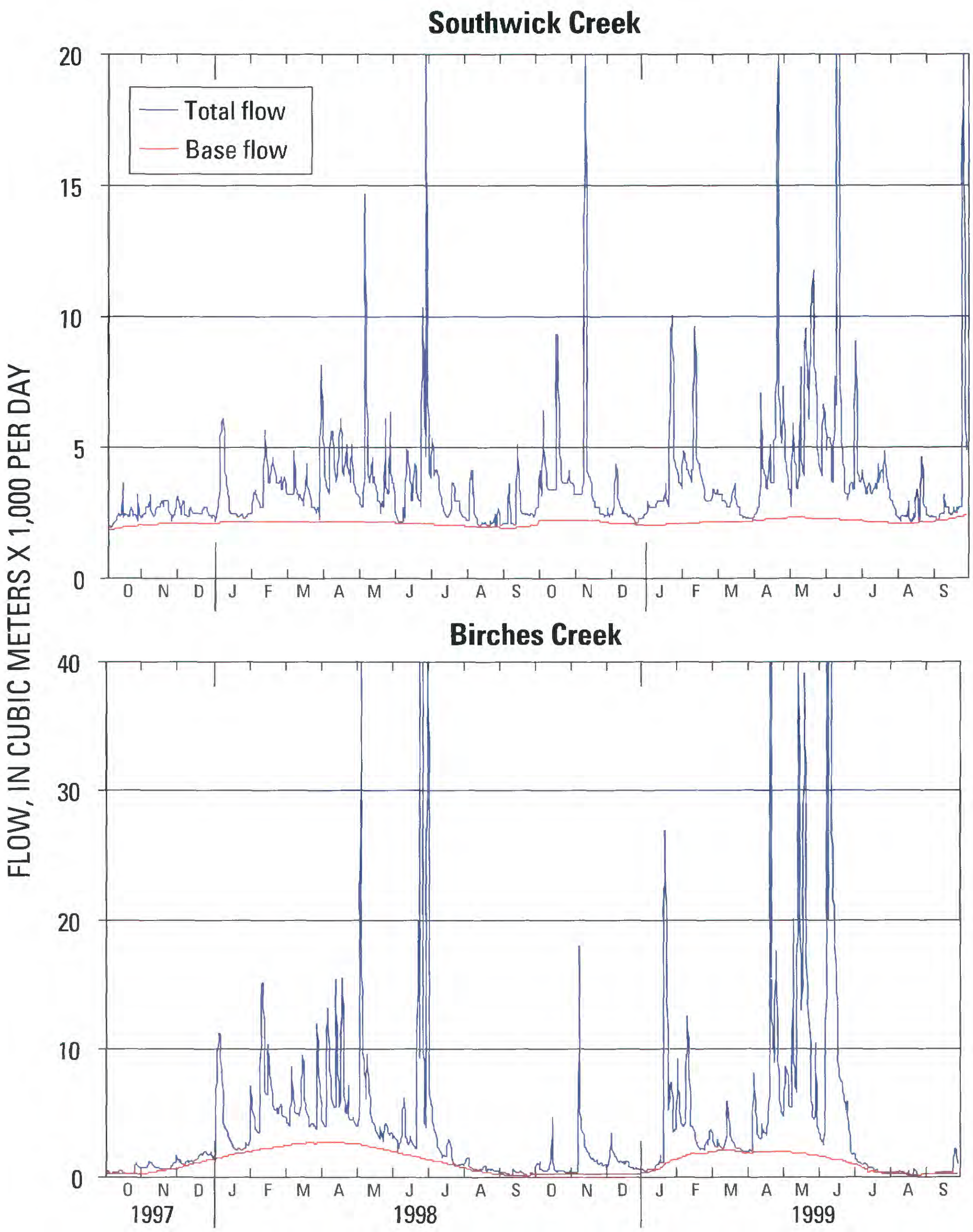


Figure 25. Flow separation into base-flow and surface-runoff components for Southwick and Birches Creeks, tributaries to Geneva Lake, Wisconsin, October 1997 through September 1999.

daily evaporation was estimated by use of a one-dimensional hydrodynamic model, the Dynamic Lake Model (DLM). In the process of estimating the hydrodynamics of the lake, the model computes detailed energy and hydrologic budgets, including evaporation. Net ground-water contributions to the lake also can be estimated by use of DLM, by examining discrepancies in modeled and measured lake levels.

DLM is a lake and reservoir model modified from the DYnamic REservoir Simulation Model, DYRESM (Imberger and Patterson, 1981) by McCord and others (2000). These models are process based and have been successfully used to simulate changes in vertical temperatures, oxygen structure, and ice cover of many lakes and reservoirs around the world. DLM is based on parameterizations of each of the individual mixing processes, so site-specific model calibration is not necessary. DLM is based on a Lagrangian layer scheme in which the water body is represented by a series of horizontal layers, each with uniform properties but variable thickness. Surface heat transfers (sensible heat transfer, evaporation, and short- and long-wave radiation) are modeled on the basis of bulk aerodynamic formula for all processes except for short-wave radiation, which is modeled as an exponential decay of energy (Wetzel, 1983). Mixed-layer deepening is modeled as convective overturn resulting from cooling and wind stirring at the surface and as seiche-induced shear and billowing resulting from shear instability at the pycnocline. Turbulent transport in the hypolimnion is modeled as a diffusion-like process, with eddy diffusivities depending on the local density gradient and rate of dissipation of kinetic energy. Inflows and outflows generally are confined to narrow vertical regions adjacent to the insertion and outflow levels of the water body (Patterson and Imberger, 1989).

Inputs to DLM include lake morphometry (table 1), meteorological data, inflow and outflow data, and initial water temperature and dissolved oxygen profiles. Daily averages of air temperature, vapor pressure, wind speed, and inflow characteristics (temperature, specific conductivity or salinity, dissolved oxygen concentration, and biological oxygen demand) are used, whereas daily total values for inflows, outflows, long-wave radiation (or average percent of clear sky), short-wave radiation, rainfall, and snowfall are used. Daily air temperature and snowfall data were obtained from a National Weather Service observer in Lake Geneva. Daily precipitation, inflow, and outflow used in the model were described earlier in this section. Daily average water temperatures of the inflow were assumed to be equal to the average air temperature of the previous 14 days (fig. 16). When the moving average air temperature was below 2.5°C, it was set to 2.5°C, approximately the average temperature observed in Southwick and Birches Creeks (fig. 16). The inflow was assumed to have a constant salinity of 330 mg/L and dissolved oxygen concentration of 8 mg/L. Total daily solar radiation and daily average wind speeds were obtained from St. Charles, Ill., approximately 75 km south of the lake. Daily average vapor pressures were obtained from a weather station at the University of Wisconsin in Whitewater, Wis., approximately 35 km northwest of the lake. An average Secchi depth of 4.5 m (the average during the open-water period during the 2-year model simulation period) was used to estimate a light-extinction coefficient of 0.38 m^{-1}. Water temperature and dissolved oxygen profiles measured in the West Bay on October 16, 1997, were used as initial conditions in the simulations.

To demonstrate that DLM correctly simulated the energy balance of the lake, the modeled thermal structure was compared with that measured. The model accurately simulated the changes in the thermal structure of the lake during the entire 2-year period. With minor modifications to the ice algorithm that was developed for lakes smaller than Geneva Lake, the model simulated the differences in ice cover in both years. The ice algorithm was modified such that the surface of the lake froze when the modeled water surface was less than 0.05°C, the daily average air temperature was colder than –6°C, and the daily average wind velocity was less than 2.5 m/s. During the 2-year simulation period, however, the modeled water level of lake gradually declined below what was measured. By the end of the 2-year simulation, the modeled water level was 262 mm below the measured water level. This difference in the volume of water in the lake was assumed to be caused by the absence of contributions from ground water.

On the basis of the altitude of the surface of Geneva Lake relative to the altitudes of water in nearby lakes and streams, it was assumed that ground water only moved from the aquifer to the lake and not from the lake to the aquifer. A water-table map in Borman (1976) indicates that ground-water gradients slope toward the lake except in a relatively small area at the northeastern end of the lake. Therefore, on the basis of the 2-year simulation, total ground-water input was estimated to be 262 mm or 5,580,000 m^3 (table 12). To determine the

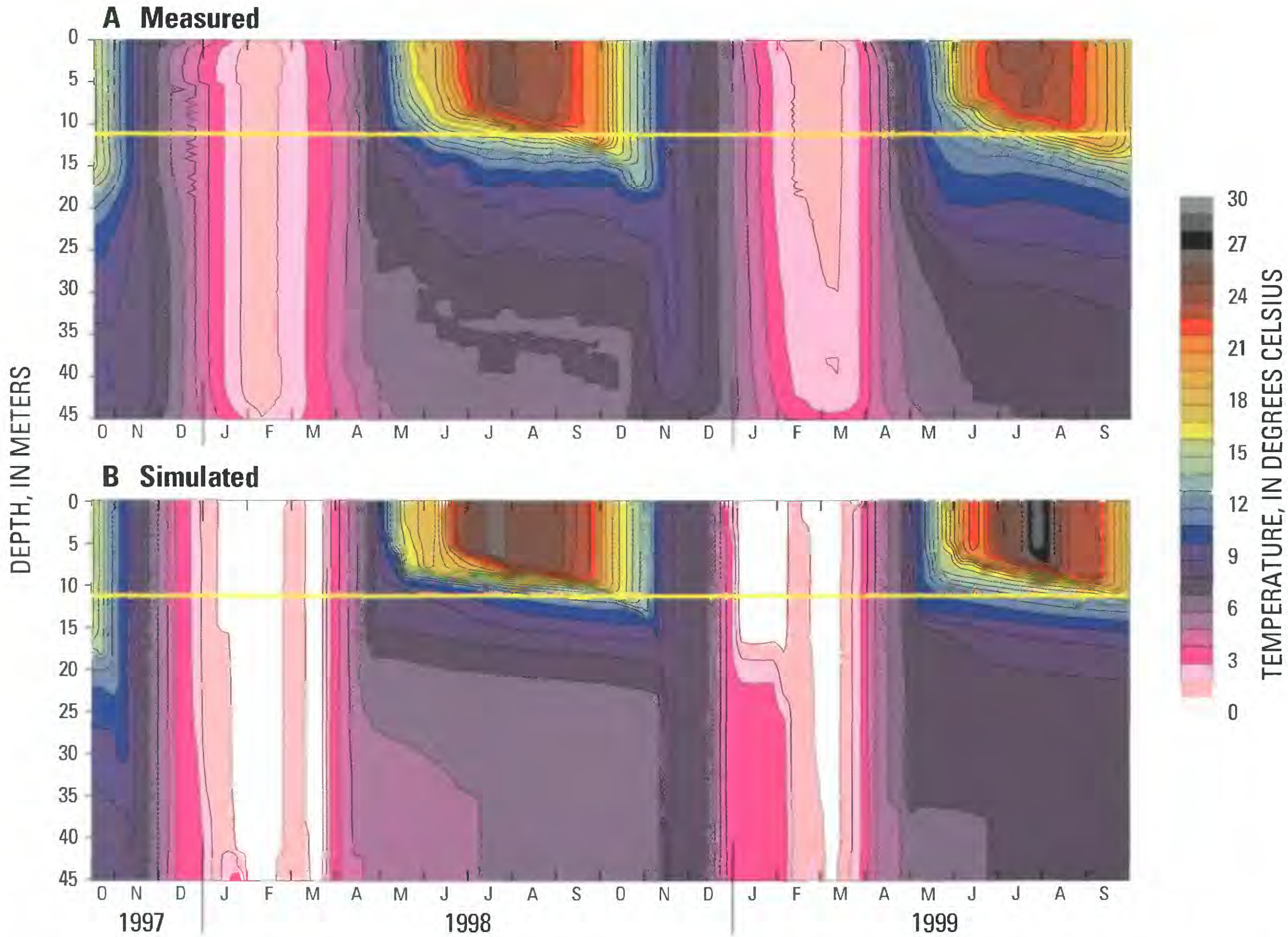


Figure 26. Water temperature distribution in Geneva Lake, Wisconsin, October 17, 1997 through September 30, 1999: (A) measured data, and (B) simulated data from the Dynamic Lake Model. (Yellow lines on plots represent the depth of the thermocline.)

daily input of ground water, the temporal distribution of ground water flowing into the lake was assumed to be similar to the temporal distribution of stream base flow. Therefore, the pattern of daily inflow from ground water was the same as that for the daily base-flow input (fig. 24). The annual ground-water input to the lake was 2,940,000 m^3 in WY1998 and 2,650,000 m^3 in WY1999 (table 12), or about 6 percent of the total inputs.

Daily ground-water inputs were then added to the total inflow to the lake, and DLM again was used to simulate the hydrodynamics of Geneva Lake from October 17, 1997, to September 30, 1999. To demonstrate that DLM correctly simulated the energy and hydrologic balances of the lake, the modeled thermal structure of the lake again was compared with that measured during this period (fig. 26). The model accurately simulated the changes in the thermal structure of the lake during the entire 2-year period (onset and breakdown of stratification, surface and deep temperatures, and the depth of the thermocline) and fairly accurately simulated the changes in the water level of the lake (fig. 23). The largest error in the hydrologic budget of the lake was less than 0.1 m.

The daily evaporation computed by DLM is shown in fig. 27. Daily evaporation was variable, but demonstrated a strong seasonal pattern, with higher evaporation in summer (especially July–August) and lower evaporation in winter. The highest daily evaporation was about 10–11 mm per day in early September. During winter, evaporation ranged from 0 to 2 mm per day. Daily evaporation during October 1–16, 1997, prior to the start of the simulation, was assumed to be about 3 mm per day (64,000 m^3/d, the average rate for the first

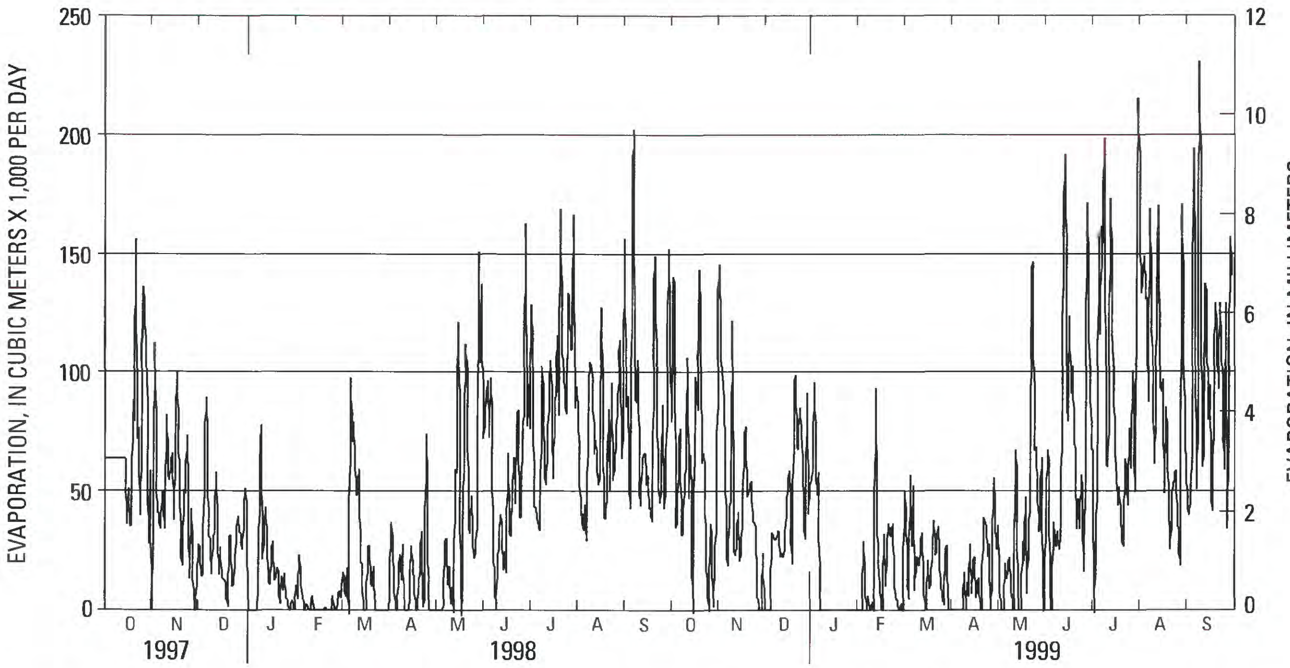


Figure 27. Daily evaporation from the surface of Geneva Lake, Wisconsin, October 1997 through September 1999, based on output from the Dynamic Lake Model.

10 days of the simulation). The annual evaporation from the surface of the lake was 760 mm (16,200,000 m^3) in WY1998 and 840 mm (17,900,000 m^3) in WY1999 (table 12). These estimates are consistent with the total annual free surface evaporation estimated for the area (approximately 890 mm) by Farnsworth and others (1982). Evaporation represented about 40 percent of the total water leaving the lake.

Summary of Hydrologic Budget

The major sources of water to Geneva Lake are precipitation (approximately 48 percent) and surface-water inflow (approximately 46 percent) (table 12 and fig. 28). The surface-water inflow was about equally divided between base flow and runoff during a normal year (WY1998), but was dominated by runoff in a wet year (WY1999). Ground water contributed about 6 percent of the water supplied to the lake. The major output of water from the lake was through its outlet (approximately 60 percent) and secondarily from evaporation (approximately 40 percent).

PHOSPHORUS BUDGET

The N:P ratios for the surface water of Geneva Lake indicated that, if just N and P are considered, P should be the limiting nutrient. Therefore, to define the main sources of P and where best to concentrate efforts to reduce P contributions to the lake, a detailed P budget was computed (eq. 2). Terms in the P budget are similar to those in the hydrologic budget, except there are additional sources of P from waterfowl and septic systems, and there is no evaporation component because evaporation does not remove P from the lake. In addition, P stored in the sediments of the lake may be released to the water column through equilibrium reactions and by the action of microorganisms. Although the internal loading of P from the sediments is not a new source to the lake/sediment system, it can be important to the P available in the lake; therefore, it also was examined.

$$\text{Total P Input} = (PPT + SW_{In} + GW_{In} + \text{Waterfowl} + \text{Septic Systems}) - (SW_{Out} + GW_{Out}) \qquad (2)$$

Precipitation

To estimate the P supplied to the lake by precipitation on the lake's surface, the daily precipitation vol-

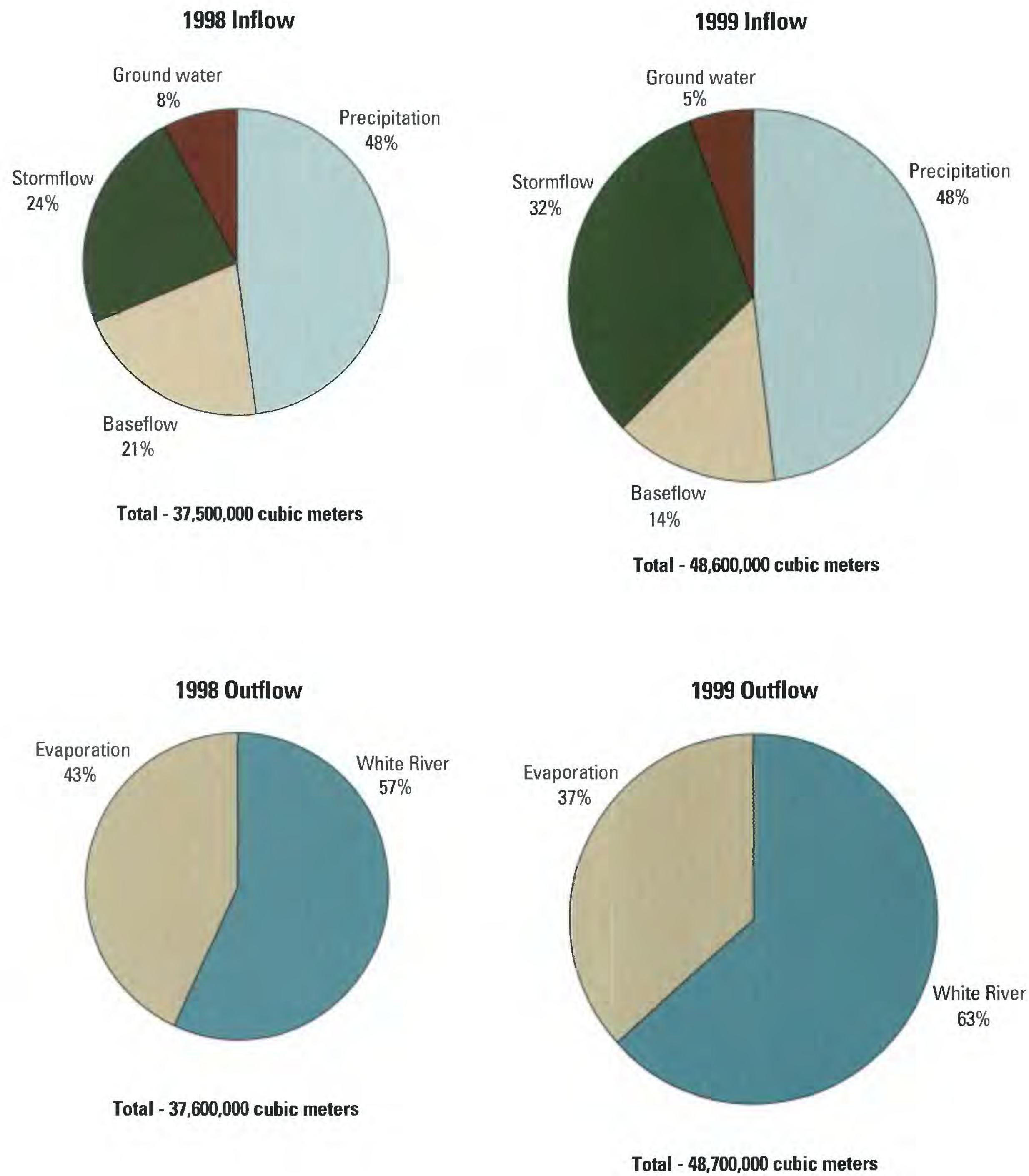


Figure 28. Hydrologic budget for Geneva Lake, Wisconsin, water years 1998 and 1999. [%, percent]

Table 13. Phosphorus inputs to Geneva Lake, Wisconsin, water years 1998 and 1999
[All data in kilograms or percent of total]

Source	1998	1998 (percent)	1999	1999 (percent)	2-year total	Total (percent)
Precipitation	358	11.2	467	5.5	825	7.0
Tributaries	2,341	73.0	7,559	88.6	9,900	84.4
Southwick Creek	98	3.0	121	1.4	219	1.9
Birches Creek	283	8.8	1,273	14.9	1,556	13.3
Buttons Bay Creek	373	11.7	425	5.0	798	6.8
Unmonitored areas	1,587	49.5	5,739	67.3	7,326	62.4
Base flow	165	5.2	149	1.7	314	2.7
Runoff	1,421	44.4	5,590	65.6	7,012	59.8
Ground water	23	.7	21	.2	45	.4
Septic systems	91	2.8	91	1.1	182	1.6
Geese	390	12.2	390	4.6	780	6.6
Total input	**3,204**		**8,528**		**11,731**	

umes (fig. 24) were multiplied by a P concentration of 20 µg/L. Field and Duerk (1988) determined that the average concentration of bulk precipitation at the Delavan Lake Sanitary District (5 km north of the lake) was about 20 µg/L in WY1984 and WY1985. The annual input of P from precipitation on the surface of the lake was 358 kg in WY1998 and 467 kg in WY1999 (table 13), or 7 percent of the total P input to the lake (fig. 29).

Surface-Water Inflow

Loadings of P from Southwick and Birches Creeks (fig. 20) were used to estimate all tributary loading of P except that from Buttons Bay Creek. Daily P loadings from Buttons Bay Creek were computed from the P concentration time series (fig. 17) and the flows estimated as for other ungaged areas (described in more detail in the following paragraph). In estimating the surface-water loading to the lake from all areas except Southwick, Birches, and Buttons Bay Creeks, daily P loading was estimated for base flow and surface-runoff events separately.

Buttons Bay Creek

Daily flows for Buttons Bay Creek were estimated the way that flows were estimated for all ungaged areas as described in the "Hydrologic Budget" section. Daily base flows were computed with the assumption that flow from Buttons Bay Creek represented approximately 6 percent of the total extrapolated base flow, the average percentage that Buttons Bay Creek represented relative to the total discharge from the 24 additional streams measured during the two base-flow synoptic studies. Surface-runoff flows were computed with the assumption that Buttons Bay Creek represented approximately 11 percent of the total extrapolated runoff, the percentage of the ungaged area in the Buttons Bay subwatershed. The amount of water from base flow and surface runoff then were summed and multiplied by the daily P concentrations for Buttons Bay Creek (fig. 17) to obtain daily P loads. Daily concentrations were estimated by linear interpolation between the measured concentrations from all years. The annual input of P from Buttons Bay Creek was 373 kg in WY1998 and 425 kg in WY1999, or about 7 percent of the total P input to the lake (table 13).

Base-flow Contributions from Unmonitored Areas

Daily P contributions from base flow for the remainder of the ungaged areas were computed by multiplying the average P concentration from the 23 sites (all synoptic sites except Buttons Bay Creek) measured during the two base-flow synoptic studies, times the daily base flows estimated for the ungaged areas (total base flow, shown in fig. 24, less that from Southwick, Birches, and Buttons Bay Creeks). The average P concentration (27 µg/L) was the average of the two flow-weighted mean concentrations (25 and 28 µg/L) from

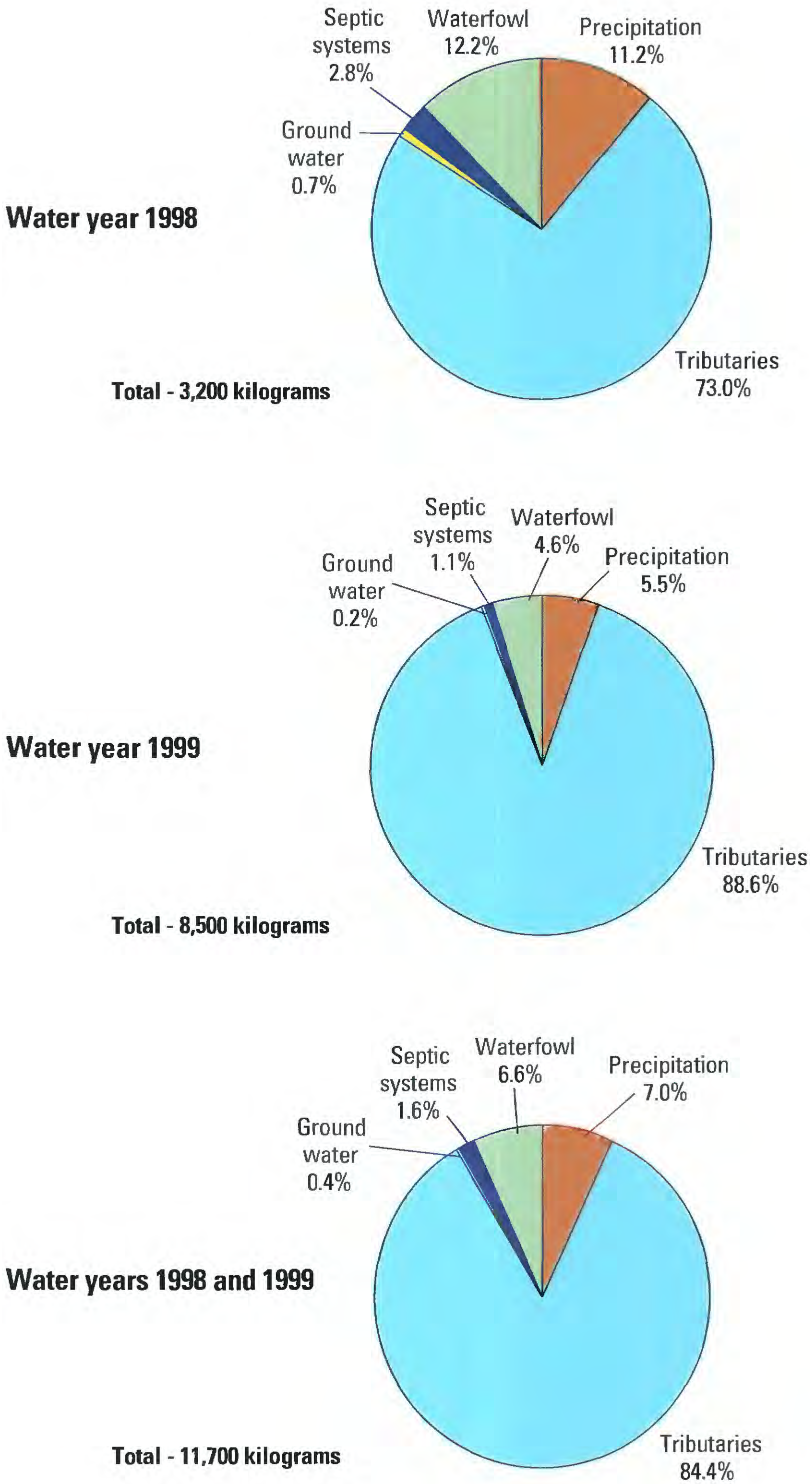


Figure 29. Phosphorus budget for Geneva Lake, Wisconsin, water years 1998 and 1999, and that for both years combined. [%, percent]

Table 14. Allocation of tributary phosphorus load to land-use/land-cover categories in the Geneva Lake, Wisconsin, watershed
[km², square kilometers per square meter; kg/yr, kilograms per year; kg/ha, kilograms per hectare; --, not applicable]

Source	Area (km^2)	Percent of basin	Percent of watershed	Export rate (kg/ha)	Phosphorus load (kg/yr)	Percent of tributary load	Percent of total load
Lake surface	21.2	28.5	--	[1]0.17	360	--	11.2
Cropland	14.5	19.5	27.3	[2]0.61	883	37.7	27.6
Pasture/grassland	8.0	10.8	15.0	[3]0.30	239	10.2	7.5
Moderate and high intensity urban[7]	3.2	4.3	6.1	[4]0.96	308	13.2	9.6
Rural residential[7]	1.4	1.9	2.7	[5]0.45	64	2.7	2.0
Barren/shrubland	4.2	5.6	7.8	[3]0.30	125	5.3	3.9
Golf courses	1.2	1.6	2.3	[6]0.61	74	3.2	2.3
Forest	18.1	24.5	34.2	[3]0.30	544	23.2	17.0
Wetlands	1.4	1.9	2.7	[6]0.61	87	3.7	2.7
Streams	1.0	1.3	1.8	[1]0.17	17	.7	.5
Ground water	--	.0	--	--	23	--	.7
Septic systems	--	.0	--	--	91	--	2.8
Waterfowl	--	.0	--	--	390	--	12.2
Total basin area	74.2	**100.0**		**Total tributary load**	**2,342**		
Total watershed area	53.0			**Total phosphorus load**	**3,205**		

[1]Based on a phosphorus concentration of 20 micrograms per liter and the precipitation in water year 1998.
[2]Based on yields (loads per unit area) for agricultural areas from Corsi and others (1997).
[3]Pasture/grassland, barren/scrubland, and forest all assumed to be 0.30 kg/ha.
[4]Based on yields for extensive urban basins from Corsi and others (1997).
[5]Rural residential assumed to have a yield midway between that for agricultural and forested areas.
[6]Golf courses and wetlands assumed to have similar yields as cropland.
[7]Low intensity urban areas were equally divided into moderate-intensity urban and rural-residential areas.

the two synoptic studies. The flow-weighted mean concentration was computed by dividing the sum of flow times concentration for each site by the sum of the flows at each site. The annual input of P from base flow from the unmonitored areas was 165 kg in WY1998 and 149 kg in WY1999, or about 3 percent of the P input to the lake (table 13).

Surface Runoff

Daily P contributions from the remainder of the ungaged areas from surface runoff were computed by extrapolating the surface-runoff loads from Southwick and Birches Creeks, such that each site was extrapolated to 50 percent of the ungaged areas excluding the Buttons Bay subwatershed. Daily runoff loads for Birches and Southwick Creeks were determined by multiplying the daily runoff flow for each site by the average daily P concentration (daily P load divided by daily flow volume). The daily runoff loads for Southwick Creek then were multiplied by 8.95 (the yield from Southwick Creek was used to compute the load from an unmonitored area 8.95 times the size of its own monitored subwatershed) and the loads for Birches Creek were multiplied by 3.76 (the yield from Birches Creek was used to compute the load from an unmonitored area 3.76 times the size of its own monitored subwatershed). The annual input of P from surface runoff from unmonitored areas was 1,421 kg in WY1998 and 5,590 kg in WY1999, or 62.4 percent of the P input to the lake (table 13).

Total Tributary Loading

Tributary input was the main source of P to the lake, contributing about 84 percent of the total P load during the study (table 13 and fig. 29). The total daily tributary loading was computed by summing the contributions from Southwick, Birches, and Buttons Bay Creeks and that from unmonitored areas from base flow and surface runoff. The annual input of P from all tributaries was 2,341 kg (44.2 kg/km^2) in WY1998 and 7,559 kg (143 kg/km^2) in WY1999. Tributaries contributed about 73 percent of the annual P load in WY1998 and 89 percent in WY1999.

Of the total tributary load, Southwick Creek contributed only between 2 and 4 percent, Birches Creek contributed between 12 and 17 percent, and Buttons Bay Creek contributed between 6 and 16 percent. Buttons Bay Creek contributed a higher percentage of the total load during WY1998, the drier of the 2 years. The remaining ungaged areas contributed between 68 and 76 percent of the total tributary load.

Allocation of Tributary Loading

To describe where the P in the tributaries originated, the total tributary load for WY1998 (the more typical hydrologic and loading year of the 2 years sampled) was allocated to various land uses/land covers on the basis of published export rates (unit-area loads) and the percentage of the watershed with various land uses/land covers (table 14; fig. 30). Corsi and others (1997) published export rates from small watersheds in southeastern Wisconsin: average export rates for predominantly agricultural basins was 0.61 kg/ha and for urban basins was 0.96 kg/ha. To estimate the loading from other land-use/land-cover categories, the unit-area loads shown in table 14 were used. Wetlands and golf courses were assumed to have export rates similar to that for agricultural land (0.61 kg/ha) because streams draining these types of areas were found to have some of the highest P concentrations in the study area of all tributaries monitored. Pasture, grassland, barren land, and forests were all assumed to have an export rate of 0.30 kg/ha. Export rates from forests were assumed to be relatively high because of the amount of development occurring in these areas. The export rate for rural residential land was assumed to be midway between rates for agriculture and forested areas (0.45 kg/ha). The small area covered by streams was assumed to have the same export rate (or deposition rate) as the surface of the lake (0.17 kg/ha, computed based on a concentration of 20 µg/L of P in precipitation and the total precipitation for 1998). Loads computed on the basis of these export rates provided a load similar to that estimated from the tributaries in WY1998.

The total contribution from each land-use/land-cover category is shown in figure 30. In general, the contributions from various land-use/land-cover categories are similar to the percentages of land covered by the various categories; however, urban and agricultural areas contribute a higher percentage and forested areas contribute a lower percentage than their respective land areas. Of the overall tributary load, agricultural cropland and fields contributed about 50 percent, forested areas contributed about 25 percent, and urban areas contributed about 16 percent (table 14 and fig. 30).

Ground Water

Phosphorus loading to the lake from ground water was computed from the estimated daily flows described in the "Hydrologic Budget" section and an estimated constant P concentration. The estimated constant P concentration of ground water (8 µg/L) was based on the lowest concentrations measured at Birches and Southwick Creeks during base-flow periods, when flow mostly was ground-water discharge to the streams. The daily ground-water loads of P then were computed as the product of daily average flow and 8 µg/L. The annual input of P from ground water was 23 kg in WY1998 and 21 kg in WY1999, or only about 0.4 percent of the P input to the lake (table 13 and fig. 29).

Septic Systems

Phosphorus loading from near-shore septic systems also was considered in the P budget. The input of P from septic systems was estimated from equation 3 (Reckhow and others, 1980):

$$M = E_S * (\text{Number of Capita Years}) * (1 - S_R), \quad (3)$$

where M is the input of P, E_S is an export coefficient, S_R is a soil-retention coefficient, and the number of capita years is the equivalent number of annual residents. A range in the loading from septic systems was estimated on the basis of the number of capita years (computed from the number of homes in the watershed with septic systems and an assumed number of residents), a typical septic tank output (E_s, 0.5 kg per capita per year; Reckhow and others, 1980), and a range in the P retained by the soil. It was estimated that there

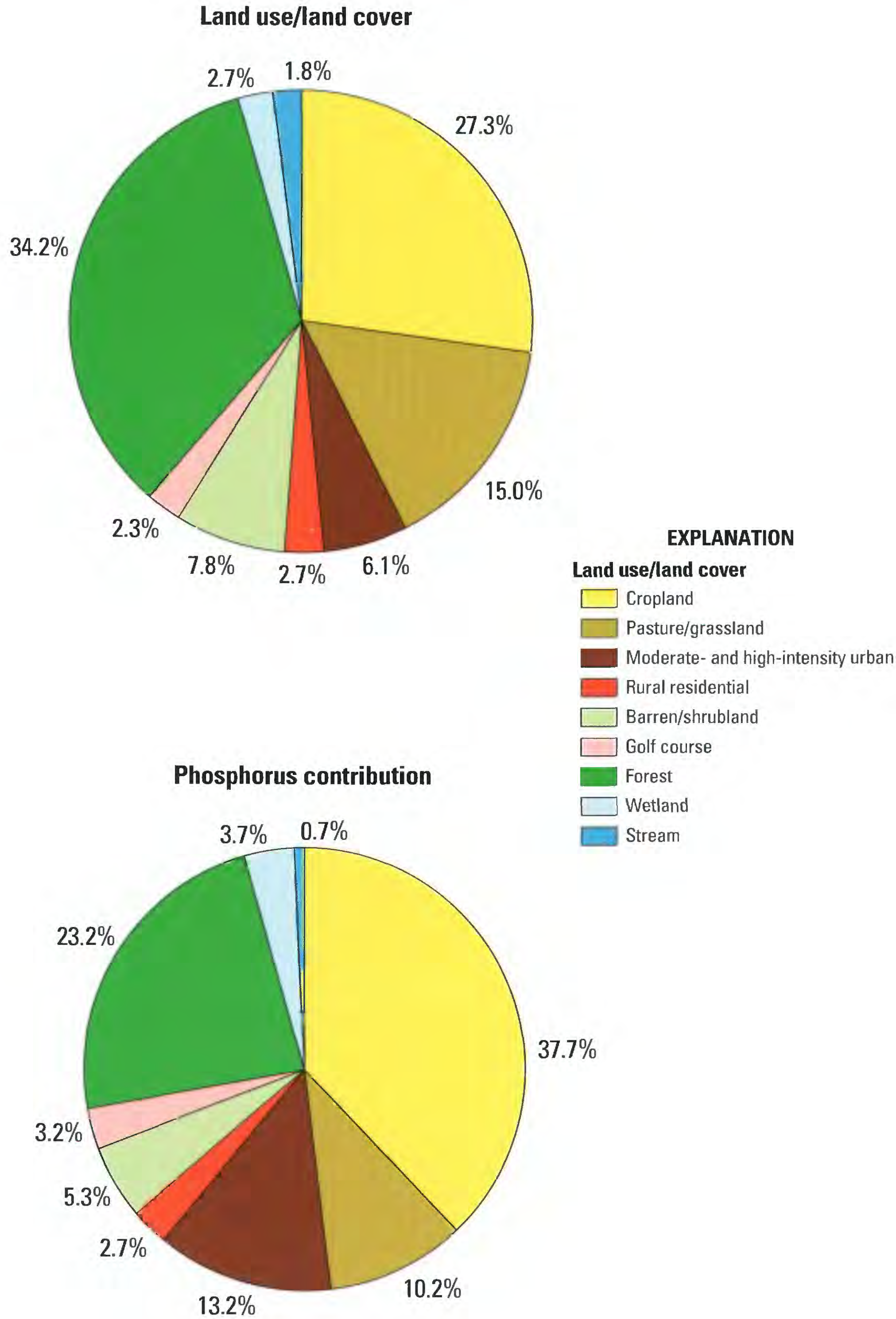


Figure 30. Distribution of land use/land covers in the Geneva Lake, Wisconsin, watershed and allocation of phosphorus load from tributaries to Geneva Lake to land uses/land covers in the watershed. [%, percent]

were 1,634 homes in the Linn Sanitary District that encompasses most of the unincorporated land within the watershed. The number of homes with septic systems was estimated from data issued by the Linn Sanitary District, and permits issued by Walworth County since 1982 (T. Peters, Geneva Lake Environmental Agency, written commun., 2000). From records of 1,309 residences, it was estimated that 985 homes had septic systems. For the homes not accounted for in these estimates, it was assumed that 70 percent had septic systems. Therefore, there were about 1,200 septic systems in the watershed. This represents a maximum number of contributing systems because many of them are not near-shore systems. To determine the total capita using the septic systems, it was assumed that each household had three people using the septic system for half of the year. For a well-functioning septic system, typically about 90 percent of the P released from a septic system is retained by the soil; however, for a failing septic system, this amount can be much less. Therefore, if all of the septic systems were assumed to be well functioning, it is estimated that 91 kg per year would be contributed to the lake (table 13). If it is assumed that only half were well functioning and the other half were failing totally, it is estimated that about 500 kg per year would be contributed to the lake. Therefore, septic systems contribute between 2 and 8 percent of the total P contributed to the lake. In all of the following computations and percentages for other sources, it is assumed that all of the septic systems were well functioning and, therefore, septic systems represent about 2 percent of the P contributed to the lake (fig. 29).

Waterfowl

Geneva Lake has a large population of waterfowl, especially migratory Canada Geese during winter. To consider the potential impact from waterfowl, their input was included in the P budget. No direct measurements of loading from waterfowl were made in this study; therefore, published rates were used. To estimate the contributions from waterfowl, the number of days waterfowl were on the lake was needed and various assumptions had to be made. It was assumed that Canada Geese were the only waterfowl contributing P to the lake, that nutrients consumed by the geese (that is, the food source) came from outside of the lake and, therefore, represent additional loading that would not be included in other sources, that the amount of P released per goose was a constant at 0.56 g/d per goose (the average of two published values: 0.62 g/d per goose (Grimillion and Malone, 1986) and 0.49 g/d per goose (Marion and others, 1994)), and that all of the P released by the geese is input into the lake. The number of geese on the lake changes seasonally. Average monthly goose populations (table 15) were approximated from estimates made by WDNR district wildlife managers and visual estimates made by GLEA (T. Peters, Geneva Lake Environmental Agency, written commun., 2000). Monthly estimates ranged from as low as 200 geese per day in late summer to 10,000 geese per day in January. From this information, the total P loading from geese was 390 kg per year (about 12 percent of the total for WY1998), or about 7 percent of the P input to the lake for the two years (table 13 and fig. 29).

Table 15. Estimated phosphorus loading from Canada Geese into Geneva Lake, Wisconsin

Month	Average number of geese per day	Number of goose days[1]	Phosphorus load (kilograms)
January	10,000	310,000	172.1
February	2,000	58,000	32.2
March	1,000	31,000	17.2
April	400	12,000	6.7
May	300	9,300	5.2
June	300	9,000	5.0
July	200	6,200	3.4
August	200	6,200	3.4
September	500	15,000	8.3
October	1,000	31,000	17.2
November	2,000	60,000	33.3
December	5,000	155,000	86.0
Total	**22,900**	**702,700**	**390.0**

[1]Number of geese per day multiplied by average monthly goose populations.

Internal Release From Hypolimnetic Sediments

Phosphorus stored in the lake sediments (table 8) may be released back to the water column through reduction/oxidation reactions and by the action of bacteria, fungi, and other organisms. The rate of P release from the sediments is increased greatly when the sediment-water interface becomes anoxic (Wetzel, 1983), such as in the hypolimnion of Geneva Lake (fig. 6). Phosphorus released from hypolimnetic sediments typically results in a steady increase in P concentrations above the sediment-water interface after the onset of anoxia. This increase in P concentrations was observed near the bottom in the West Bay, but very little increase was observed in other areas of the lake (fig. 8).

To quantify the internal P loading in Geneva Lake, the accumulation of P in the hypolimnion was computed for summer stratification periods (1997, 1998, and 1999). Total P accumulation in the hypolimnion was computed as the P mass in the hypolimnion in fall (before breakdown of stratification) minus the P mass in the hypolimnion in the preceding April at the onset of stratification. The P profiles and lake morphometry (table 1) were used to compute the P mass in the hypolimnion at the times of interest. It was assumed that P concentrations changed linearly with depth between vertical P measurements, that the total accumulation from internal loading occurred below 15 m, and that P export from the hypolimnion to the epilimnion during summer and P released during winter and from oxic areas of the lake were small. The net accumulation of P in the hypolimnion was -1,150 kg in 1997, +68 kg in 1998, and +35 kg in 1999. During summer 1997, the accumulation of P just above the bottom of the lake was more than compensated for by a loss of P in the water column between 15 to 30 m, and resulted in a net loss of P in the hypolimnion. A decrease in P concentrations in the area from 15 to 30 m may have been caused by particulate P settling into deeper zones of the lake. During summers of 1998 and 1999, the losses of P from 15 to 30 m approximately balanced with the accumulations in the deepest zones of the lake. It has been shown that, in some lakes, the high P concentrations just below the thermocline can be released to the epilimnion during seiches of the thermocline (Stauffer and Lee, 1973). However, in mid to late summer, P concentrations in Geneva Lake were uniformly low down to about 30 m and, therefore, seiches would cause little or no net P movement around the thermocline and into the epilimnion. Therefore, it was assumed that internal loading of P did not contribute additional P to the lake and did not contribute to the productivity of the lake.

Summary of Phosphorus Budget

By combining all of the sources of P to Geneva Lake, the total annual input of P was about 3,200 kg in WY1998 and about 8,500 kg in WY1999 (table 13). During this study, the major source of P to Geneva Lake was from its tributaries (table 13 and fig. 29), which contributed about 84 percent of the total input (ranging from 73.0 percent in WY1998 to 88.6 percent in WY1999). Most of this load was delivered to the lake during surface-runoff events. Greater precipitation and more intense storms were the main reason for the increased P loads in WY1999 compared to WY1998. Contributions from waterfowl and precipitation were the second and third most important sources of P, each contributing about 7 percent of the total input (ranging from about 5 to 12 percent). Septic systems were estimated to have contributed between 2 and 8 percent of the total P input. There is a wide range in the estimated contributions from septic systems because of uncertainty in how present systems were installed (for example, proper soils), how well they are maintained and functioning, and how far they are from the lake. Contributions of P from ground water and lake sediment were minor.

In order to determine whether and how much P loading to the lake has changed since implementation of the original management plan for Geneva Lake (Southeastern Wisconsin Regional Planning Commission, 1985), the P budget (and estimates for the individual sources) from WY1998 (a year with near normal precipitation, 53 mm less than normal) was compared with that estimated for 1975 (another year with near-normal precipitation, 23 mm less than normal) (table 5 and fig. 31). In 1975, the total estimated P input was about 6,000 kg (Southeastern Wisconsin Regional Planning Commission, 1985); however, it appears that this total included an over-estimated input from precipitation. If it is assumed that the input of P from precipitation has not changed from 1975 to 1998, then the total estimated P load to the lake in 1975 would have been about 5,200 kg. Hence, in WY1998, P loading was about 2,000 kg less (approximately 38 percent) than that measured in 1975.

The major differences in P loading to Geneva Lake from 1975 to 1998 were associated with the decrease in loading from the Fontana sewage-treatment plant, the increased loading from more waterfowl inhabiting the lake, and an apparent decrease in loading from septic systems. In 1975, the effluent from the Fontana sewage-treatment plant was released into Buena Vista Creek; however, with the discontinuation of the plant and the diversion of its influent, P concentrations in Buena

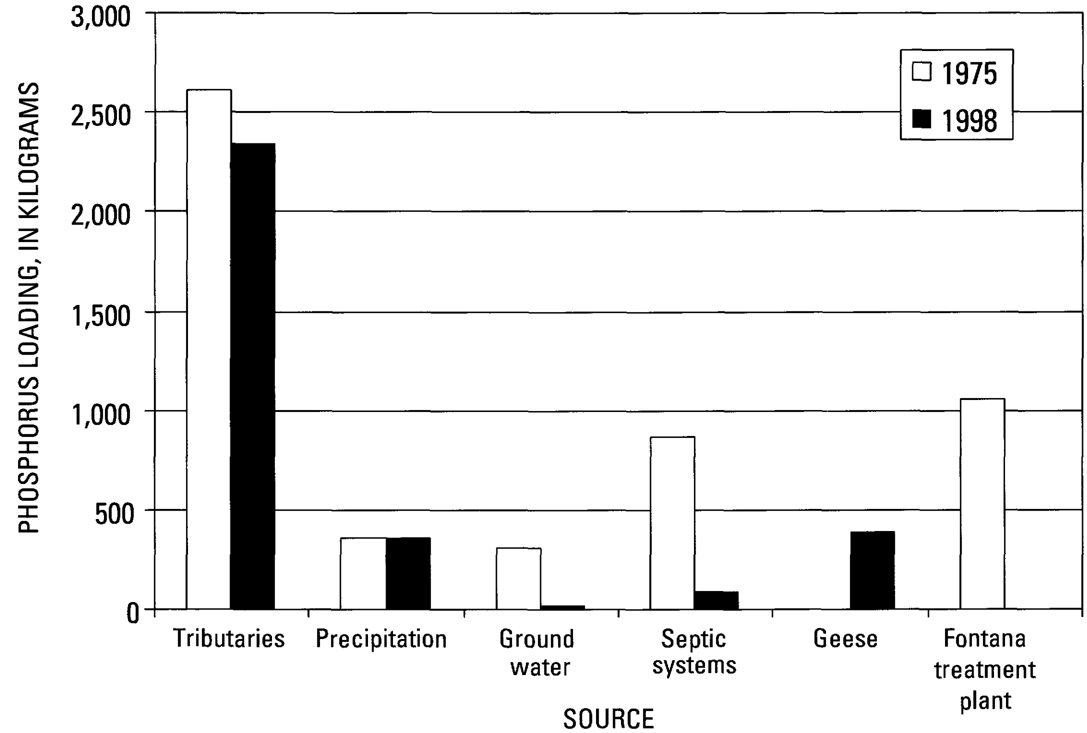


Figure 31. Comparison of phosphorus loading from individual sources in water year 1998 with that estimated for calendar year 1975 for Geneva Lake, Wisconsin. (Data for 1975 from Southeastern Wisconsin Regional Planning Commission, 1985.)

Vista Creek were reduced. In this study, P concentrations were not observed above 50 µg/L (table 10), compared to routinely measured concentrations of 1,500 µg/L in 1976 (T. Peters, Geneva Lake Environmental Agency, written commun., 2000). Contributions from waterfowl were not included in the 1975 P budget, but waterfowl were expected to be very minor contributors in 1975 because the number of waterfowl inhabiting the lake was quite low compared to recent years (T. Peters, Geneva Lake Environmental Agency, oral commun., 2000). The apparent decrease in loading from septic systems from 1975 to 1998, shown in figure 31, results because of the assumption that all of the septic systems are now fully functioning. If 50 percent were failing totally, then the estimated load from septic systems would be about 500 kg, an additional 400 kg. Septic systems in WY1998 then would have contributed about 55 percent of that estimated from septic systems in 1975.

NUTRIENT-REDUCTION SCENARIOS

The total P input into Geneva Lake was estimated to be 3,200 kg in WY1998 and 8,500 kg in WY1999. One way to determine how much of this P would need to be eliminated to improve the water quality of the lake, compared to that measured in this study (figs. 7–11), is by means of empirical models that relate P loading with measured water quality. These models were developed by comparing hydrologic and P loading from many different systems to measures describing lake-water quality, such as near-surface P and chlorophyll *a* concentrations and Secchi depth. Thirteen of these empirical models are contained within the Wisconsin Lakes Modeling Suite (WiLMS; J. Panuska, Wisconsin Department of Natural Resources, written commun., 1999).

Of the 13 models contained in WiLMS, 9 were applicable to the hydrologic system of Geneva Lake (table 15). Therefore, the morphometry of the lake and the hydrologic and P loadings for WY1998 (the more typical hydrologic and loading year of the 2 years sampled) were input into these models to simulate near-surface P concentrations for WY1998. Various P-reduction scenarios then were input into these models to simulate the effects of changes in near-surface P concentrations. The average P concentration simulated by the models for WY1998 was 24 µg/L (table 16), higher than that measured in the lake (about 10 µg/L at spring overturn

Table 16. Estimated near-surface total phosphorus concentrations in Geneva Lake, Wisconsin, from hydrologic and phosphorus loading models contained in the Wisconsin Lakes Modeling Suite for various scenarios of total phosphorus loading

[Wisconsin Lakes Modeling Suite from J. Panuska, Wisconsin Department of Natural Resources, written commun., 1999; data are concentrations, in micrograms per liter; %, percent; kg, kilograms; µg/L, micrograms per liter]

	Scenario				
	Full loading 1998	**Base scenario (27% reduction from 1998)**	**25% reduction**	**50% reduction**	**85% increase (1975 loading)**
Total phosphorus loading (kg)	**3,200**	**2,350**	**1,755**	**1,170**	**4,350**
Lake phosphorus model					
Canfield and Bachman, 1981 (natural lakes)	23	19	15	12	31
Canfield and Bachman, 1981 (artificial lakes)	24	20	18	14	30
Rechow, 1979 (general lakes)	12	9	6	4	19
Rechow, 1977 (anoxic lakes)	35	25	19	13	56
Walker, 1977 (general lakes)	12	9	6	4	19
Vollenweider, 1982 (general lakes)	36	27	20	13	59
Vollenweider, 1982 (shallow lakes)	24	19	15	10	36
Larsen and Mercier, 1976 (general lakes)	19	15	11	8	30
Nurnberg, 1984 (oxic lakes)	32	23	17	12	52
Average of models	**24.1**	**18.4**	**14.1**	**10.0**	**36.9**
Reduction from base scenario, µg/L		**0.0**	**4.3**	**8.4**	**-18.5**
Reduction from base scenario, percent	**-31**	**0**	**24**	**46**	**-100**

and during summer; table 5). In general, these empirical relations were calibrated with P loading data that did not include contributions from precipitation, ground water, waterfowl, and septic systems. Therefore, the contributions from these sources were removed (total P input was reduced from 3,200 kg to 2,350 kg) and the reduced loading was again input into the empirical relations. The average P concentration then simulated by the models for WY1998 was 18 µg/L (table 16), closer to but still higher than that measured in the lake. The models then were applied with P loading from various P-reduction scenarios: 25-percent reduction (1,755 kg of P input) and 50-percent reduction (1,170 kg of P input) in tributary loading (table 16). The models then simulated the average P concentration in the lake to decrease by 4 and 8 µg/L, respectively, or a reduction in P concentration of 24 and 46 percent, respectively, from that simulated with 2,350 kg P loading (base scenario). If it is assumed that the results from the models only should be used to determine the percentage of change in total P rather than absolute concentrations, then it is predicted that average near-surface P concentration should decrease from 10 µg/L to 7.7 µg/L in response to a 25-percent reduction in P loading and should decrease to 5.4 µg/L in response to a 50-percent reduction in P loading.

Empirical relations also have been developed that simulate average summer chlorophyll *a* concentrations and Secchi depths from near-surface P concentrations. Empirical relations developed for Wisconsin lakes by Lillie and others (1993), also contained within WiLMS, were applied to determine how 25- and 50-percent reductions in P loading should affect near surface chlorophyll *a* concentrations and Secchi depths in Geneva Lake. With the measured average summer P concentration of 10 µg/L, the models simulated a chlorophyll *a* concentration of 5.1 µg/L (compared to 3–3.5 µg/L measured in WY1998; table 6) and a Secchi depth of 3.0 m (compared to 3.9–4.2 m measured in WY1998; table 6). The predicted average summer P concentrations of 7.7 µg/L and 5.4 µg/L were then input into the empirical models. With a near-surface P concentration of 7.7 µg/L (25-percent reduction scenario), the models simulated a chlorophyll *a* concentration of 4.4 µg/L (a 15-percent reduction from 5.1 µg/L simulated for 10 µg/L of P) and a Secchi depth of 3.3 m (an increase of 10 percent from 3.0 m simulated for 10 µg/L of P). With a near-surface P concentration of 5.4 µg/L (50-percent reduction scenario), the models simulated a chlorophyll *a* concentration of 3.7 µg/L (a 28-percent reduction from that simulated for 10 µg/L of P) and a

Secchi depth of 3.9 m (an increase of 30 percent). If it is again assumed that the models should be used only to determine the percentage of change, then on the basis of these model results, it is predicted that average chlorophyll *a* concentrations would drop from the measured summer average of 3.4 µg/L to 2.9 µg/L with a 25-percent reduction in P loading and to 2.5 µg/L with a 50-percent reduction in P loading. On the basis of these model results, it also is predicted that the average summer Secchi depth would increase from 4.0 m to 4.4 m with a 25-percent reduction in P loading and to 5.2 m with a 50-percent reduction in P loading.

Therefore, decreases in P loading to the lake of 25 and 50 percent should improve the water quality of the lake. Phosphorus concentrations in the lake should decrease by almost the same percentage as the decrease in P loading rates, whereas chlorophyll *a* concentrations should decrease and Secchi depths should increase but by smaller percentages (about 10 and 30 percent for 25- and 50-percent P load reductions, respectively).

HISTORICAL CHANGES IN ICE COVER AND WATER QUALITY

Ice Cover

Freeze and breakup dates and total duration of ice cover for the lake have been recorded from 1862 to the present (appendix). The annual data and 10-year moving average for each of these records are shown in figure 32. Around 1862, when the record began, the lake typically froze completely over around December 20. Beginning around 1880, freeze dates gradually became later until about 1935, when the typical freeze date was around January 20. Freeze dates then again became earlier until about 1950 when the lake typically froze over around January 1. Freeze dates remained relatively stable until just recently, when they again became later. The typical freeze date currently is around January 10. In the winter of 1997–98, the lake did not completely freeze for the first time in the 139-year record. For computation of moving averages and graphing, the freeze date for the winter of 1997–98 was set to March 10, the latest observed freeze date. Even during the period around 1935 when the 10-year moving average freeze date was later than any other period, the lake always froze completely over. No consistent long-term changes in freeze dates were observed when the entire period of record was examined; however, the period from about 1905 to 1940 had unusually late freeze dates, an indication that air temperatures in fall and early winter were higher than normal and higher than they are currently.

Throughout most of the record, ice cover has typically broken up around the beginning of April. Starting around 1980, however, breakup dates have become earlier. The most recent 10-year moving average breakup date is March 12, about 24 days earlier than in most of the 139-year record. For computation of moving averages and graphing, the breakup date for the winter of 1997–98 was set to January 3, the earliest observed breakup date. The most recent change toward earlier breakup dates was not observed when the breakup dates of Geneva Lake were examined by Magnuson and others (2000) because they did not include the period after 1995. The recent change toward earlier breakup dates, however, is consistent with that observed in lakes throughout the northern hemisphere (Magnuson and others, 2000).

The total duration of ice cover also has changed during the past 139 years. Total ice duration gradually decreased from about 110 days around 1862 to about 65 days around 1935 (caused by later freeze dates), then increased back to about 100 days around 1965. Most recently, the total duration again has become shorter, with a 10-year moving average of about 70 days (caused by both later freeze dates and early breakup dates). During 1997–2001, the average duration was only 49 days and included two of the four shortest ice durations in the 139-year record.

The most recent changes in freeze and breakup dates and total ice duration are indicative of higher fall, winter, and spring air temperatures, especially higher spring temperatures. These changes in air temperature and ice cover are consistent with what would be expected with climatic warming (Robertson, 1989). Although the most recent data suggest warming, examination of the entire record indicates the present climate is not much different from that around 1935, except that recent spring air temperatures appear to be higher than those around 1935 and that late-fall and early-winter air temperatures appear to be lower than they were around 1935. The increased number of waterfowl inhabiting the lake may be associated with the change in climate and the decreased length of ice cover.

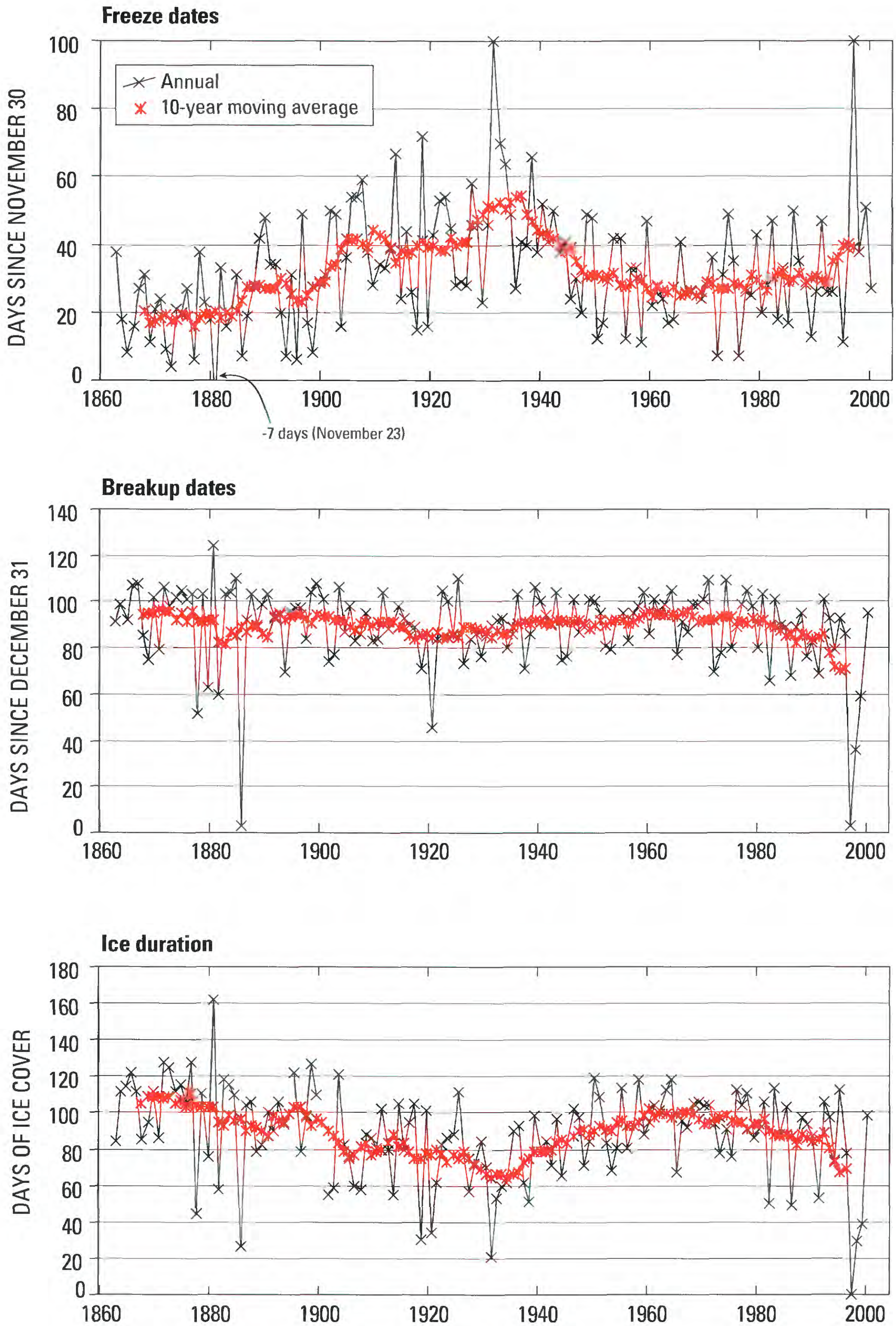


Figure 32. Annual and 10-year moving averages of freeze dates, breakup dates, and total duration of ice cover for Geneva Lake, Wisconsin, 1862 to 2001.

Measured Changes in Water Quality

Water-quality data in Geneva Lake have been collected intermittently since 1901 by scientists from the University of Wisconsin, WDNR, GLEA, and the USGS. These data primarily describe water temperature, dissolved oxygen, and Secchi depths; however, a few water-chemistry analyses also are available for chemical constituents such as total P. The data presented below were collected for various reasons and by means of different types of devices, and analytical techniques also have differed. Therefore, the data quality is difficult to ascertain, and changes in the values of any of these measurements only should be used as indicators of relative changes in the lake rather than absolute changes.

Water Temperature

If climatic warming has occurred during the summer months, surface-water temperatures during summer should have increased. Surface-water temperatures in Geneva Lake were measured as early as 1901; however, routine annual monitoring did not begin until about 1968. Measurements made in July, August, and September (the 3 months with the most measurements) are shown in figure 33, along with trend lines computed by means of linear regression. The trend lines indicate that July water temperatures may have increased, August water temperatures may have decreased, and September water temperatures have not changed throughout the period. However, because of the large annual variability in these measurements caused by annual climatic variability and the day in the month the measurements were made, it is uncertain whether any long-term changes in surface temperatures have occurred since 1900.

Dissolved Oxygen

Various changes in a lake may be reflected by changes in dissolved oxygen concentrations. Increases in water temperature may result in decreased near-surface dissolved oxygen concentrations because the solubility of dissolved oxygen decreases with higher water temperatures, whereas increases in lake productivity may cause near-surface dissolved oxygen concentrations to increase during the daytime and decrease at night. Increases in productivity may also cause dissolved oxygen concentrations in the hypolimnion to decrease more rapidly throughout the summer. Dissolved oxygen measurements in Geneva Lake were measured as early as 1905; however, routine annual monitoring also did not begin until about 1968. Measurements were most consistently made in July, August, and September and usually were measured at about 5-m increments throughout the water column. To demonstrate changes in dissolved oxygen concentrations from 1906 to 2000, measurements made in August at the surface, at 20 m and at 30 m are shown in figure 34, along with trend lines computed by means of linear regression. Surface dissolved oxygen concentrations made after 1940 in all summer months (shown only for August) were higher than those measured prior to 1910, possibly indicative of increased productivity. Dissolved oxygen concentrations made at 20 and 30 m after 1940 were slightly higher than those measured prior to 1910 (fig. 34); however, because of the few number of measurements made prior to 1910, it was difficult to determine whether these changes were statistically significant. These data do not demonstrate a long-term decline in dissolved oxygen in the hypolimnion indicative of a decline in water quality in Geneva Lake.

Total Phosphorus

Although P concentrations often regulate or limit the productivity of a lake, few direct historical measurements of P were available for Geneva Lake. The older measurements that were available are of questionable quality because of analytical problems associated with measuring the low P concentrations in the lake. Total P measurements that were considered to be reliable were first measured in 1972 (Stauffer, 1974), next as part of the 1976–77 study by SEWRPC and GLEA, then as routine measurements by GLEA from 1981–94, and finally those in this study. The most consistent measurements were collected as part of spring sampling efforts (fig. 35). These measurements indicate that P concentrations in the lake were about 15 to 25 μg/L prior to the mid-1980s. The concentrations then dropped to about 10 μg/L from about 1988 to about 1993, then increased to about 20 μg/L in 1995, and then decreased to about 6 to 13 μg/L since 1995. These changes suggest that the current productivity in the lake may be higher than that prior to about 1988.

Chlorophyll *a*

Chlorophyll *a* concentrations are a direct indication of the standing crop of algae in a lake. Summer-average (June through August), near-surface chlorophyll *a* concentrations indicate extensive annual variability but no long-term changes from 1981 to the present (fig. 36).

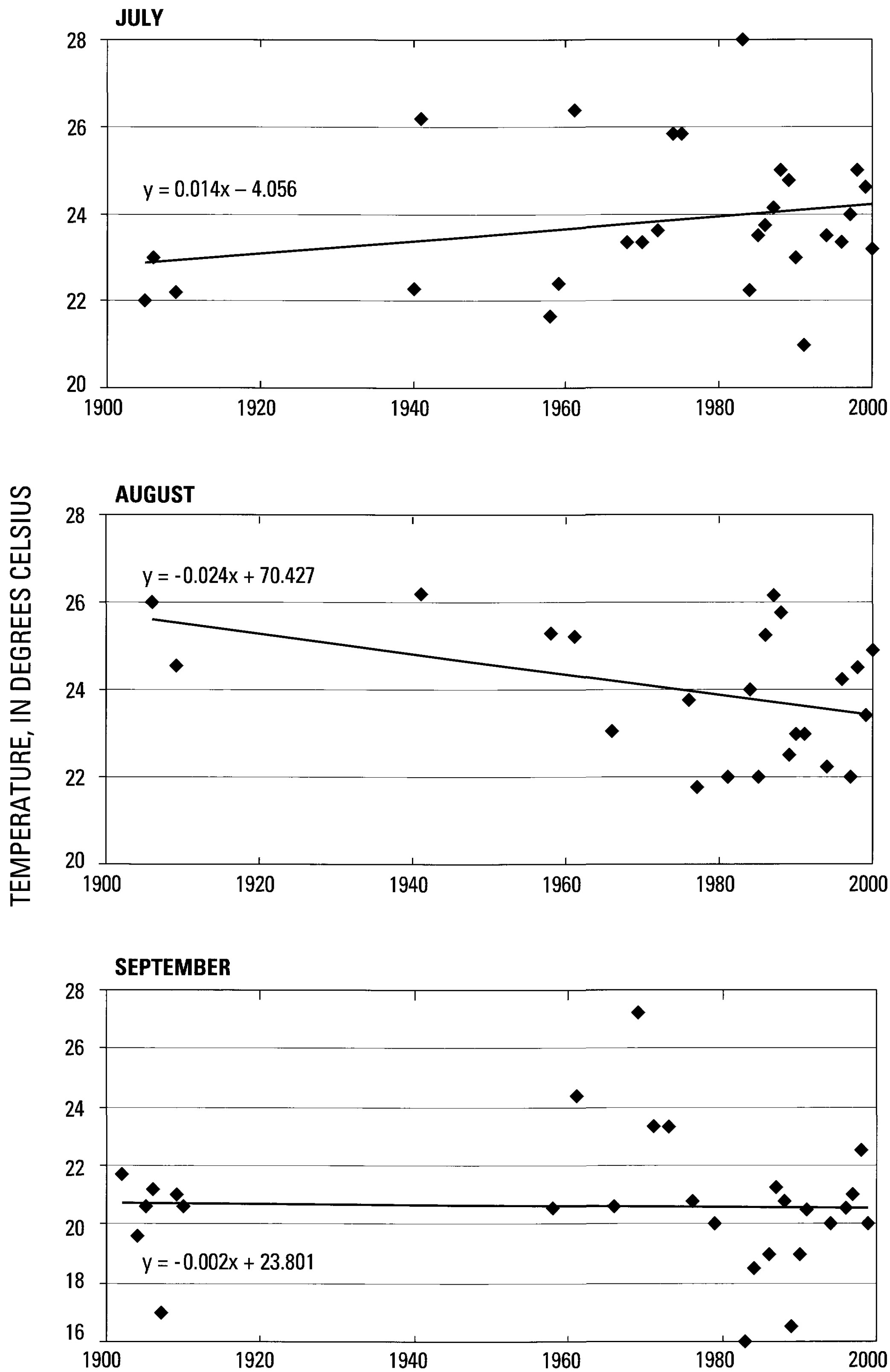


Figure 33. Near-surface water temperatures measured in Geneva Lake, Wisconsin, during July, August, and September, 1902 to 2000. (Trend lines are based on linear regression through the data.)

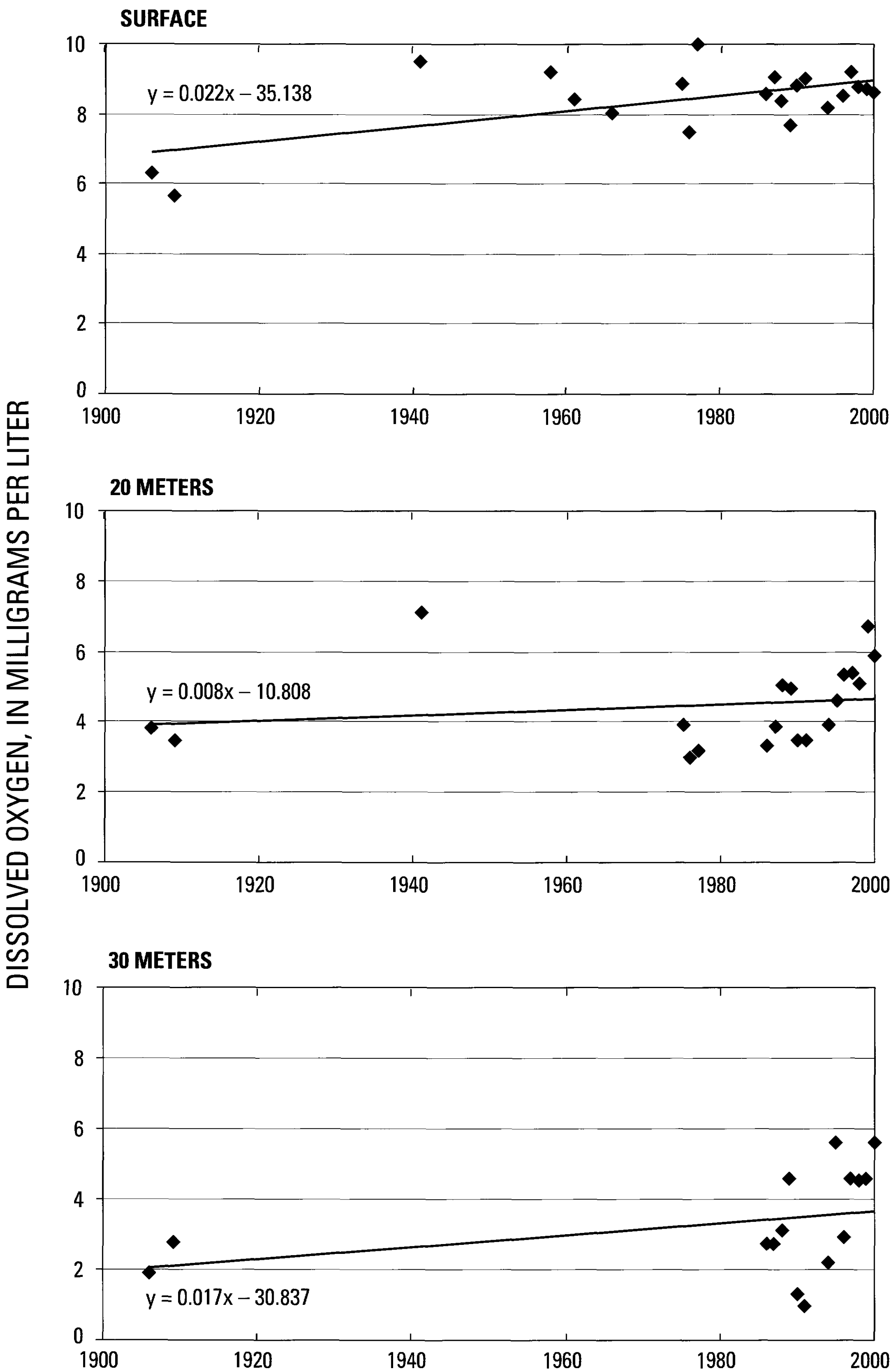


Figure 34. Dissolved oxygen concentrations measured in Geneva Lake, Wisconsin, during August at near-surface, 20-meter, and 30-meter depths, 1906 to 2000. (Trend lines are based on linear regression through the data.)

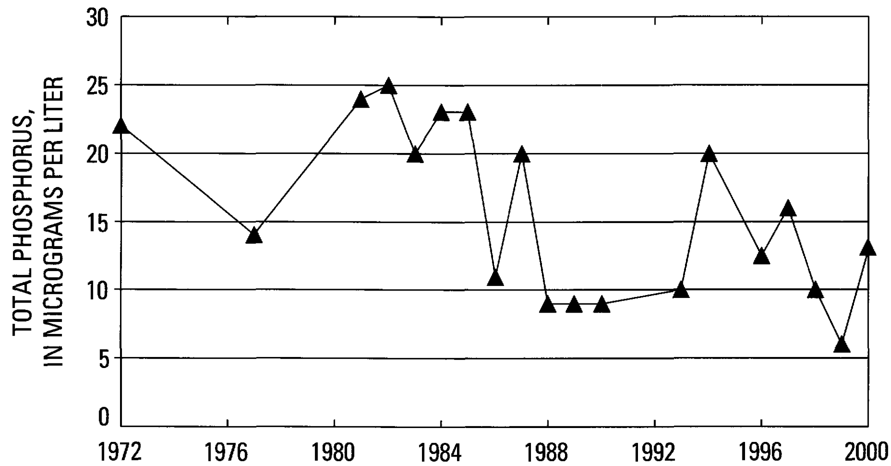


Figure 35. Near-surface total phosphorus concentrations measured at spring overturn in Geneva Lake, Wisconsin, 1972 to 2000.

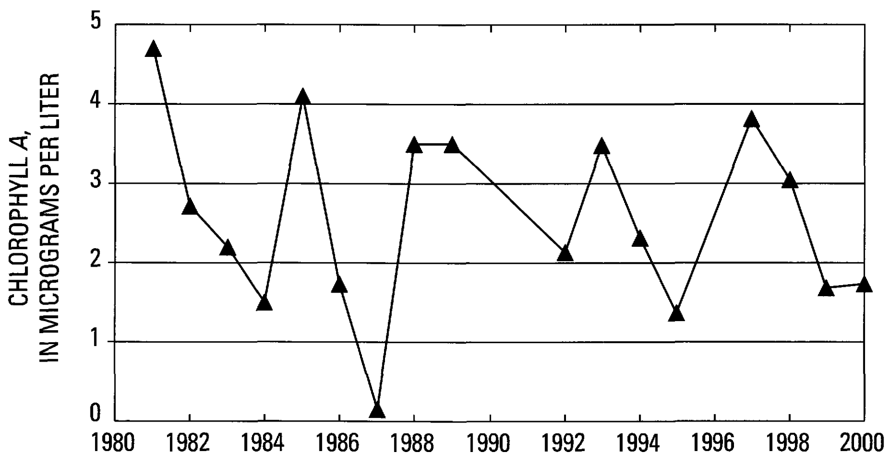


Figure 36. Summer-average (June through August) near-surface chlorophyll *a* concentrations in Geneva Lake, Wisconsin, 1981 to 2000.

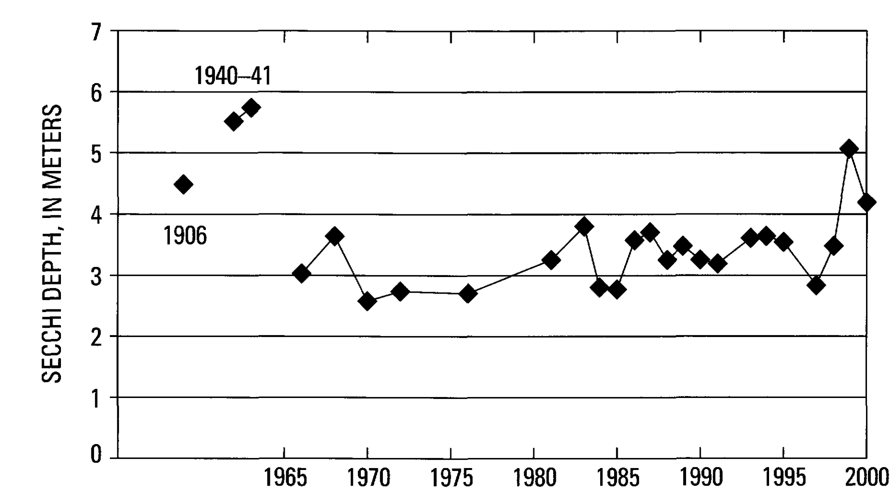


Figure 37. Average July/August Secchi depths in Geneva Lake, Wisconsin, 1906 to 2000.

These concentrations indicate that the trophic status of the lake has varied from mesotrophic to oligotrophic.

Secchi Depth

A Secchi depth measurement is an indicator of water clarity and algae/chlorophyll *a* concentrations in a lake. It may be one of the least sophisticated limnological measurements, but probably it is least prone to measurement biases and can indicate much about the trophic status of a lake. Secchi depth measurements in Geneva Lake were measured as early as 1904; however, routine monitoring did not start until about 1966. Measurements were most consistently made in May, July, August, and September. To demonstrate the long-term changes in summer Secchi depths, July and August measurements were averaged and are shown in figure 37. Only seven summer measurements from 5 different years were made prior to 1966; therefore, it is difficult to quantify changes prior to 1966. From 1966 to 1998, average July/August Secchi depths ranged from 2.6 to 3.8 m, all indicated that the lake was mesotrophic. During 1966–98, the 3 years with the poorest clarity were from 1970, 1972, and 1976. In 1999 and 2000, the average July/August Secchi depths improved to 5.1 and 4.2 m, respectively, indicating oligotrophic conditions in the lake. The most recent July/August Secchi depths are similar to that measured in 1906 (4.5 m) and slightly less than those measured in 1940–41 (5.5 and 5.8 m). These measurements suggest the water clarity was worse and productivity was higher from the 1960s to late 1990s (especially the 1970s) than the early 1900s and 1999–2000, when the lake was classified as oligotrophic.

It is difficult to ascertain whether the increase in water clarity during 1999–2000 was caused by the decrease in P concentrations in the lake and an associated decrease in algal productivity, the effects of the recent increase in zebra mussel populations, a foodweb change associated with a change in the fish community, or a combination of these factors. If the increase in water clarity only were due to decreased P concentrations, the clarity would be expected to be similar to that in the late 1980s and early 1990s, when P concentrations also were 10 µg/L or less. If the increase only were due to zebra mussels, the water clarity in 2000 would be expected to be similar to or better than that in 1999 because zebra mussel populations in the lake have been increasing. The most recent improvement in water clarity appears to be caused by a combination of the decrease in P concentrations associated with a decrease in P loading to the lake (primarily caused by the Fontana sewage diversion) and the effects of the increase in zebra mussel populations.

Historical Changes in Loading to Geneva Lake and Changes in Water Quality, as Inferred From Sediment Cores

Water-chemistry measurements available for Geneva Lake do not enable much description of the nutrient concentrations in the lake before 1972 and the productivity in the lake before about 1968. However, information contained in the bottom sediments of the lake represent a natural record of the lake history and enable much to be inferred about historical changes in sediment and nutrient loading to the lake and water chemistry in the lake. Sediment cores were extracted from three locations in the lake (center of the West Bay, Geneva Bay, and Williams Bay; fig. 1) and used to reconstruct the water-quality history of the lake for the last 170 years, a period encompassing the time from the end of habitation by Native Americans and the entire time following European settlement. A detailed analysis of these cores was described in a report entitled "Paleoecological study of Geneva Lake, Walworth County," by Garrison (2000); therefore, only a summary of the findings is presented here.

Changes in isotopic activities throughout the core were used to age-date specific depths in the core and to quantify changes in sedimentation rates in the various parts of the lake (fig. 38). The mean sedimentation rate in the deepest part of the lake (0.032 $g/cm^2/yr$ in West Bay) was low compared to other hardwater lakes in southern Wisconsin (ranging from 0.018 to 0.120 $g/cm^2/yr$) and was relatively constant for the last 170 years. However, the sedimentation rates in Geneva and Williams Bays (0.041–0.042 $g/cm^2/yr$) were similar to other hardwater lakes in Wisconsin. In all three cores, sedimentation rates peaked between 1900 and about 1920, then dropped to rates similar to pre-1850, before increasing slightly in recent years. In both bays, the high sedimentation rates were measured over a short period, around 1900, when much development occurred around the bays. In the West Bay, the greatest influx of sediment was measured a little later, from about 1920 to 1940. In the bays, especially Williams Bay, the input of soil erosional materials currently is greater than it was before European settlement. Much of the increased

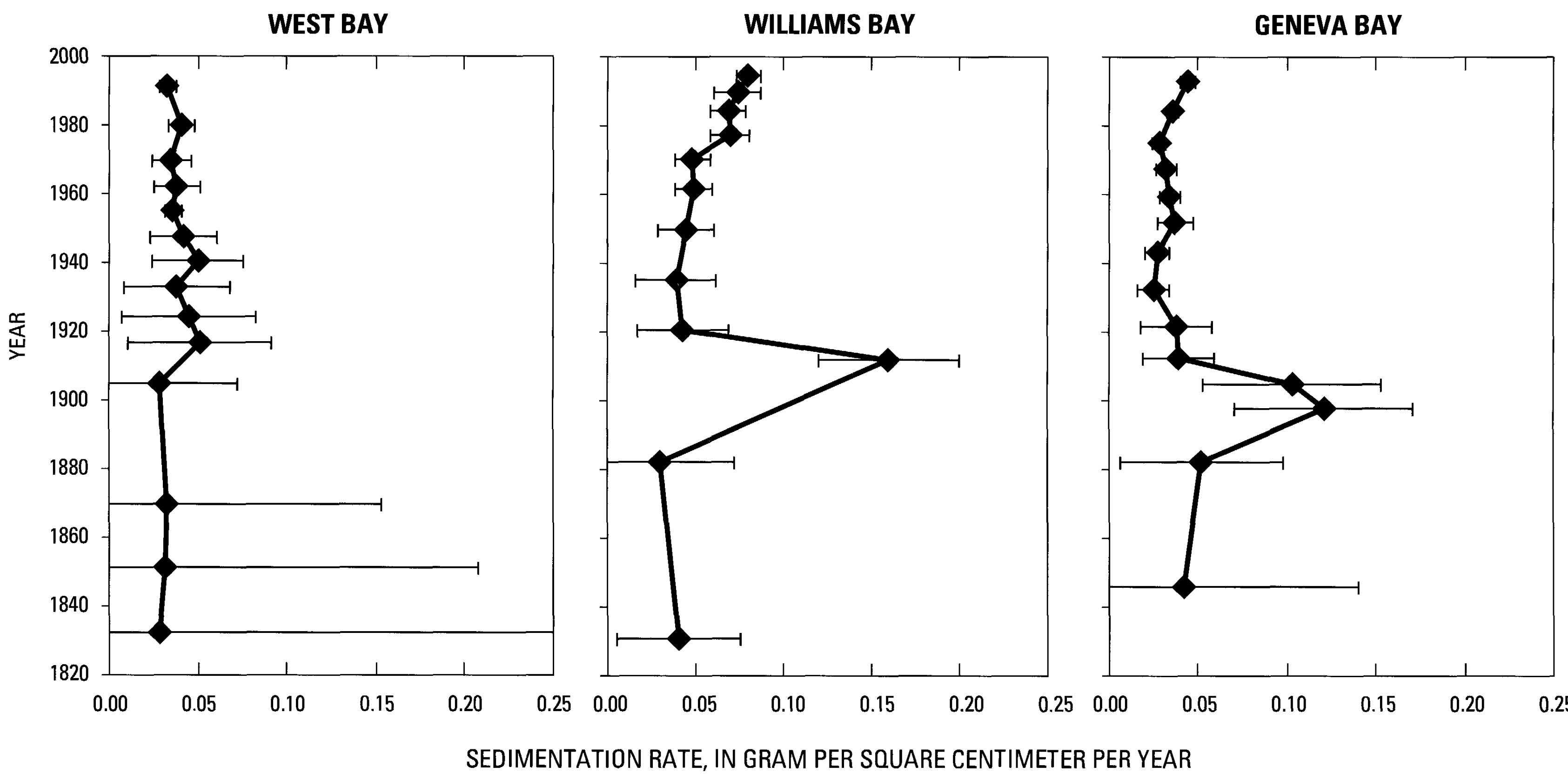


Figure 38. Sedimentation rates from three sediment cores from Geneva Lake, Wisconsin, 1830 to 1995. [Error bars are plus and minus 1 standard deviation (obtained from Garrison, 2000.)]

input of materials into the bays was from urban runoff, as demonstrated by increased zinc concentrations in the upper parts (more recently deposited material) of the sediment cores (fig. 39). Currently, the loading of zinc, a very common heavy metal and a surrogate for urban runoff (Bannerman and others, 1993; Good, 1993; Steuer and others, 1997), is more than 10 times the historical loading. Urban runoff into Geneva Bay increased earlier than other places in the lake because the area around this bay developed more quickly and more intensively than around other areas; also, storm sewers were installed earlier here than in most other areas around the lake, resulting in more direct delivery to the lake.

Specific aquatic organisms can be good indicators of the water chemistry in a lake because they are in direct contact with the water and are strongly affected by the chemical composition of their surroundings. Most groups of aquatic organisms used as water-quality indicators grow rapidly and are short lived, so the community composition responds rapidly to changing environmental conditions. Among the most useful organisms for paleolimnological analysis are diatoms. Diatoms are a type of algae that possess siliceous cell walls and usually are abundant, diverse, and well preserved in the sediments. Certain taxa of diatoms usually are found at specific nutrient concentrations and, therefore, can be used to infer historical changes in nutrient concentrations. Changes in the diatom communities in the Geneva Lake cores were used to infer historical P concentrations (described in detail by Garrison (2000). Because of the variability in diatom population estimations, the inferred P concentrations (fig. 40) should be used only as an indication of relative changes in historical P levels rather than as a description of actual historical concentrations.

Changes in the diatom community in the cores indicate that before the arrival of European settlers, P concentrations in the lake were very low, about from 5 to 12 μg/L, an indication that the lake was oligotrophic (fig. 40). With the arrival of European settlers, P concentrations began to rise, especially after 1900. Phosphorus concentrations increased throughout the lake, with highest P concentrations estimated during the 1930s through 1970s period. During most of the 1930s through 1970s, the lake would have been classified as eutrophic. Phosphorus concentrations since then appear to have decreased, but they remain above the concentrations in the lake prior to European settlement.

As mentioned earlier, long-term changes in a lake may be reflected by changes in dissolved oxygen concentrations; however, no long-term changes in dissolved oxygen concentrations were found by examining direct measurements (fig. 34). Changes in the manganese concentrations in sediment cores also can be used to describe historical changes in the extent of anoxia in the lake (Engstrom, and others, 1985). Manganese profiles from Geneva Lake demonstrated very little change over the 170-year record, also suggesting that dissolved oxygen concentrations in the lake have changed little over this time period.

The paleoecological study indicated that the water quality of Geneva Lake has deteriorated in the last 170 years, the principal cause being from urbanization. Changes in the watershed caused increases in both nutrient and sediment loading to the lake. Sedimentation rates were highest around 1900 to 1930, and P concentrations were highest between the 1930s and 1970s. Phosphorus concentrations have decreased since the 1970s; however, concentrations of chemical constituents associated with urban areas have increased to present time (mid 1990s), especially in Williams Bay and Geneva Bay.

Modeled Changes in Geneva Lake, 1975 to 1998

Measurements made in the lake and results of the sediment-core studies both indicated that P concentrations in the lake have decreased from about 20–25 μg/L in the mid-1970s to mid-1980s to about 6–15 μg/L in recent years. On the basis of two loading studies during this time period, total P loading to the lake has been reduced by approximately 2,000 kg, from about 5,200 kg in 1975 to about 3,200 kg in 1998 (fig. 31). To determine whether the measured decreases in P concentrations in the lake were what would be expected with this 38-percent reduction in loading, the hydrology and P loading of calendar year 1975 and WY1998 were input into the nine empirical models contained within WiLMS that were applicable to Geneva Lake (table 16).

To determine how an increase of about 2,000 kg of P supplied to the lake in 1975 should have affected near-surface P concentrations, the additional P was added to the base scenario from 1998 and near-surface P concentrations were simulated with the nine empirical models within WiLMS (table 16). Therefore, a total P loading of 4,350 kg (base scenario of 2,350 kg plus the addi-

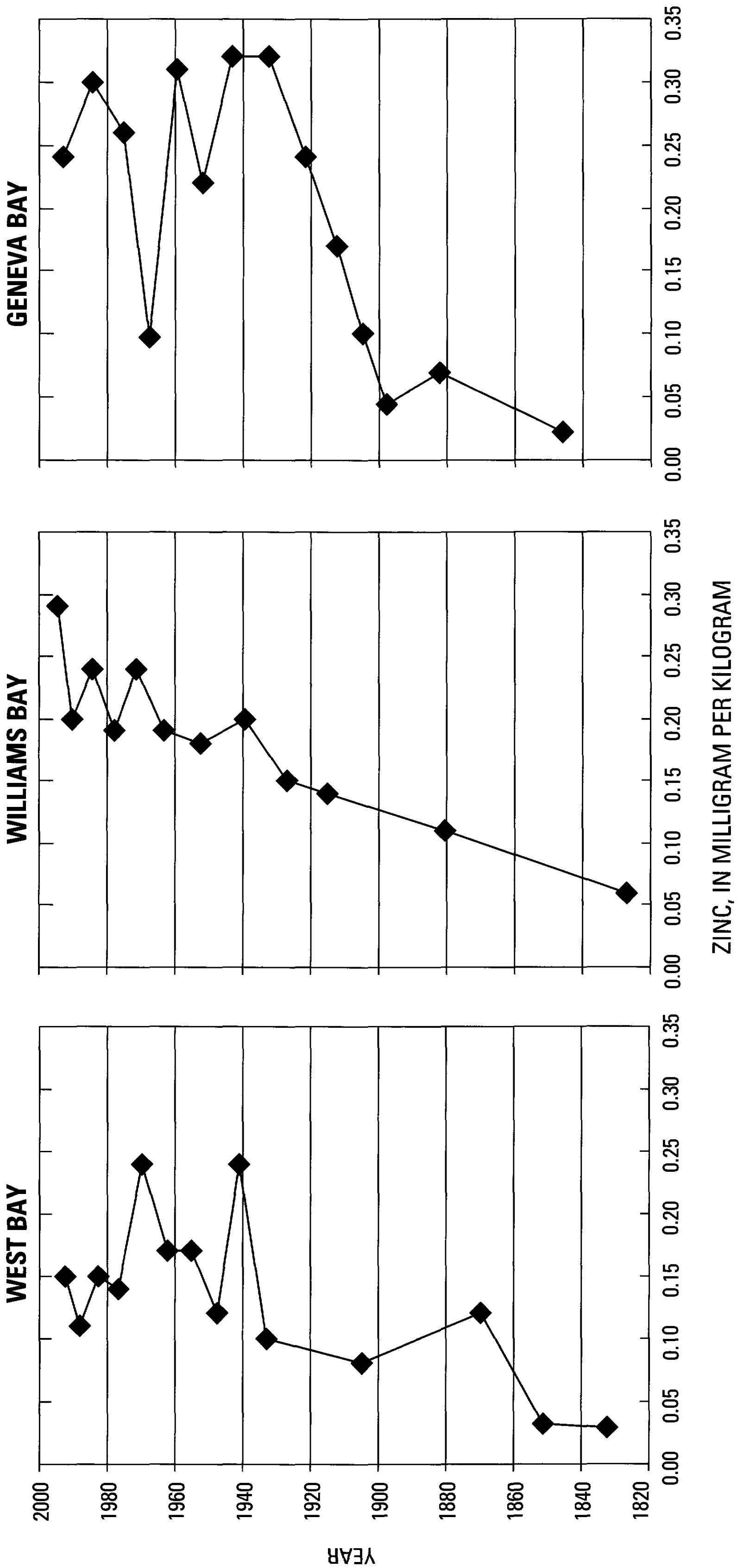


Figure 39. Profiles of zinc concentrations in three sediment cores from Geneva Lake, Wisconsin, 1830 to 1995 (obtained from Garrison, 2000).

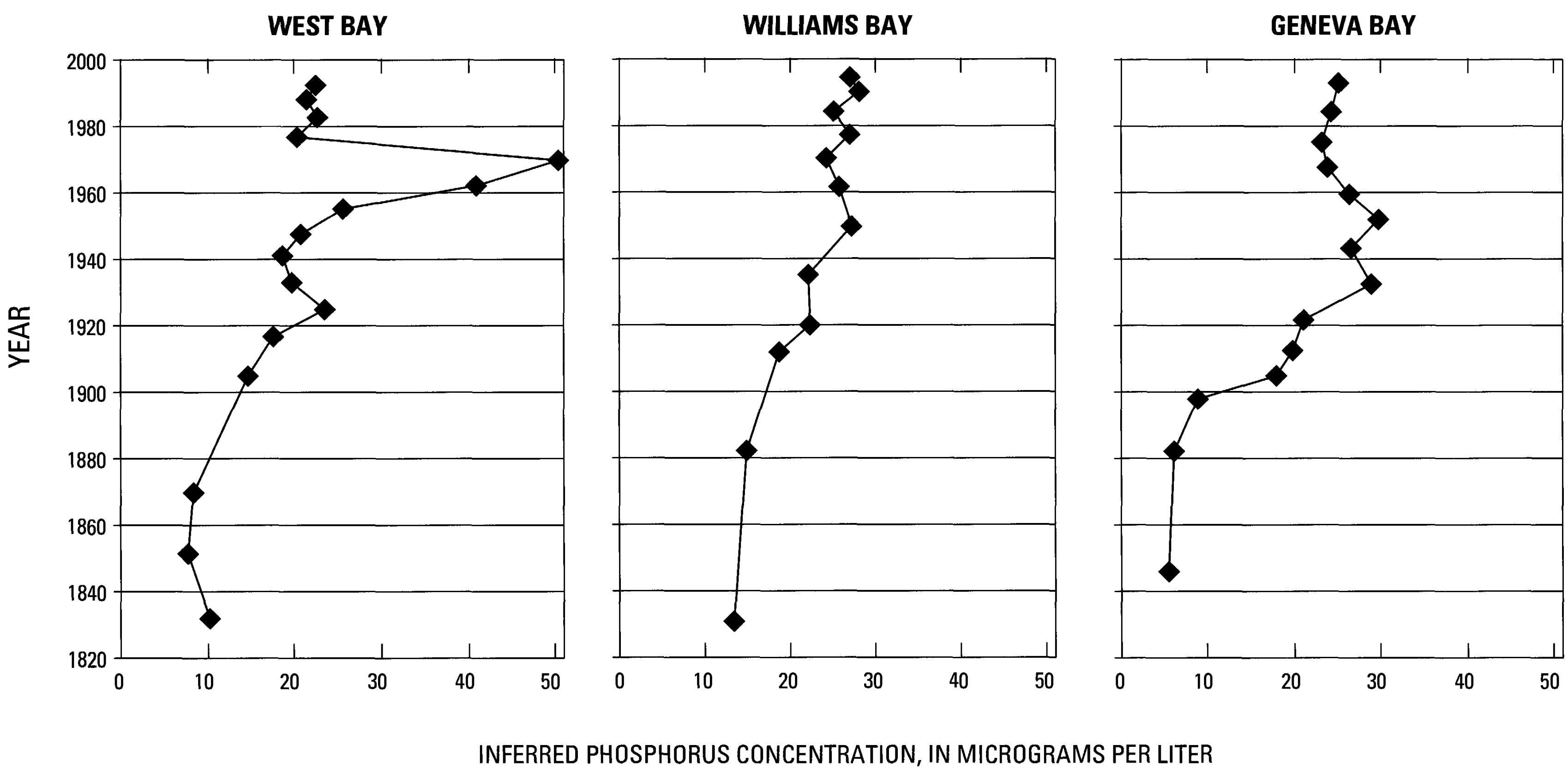


Figure 40. Inferred phosphorus concentrations for Geneva Lake, Wisconsin, 1830 to 1995, based on changes in diatom populations from three cores from the lake (obtained from Garrison, 2000).

tional 2,000 kg) was input into the nine empirical models. Average results from the nine models indicated that the increase in loading of 2,000 kg should have resulted in a near-surface P concentration of 37 µg/L or an increase of about 100 percent from the base scenario. If this percentage increase in P concentrations is assumed to be correct, then it is predicted that the average lake P concentration would increase from about 10 µg/L to about 20 µg/L, which is similar that measured in the lake in summer 1976 (18–22 µg/L; Southeastern Wisconsin Regional Planning Commission, 1985).

To determine how the increase in P loading of 2,000 kg and resulting higher P concentrations in the lake should have affected chlorophyll *a* concentrations and Secchi depths, an average near-surface P concentration of 20 µg/L was input into the other empirical relations contained within WiLMS. With an average summer P concentration of 20 µg/L, the models simulated an average chlorophyll *a* concentration of 7.4 µg/L (45 percent higher than the 5.1 µg/L estimated for a P concentration of 10 µg/L) and a Secchi depth of 2.2 m (27 percent less than the 3.0 m estimated for a P concentration of 10 µg/L). If it is again assumed that these simulations represent only the percentage change in chlorophyll *a* concentrations and Secchi depths, then it is predicted that the average chlorophyll *a* concentrations should have been 4.9 µg/L, approximately that measured in 1981, and that the average summer Secchi depth should have been about 2.9 m, approximately that measured between 1970 and 1981 (fig. 37). Therefore, it is felt that results from model can be used to provide reasonable predictions of how the lake should respond to future changes in P loading.

SUMMARY AND CONCLUSIONS

Geneva Lake, in Walworth County, has long been considered one of the most important natural resources in southeastern Wisconsin. As part of continuing efforts to improve the water quality of Geneva Lake, a collaborative effort between the USGS, WDNR, and GLEA was initiated in 1997 to document the present quality of the lake and its sediments, compute detailed hydrologic and P budgets for the lake, estimate how changes in P loading have affected water quality, and describe changes in the lake over the past 170 years by comparing water quality measured in this study with historical measurements and sediment-core information. This report summarized the results of this study.

A long-term sampling strategy for describing changes in the water quality in Geneva Lake was developed and consists of monthly sampling (with possible 2-week sampling during the breakdown in thermal stratification during October) in the center of the West and East Bays. Eliminating sampling in the East Bay also may be considered if there are financial constraints, because the only significant differences in water quality from the West Bay were found in N concentrations and dissolved oxygen concentrations below the thermocline.

Water-quality measurements collected during this study (1997–2000) indicate that the lake was between mesotrophic and oligotrophic trophic classes. During this study, the mean near-surface concentrations for total P was 9 µg/L, for total N was 550 µg/L, and for chlorophyll *a* concentration was 3 µg/L. The mean Secchi depth was 4.8 m. Near-surface N:P ratios were always greater than 28:1; therefore, if just N and P are considered, P always should be the limiting nutrient. Phytoplankton in the lake were dominated by diatoms except during some summer periods when cryptomonads were abundant. Zooplankton in the lake were dominated by small to medium-sized species, usually cyclopoid copepods; however, calanoid copepods and small bosminid types also were abundant in summer.

Analyses of the bottom surficial sediments of Geneva Lake indicated that municipal and agricultural drainage from the watershed has produced elevated (above natural levels), and potentially problematic, concentrations of some elements and compounds in the lake sediments. Because concentrations of some constituents were elevated indicates little about actual toxicological effects on any biological species, but it serves as a signal that future investigators of contaminants in the lake may want to focus on such details as bioavailability, cycling within the ecosystem, and toxicology of these compounds.

A hydrologic budget, constructed for WY1998 and WY1999, indicated that the major sources of water to Geneva Lake were precipitation (approximately 48 percent) and surface-water inflow (approximately 46 percent); surface water, in turn, was about equally divided between base flow and surface runoff during a normal-precipitation year (WY1998) but was dominated by surface runoff in a wet year (WY1999). Ground water contributed about 6 percent of the water supplied to the lake. Output of water from the lake was principally through its outlet (approximately 60 percent) and secondarily from evaporation (approximately 40 percent).

Phosphorus budgets also were constructed for WY1998 and WY1999. The total annual input of P was about 3,200 kg in WY1998 and about 8,500 kg in WY1999. The major source of P was from the tributaries to the lake, which contributed about 84 percent of total contributions. Most of this P load was delivered to the lake during surface-runoff events. Contributions of P from waterfowl and precipitation each accounted for about 7 percent of the total contributions (ranging from about 5 to 12 percent). Septic systems were estimated to contribute between about 2 and 8 percent of the total P load to the lake. The range in the estimated contributions from septic systems was because of uncertainty in how present systems were installed (for example proper soils), how well they are maintained and currently functioning, and how far they are from the lake. Only minor contributions of P came from ground water and the sediment of the lake. The total P load in WY1998 was about 2,000 kg less than that estimated in 1975. The major difference from P load estimates for 1975 was associated with the decrease in loading from the Fontana sewage-treatment plant.

Future decreases in P loading to the lake are likely to improve the water quality of the lake. Near-surface P concentrations should decrease by almost the same percentage as the decrease in P loading rates, whereas chlorophyll *a* concentrations should decrease and Secchi depths should increase, but by smaller percentages.

Direct water-quality measurements and water quality inferred from sediment-core analyses indicated that the water quality in Geneva Lake has changed over the past 170 years. Changes in the watershed caused increases in both nutrient and sediment loading to the lake. Sedimentation rates were highest around 1900 to 1930 and near-surface P concentrations were highest between the 1930s and early 1980s. As a result of reduced P loading to the lake since the mid-1980s, near-surface P concentrations decreased from 20–25 µg/L to about 10–15 µg/L, consistent with concentrations predicted through the use of empirical loading models. The reduced P concentrations in the lake may have increased water clarity; however, it is difficult to separate the effects of reduced P concentration from effects of increased zebra mussel populations in the lake. Although water clarity and nutrient and dissolved oxygen concentrations in the lake currently (1999–2000) are similar to those in the early 1900s, loading of other constituents associated with urban areas has continually increased resulting in high concentrations of these constituents in the bottom sediments, especially in Williams Bay and Geneva Bay.

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APPENDIX

Appendix. Freeze and breakup dates, and total days of ice cover for Geneva Lake, Wisconsin

[ND, no data; NF, lake did not freeze]

Winter	Freeze date		Breakup date		Duration of ice cover
	Month	Day	Month	Day	Days
1862–1863	1	7	4	1	84
1863–1864	12	18	4	8	112
1864–1865	12	8	4	2	115
1865–1866	12	16	4	17	122
1866–1867	12	27	4	18	112
1867–1868	12	31	3	25	85
1868–1869	12	11	3	16	95
1869–1870	12	21	4	12	112
1870–1871	12	24	3	20	86
1871–1872	12	9	4	15	128
1872–1873	12	4	4	8	125
1873–1874	12	21	4	12	112
1874–1875	12	20	4	15	116
1875–1876	12	27	4	10	105
1876–1877	12	6	4	13	128
1877–1878	1	7	2	21	45
1878–1879	12	23	4	13	111
1879–1880	12	18	3	3	76
1880–1881	11	23	5	4	162
1881–1882	1	2	3	1	58
1882–1883	12	16	4	13	118
1883–1884	12	19	4	13	116
1884–1885	12	31	4	20	110
1885–1886	12	7	1	3	27
1886–1887	12	19	4	2	104
1887–1888	12	28	4	12	106
1888–1889	1	11	3	31	79
1889–1890	1	17	4	9	82
1890–1891	1	3	4	13	100
1891–1892	1	3	4	2	90
1892–1893	12	20	4	5	106
1893–1894	12	7	3	11	94
1894–1895	ND	ND	4	6	ND
1895–1896	12	6	4	6	122
1896–1897	1	18	4	7	79
1897–1898	12	17	3	25	98
1898–1899	12	8	4	14	127

Appendix. Freeze and breakup dates, and total days of ice cover for Geneva Lake, Wisconsin—Continued

[ND, no data; NF, lake did not freeze]

Winter	Freeze date		Breakup date		Duration of ice cover
	Month	Day	Month	Day	Days
1899–1900	12	29	4	17	110
1900–1901	ND	ND	4	11	ND
1901–1902	1	19	3	15	55
1902–1903	1	18	3	18	59
1903–1904	12	16	4	15	121
1904–1905	1	5	3	28	82
1905–1906	1	23	4	8	75
1906–1907	1	23	3	24	60
1907–1908	1	28	3	26	58
1908–1909	1	7	4	5	88
1909–1910	12	28	3	24	86
1910–1911	1	3	3	25	81
1911–1912	1	2	4	13	102
1912–1913	1	8	3	29	80
1913–1914	2	5	4	1	55
1914–1915	12	24	4	8	105
1915–1916	1	13	4	2	80
1916–1917	12	26	3	31	95
1917–1918	12	15	3	30	105
1918–1919	2	10	3	12	30
1919–1920	12	16	3	26	101
1920–1921	1	12	2	15	34
1921–1922	1	22	3	25	62
1922–1923	1	23	4	15	82
1923–1924	1	14	4	9	86
1924–1925	12	28	3	26	88
1925–1926	12	29	4	20	112
1926–1927	12	28	3	14	76
1927–1928	1	27	3	24	57
1928–1929	1	15	3	27	71
1929–1930	12	23	3	17	84
1930–1931	1	15	3	26	70
1931–1932	3	10	3	30	21
1932–1933	2	8	4	2	53
1933–1934	2	2	4	3	60
1934–1935	1	18	3	21	62
1935–1936	12	27	3	26	90
1936–1937	1	10	4	13	93

Appendix. Freeze and breakup dates, and total days of ice cover for Geneva Lake, Wisconsin—Continued

[ND, no data; NF, lake did not freeze]

Winter	Freeze date		Breakup date		Duration of ice cover
	Month	Day	Month	Day	Days
1937–1938	1	9	3	12	62
1938–1939	2	4	3	27	51
1939–1940	1	7	4	15	99
1940–1941	1	21	4	10	79
1941–1942	1	10	4	4	84
1942–1943	1	19	3	31	71
1943–1944	1	7	4	13	97
1944–1945	1	10	3	16	65
1945–1946	12	24	3	17	83
1946–1947	12	30	4	11	102
1947–1948	12	20	3	27	98
1948–1949	1	18	3	30	71
1949–1950	1	17	4	11	84
1950–1951	12	12	4	10	119
1951–1952	12	17	4	4	109
1952–1953	12	29	3	22	83
1953–1954	1	11	3	20	68
1954–1955	1	11	4	2	81
1955–1956	12	12	4	4	114
1956–1957	1	2	3	24	81
1957–1958	1	2	4	5	93
1958–1959	12	11	4	8	118
1959–1960	1	16	4	13	88
1960–1961	12	22	3	27	95
1961–1962	12	28	4	11	104
1962–1963	12	24	4	4	101
1963–1964	12	17	4	8	113
1964–1965	12	18	4	15	118
1965–1966	1	10	3	18	67
1966–1967	12	26	4	1	96
1967–1968	12	26	3	27	92
1968–1969	12	25	4	9	105
1969–1970	12	24	4	9	106
1970–1971	12	28	4	11	104
1971–1972	1	5	4	18	104
1972–1973	12	7	3	11	94
1973–1974	12	31	3	19	78
1974–1975	1	18	4	19	91

Appendix. Freeze and breakup dates, and total days of ice cover for Geneva Lake, Wisconsin—Continued

[ND, no data; NF, lake did not freeze]

Winter	Freeze date		Breakup date		Duration of ice cover
	Month	Day	Month	Day	Days
1975–1976	1	4	3	20	76
1976–1977	12	7	3	30	113
1977–1978	12	26	4	9	104
1978–1979	12	25	4	15	111
1979–1980	1	12	4	7	86
1980–1981	12	20	3	21	91
1981–1982	12	28	4	13	106
1982–1983	1	16	3	7	50
1983–1984	12	18	4	10	114
1984–1985	1	2	3	31	88
1985–1986	12	17	3	30	103
1986–1987	1	19	3	9	49
1987–1988	1	4	3	29	85
1988–1989	12	28	4	5	98
1989–1990	12	13	3	17	94
1990–1991	12	26	3	23	87
1991–1992	1	16	3	9	53
1992–1993	12	26	4	11	106
1993–1994	12	26	4	3	98
1994–1995	1	5	3	20	74
1995–1996	12	11	4	2	113
1996–1997	1	8	3	27	78
1997–1998	NF	NF	NF	NF	0
1998–1999	1	7	2	5	29
1999–2000	1	20	2	28	39
2000–2001	12	27	4	5	99

Sources:

1863–1946: Ragotzkie (1960)—Except 1944—For which other sources (including ice data from Delavan Lake, Wis.) suggest the lake broke up in April.

Ragotzkie (1960) obtained data up to 1900 from McKinley Hodge, Field Secretary Geneva Lake Civic Association, Williams Bay, 1956.

Ragotzkie obtained data from 1900 to 1946 from Yerkes Observatory, Williams Bay.

1947–Present—Geneva Lake Environmental Agency (T. Peters, Geneva Lake Environmental Agency, written commun., 2001).

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WRIR 02–4039