Rhodhiss Lake, North Carolina: Analysis of Ambient Conditions and Simulation of Hydrodynamics, Constituent Transport, and Water-Quality Characteristics, 1993-94

By Mary J. Giorgino *and* Jerad D. Bales

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CONVERSION FACTORS, VERTICAL DATUM, AND DEFINITIONS

Multiply	***By***	***To obtain***
	Length	
centimeter (cm)	2.54	inch
meter (m)	3.281	foot
kilometer (km)	0.6214	mile
	Area	
square meter (m^2)	10.76	square foot
square kilometer (km^2)	0.3861	square mile
	Volume	
liter (L)	0.2642	gallon
cubic meter (m^3)	35.31	cubic foot
cubic meter (m^3)	264.2	gallon
	Flow rate	
cubic meter per second (m^3/s)	35.31	cubic foot per second
cubic meter per second per square kilometer [(m^3/s)/km^2]	91.49	cubic foot per second per square mile
cubic meter per second (m^3/s)	22.83	million gallons per day
	Mass	
gram (g)	0.03527	ounce, avoirdupois
kilogram (kg)	2.205	pound, avoirdupois
megagram or metric ton	1.102	ton, short

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

°F = 1.8 (°C + 32)

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

Specific conductance is given in microsiemens per centimeter at 25 degrees Celsius (µS/cm at 25 °C).

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

RHODHISS LAKE, NORTH CAROLINA: ANALYSIS OF AMBIENT CONDITIONS AND SIMULATION OF HYDRODYNAMICS, CONSTITUENT TRANSPORT, AND WATER-QUALITY CHARACTERISTICS, 1993-94

By Mary J. Giorgino *and* Jerad D. Bales

ABSTRACT

From January 1993 through March 1994, the U.S. Geological Survey conducted an investigation of Rhodhiss Lake in cooperation with the Western Piedmont Council of Governments. Objectives of the investigation were to describe ambient hydrologic and water-quality conditions, to estimate loadings of nutrients and suspended solids from selected tributaries and point sources, and to simulate hydraulic circulation and water-quality characteristics in Rhodhiss Lake using a hydrodynamic computer model.

The riverine headwaters of Rhodhiss Lake were unstratified, well oxygenated, and contained relatively high concentrations of suspended solids and nutrients throughout the study period. In general, concentrations of suspended solids, nitrate, orthophosphate, and total phosphorus decreased in a downstream direction from the headwaters to the Rhodhiss Dam. However, increases in specific conductance frequently were observed downstream from a wastewater discharge near mid-reservoir.

From mid-reservoir to the dam, Rhodhiss Lake thermally stratified during the summer of 1993. In this reach, dissolved oxygen was rapidly depleted from the bottom waters beginning in May 1993, and anoxic conditions persisted in the hypolimnion through the summer. During summer stratification, concentrations of nitrite plus nitrate, ammonia, and orthophosphate were low in the epilimnion, but concentrations of ammonia, orthophosphate, and total phosphorus increased in the hypolimnion. During fall and winter, Rhodhiss Lake was characterized by alternating periods of stratification and mixing.

A maximum chlorophyll *a* concentration of 52 micrograms per liter was observed at mid-reservoir on November 17, 1993, and was the only value that exceeded the North Carolina water-quality standard of 40 micrograms per liter. Concentrations of fecal coliform bacteria exceeded 200 colonies per 100 milliliters in the headwaters of Rhodhiss Lake 37 percent of the time, and at mid-reservoir and in the forebay 16 percent of the time.

In Lower Creek, a tributary to Rhodhiss Lake, concentrations of fecal coliform bacteria exceeded 200 colonies per 100 milliliters in 76 percent of the samples. This stream also contained elevated concentrations of nitrite plus nitrate, phosphorus, and specific conductance.

Loading estimates showed that almost all of the suspended solids and the majority of the nitrogen and phosphorus entering the headwaters of Rhodhiss Lake originated from nonpoint sources. During the investigation, point sources accounted for less than 1 percent of the suspended solids load to the reservoir headwaters, but point sources accounted for up to 27 and 22 percent of the total nitrogen and total phosphorus loads, respectively. Additional loadings of nitrogen and phosphorus entered Rhodhiss Lake by municipal wastewater discharge near mid-reservoir.

The U.S. Army Corps of Engineers CE-QUAL-W2 model is a two-dimensional, laterally averaged model that simulates hydrodynamics and water quality. The model was applied to Rhodhiss Lake from Huffman Bridge to Rhodhiss Dam—a distance of 18.5 kilometers—and was calibrated using data collected from April 1993 through March 1994.

During the simulation period, measured water levels varied a total of 1.32 meters, and water temperatures ranged from 4 to 30 degrees Celsius. The calibrated model provided good agreement between measured and simulated water levels at Rhodhiss Dam. Likewise, simulated water temperatures were generally within 2 degrees Celsius of measured values; however, the model tended to overpredict temperatures near the bottom of the reservoir by 1 to 3 degrees Celsius during warm months. This suggests that the model, as calibrated, overpredicts vertical mixing. Simulated dissolved oxygen concentrations followed the same general patterns and magnitudes as measured values, and there was good agreement between simulated and measured frequency of occurrence of dissolved oxygen concentrations less than 5 milligrams per liter.

Tracer-release simulations for stratified and unstratified conditions demonstrated the effects of stratification on dilution and rate of transport.

Simulation of increased ambient air temperatures resulted in a slightly greater amount of time that dissolved oxygen concentrations in the reservoir were less than 4 milligrams per liter, but had little effect on algal biomass. A simulated 30-percent reduction in inflow phosphate concentrations resulted in a 20-percent decrease in the maximum algal concentration, with the greatest effects in late summer and early fall. A 50-percent reduction in phosphate releases from bottom sediments resulted in only a small reduction in algal concentration, primarily in the spring. Changes in phosphate load from a municipal wastewater discharge near mid-reservoir were simulated. Algal concentrations were fairly insensitive to increases in loading, but dissolved oxygen concentrations were reduced as the loading increased. Simulated algal concentrations increased 2 to 3 times when the discharge was moved from the reservoir bottom to near the surface.

INTRODUCTION

Rhodhiss Lake is an impoundment of the Catawba River (fig. 1) and was constructed in 1925 to supply hydroelectric power for a growing population and textile industry. Today, in addition to power generation, the reservoir is used for drinking- and industrial-water supply, recreation, waste assimilation, and habitat for fish and wildlife.

Historical information regarding hydraulic circulation and constituent transport in the reservoir is lacking. This information is needed to more effectively manage the reservoir and its watershed and also to predict water-quality responses to changes in constituent loading or hydrologic regime. Such changes could result from recently adopted State watershed-protection regulations, proposed wastewater-treatment plant expansions in the Rhodhiss Lake watershed, and a new basinwide approach used by State regulators to permit point-source discharges. Loadings of nonpoint-source pollutants could change as a result of the adoption of forestry best-management practices (BMP's), the continued administration of a statewide sedimentation control program, and the availability of cost-share funding for installing agricultural BMP's.

In January 1993, an investigation was undertaken to characterize hydrologic and water-quality conditions in Rhodhiss Lake and to estimate the effects of changes in external conditions on reservoir water quality. This investigation was part of a larger study conducted in the upper Catawba River Basin by the U.S. Geological Survey (USGS) in cooperation with the Western Piedmont Council of Governments, Hickory, North Carolina, to monitor water levels, streamflow, and water quality (Jaynes. 1994) during a 15-month period, and to simulate circulation and water-quality processes in Rhodhiss Lake and Lake Hickory using the U.S. Army Corps of Engineers CE-QUAL-W2 model. The Western Piedmont Council

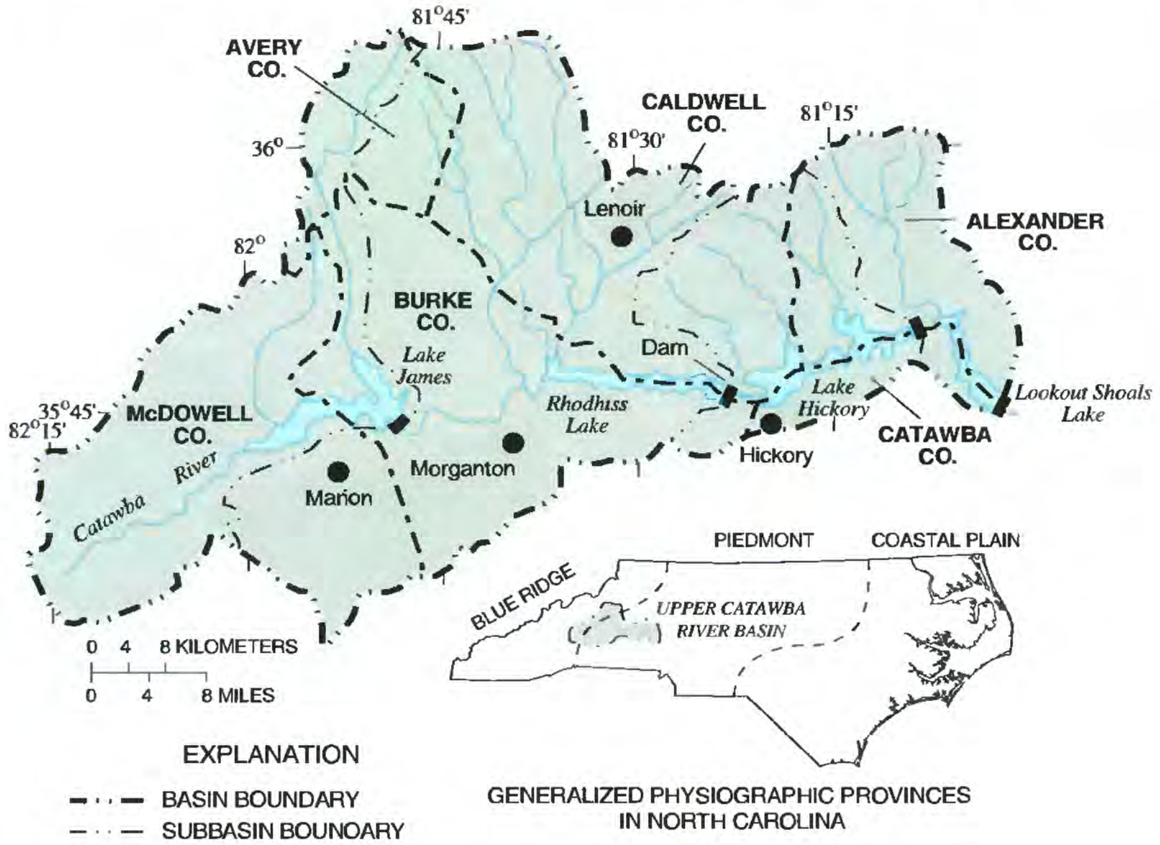

Figure 1. Location of Rhodhiss Lake in the upper Catawba River Basin of North Carolina.

of Governments plans to use the model as a tool to guide land and water-resource management efforts in the basin, especially to evaluate effects of changing nutrient loads on reservoir water quality.

Description of Study Area

The upper Catawba River Basin extends from the lower Blue Ridge Mountains into the Piedmont Physiographic Province of North Carolina (fig. 1). Four mainstem reservoirs are located within about a 150-kilometer (km) segment of the Catawba River as it flows through McDowell County and continues eastward through Burke, Caldwell, Catawba, and Alexander Counties. In downstream order, these reservoirs are Lake James, Rhodhiss Lake, Lake Hickory, and Lookout Shoals Lake.

The study area has a warm, temperate climate (Lee, 1955), with mean annual precipitation of approximately 120 centimeters (cm) (Eder and others, 1983). Geologic formations are mostly Precambrian gneisses and schists, and soils are predominately erodible red clay loams (Lee, 1955). The Rhodhiss Lake watershed is characterized by foothills and rounded uplands. Land uses in the watershed include silviculture, agriculture, and urban and industrial areas around the cities of Lenoir and Morganton (fig. 1). The basin contains numerous textile mills, textile machinery and dye plants, and furniture-manufacturing plants.

Rhodhiss Lake is the drinking-water source for the city of Lenoir and several smaller towns and communities in Burke and Caldwell Counties, and is used by Duke Power Company to generate hydroelectric power. The reservoir receives treated effluent from several domestic and industrial facilities, along with runoff from urban, agricultural, and silvicultural areas. Water-quality problems, such as algal blooms and elevated nutrient concentrations, have been observed in Rhodhiss Lake by the North Carolina Department of Environment, Health, and Natural Resources (1992, 1994b).

Rhodhiss Lake is approximately 22 km long and is narrow throughout its length (fig. 2). The surface area of the lake is 10.4 square kilometers (km^2); the mean depth is 8.0 meters (m); and the maximum depth is 16 m. The predominant inflow to Rhodhiss Lake is the Catawba River; other major tributaries include the Johns River and Lower Creek, both of which enter at the upper end of the reservoir (fig. 2). Additional smaller streams flow into the reservoir along its length. The water release at Rhodhiss Dam is regulated by Duke Power Company.

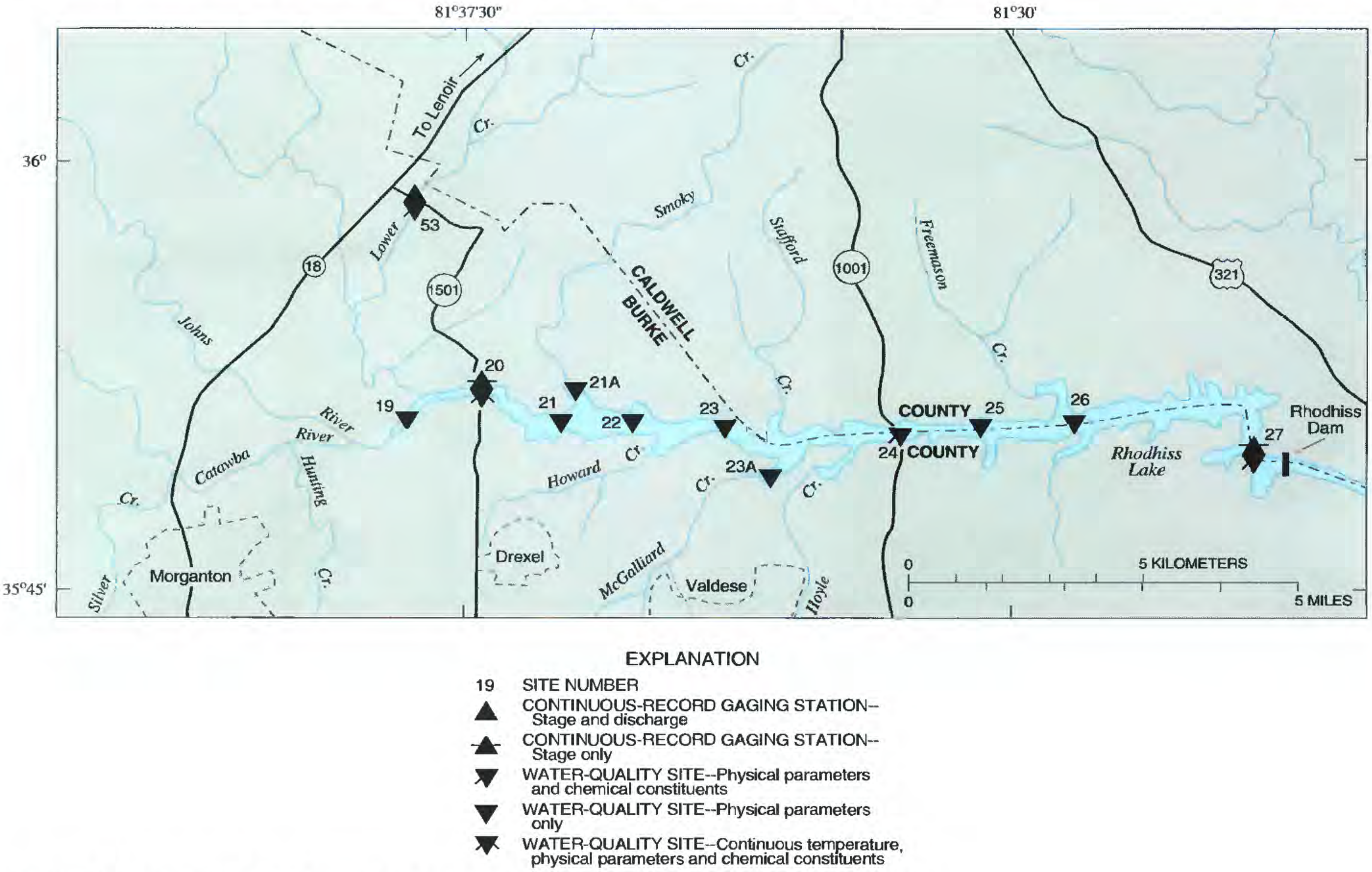

Figure 2. Locations of data-collection sites in the Rhodhiss Lake study area, 1993-94.

Purpose and Scope

The purpose of this report is (1) to characterize ambient hydrologic and water-quality conditions in Rhodhiss Lake and Lower Creek, a tributary to Rhodhiss Lake, (2) to document calibration of the CE-QUAL-W2 model for Rhodhiss Lake, and (3) to present results of model application for hypothetical scenarios identified by the Western Piedmont Council of Governments.

Characterization of Rhodhiss Lake water-quality conditions was based on data collected from January 1993 through March 1994. Hydrologic data included water level and streamflow measured in the reservoir headwaters and in Lower Creek (fig. 2), water levels in the reservoir forebay, and dam-release records. Water-quality characteristics included water temperature, dissolved oxygen (DO) concentration, pH, and specific conductance measured in the field at 11 sites in Rhodhiss Lake and at one site in Lower Creek, and a suite of chemical constituents determined from samples collected from three sites in Rhodhiss Lake and one site in Lower Creek. These data, as well as meteorologic data, were used to calibrate and apply the water-quality model.

Results are summarized for physical water-quality characteristics and selected chemical and biological constituents, including total suspended solids, nitrogen and phosphorus fractions, chlorophyll *a*, and fecal coliform bacteria. Temporal and spatial patterns of water quality in Rhodhiss Lake are described and when applicable, conditions are compared to prevailing water-quality standards. Loadings of total suspended solids, nitrogen, and phosphorus to Lower Creek and to the headwaters of Rhodhiss Lake are presented, as are estimated loadings from major point-source discharges in the Rhodhiss Lake watershed.

Hydraulic circulation and water-quality characteristics in Rhodhiss Lake were simulated using the CE-QUAL-W2 model. This laterally averaged, two-dimensional model was calibrated to simulate physical and water-quality conditions along the length of the reservoir and with depth. The calibrated model was used to simulate the effects of changes in external conditions including wind, air temperature, nonpoint-source loads, and point-source loads on reservoir water quality. The simulations were made to (1) demonstrate model utility, (2) provide insight into reservoir mixing and transport processes, (3) show the effects of small changes in external conditions on reservoir processes, and (4) simulate the effects of possible changes in nutrient inputs.

Previous Studies

Although water-quality conditions in Rhodhiss Lake and its tributary streams have been investigated by various Federal, State, and university researchers, hydraulic circulation and its effects on water quality in the lake have not been investigated previously. In the late 1950's, the North Carolina Department of Water Resources conducted the first comprehensive survey of pollution in the State's surface waters. During this survey, numerous problems were observed in streams draining urban areas of the Rhodhiss Lake watershed, including Lower, Silver, Hunting, Howard, and McGalliard Creeks (fig. 2; North Carolina Department of Water Resources, 1961). Domestic sewage from municipal, industrial, and private outfalls depleted DO concentrations in the streams and raised concentrations of fecal coliform bacteria to millions of colonies per 100 milliliters (ml) of water. Industrial effluents contributed additional oxygen-demanding wastes, bacteria, oil and grease, dyes, and toxic constituents such as copper, lead, cyanide, phenols, and formaldehyde to streams near Marion, Morganton, Drexel, Valdese, and Lenoir (figs. 1 and 2; North Carolina Department of Water Resources, 1961). Similar problems were observed in other urban-industrial areas of the Piedmont during the late 1950's, before many technological advances in wastewater-treatment and water-quality protection measures were implemented.

Water-quality conditions continued to be monitored in Lower Creek during subsequent decades. Based on data collected in 1967, the USGS reported that concentrations of nitrate, phosphate, and chloride in Lower Creek were elevated compared to levels in undisturbed streams (Wilder and Slack, 1971a, 1971b). Results of another USGS investigation conducted during the 1970's indicated that concentrations and yields of suspended sediment in Lower Creek were comparable to those in other streams affected by urban construction in the Catawba River Basin, and were higher than those streams in forested or agricultural watersheds nearby (Simmons, 1993). A recent evaluation by the North Carolina Department of Environment, Health, and Natural Resources (1994b) indicated that Lower Creek only partially supported its designated uses because high concentrations of sediment and fecal coliform bacteria impaired water quality. Water-quality problems in Lower Creek were attributed to both point and nonpoint sources (North Carolina Department of Environment, Health, and Natural Resources, 1994b, 1994c).

Investigations describing the water quality of unpolluted streams near Rhodhiss Lake are valuable for comparative purposes. From 1973 through 1978, the USGS, in cooperation with the North Carolina Department of Natural Resources and Community Development, investigated 39 streams with 90- to 100-percent forested basins. Results were used to characterize water-quality conditions in relatively undisturbed watersheds and to delineate five geochemical zones within North Carolina (Simmons and Heath, 1982). Although none of the sites were in the Rhodhiss Lake drainage area, 16 sites were located in the same geochemical zone (zone I). As might be expected, the pristine streams contained low concentrations of major inorganic constituents, nitrogen and phosphorus fractions, and trace metals (Simmons and Heath, 1982). A supplemental investigation at nine of the original sites, including three sites in zone I, was conducted from 1985 through 1988 (Caldwell, 1992).

Previous investigations of Rhodhiss Lake primarily focused on classifying the trophic status of the reservoir. Since the early 1970's, investigators have described Rhodhiss Lake as eutrophic (U.S. Environmental Protection Agency, 1975; North Carolina Department of Natural Resources and Community Development, 1982, 1984; North Carolina Department of Environment, Health, and Natural Resources, 1994b) or mesotrophic (Weiss and Kuenzler, 1976; North Carolina Department of Environment, Health, and Natural Resources, 1992), depending on the trophic classification system used and on temporal variations in water-quality components. Moderately elevated concentrations of phosphorus are consistently found in Rhodhiss Lake and should support abundant phytoplankton biomass. However, previous investigators noted that short retention time and high levels of inorganic turbidity in the reservoir may have depressed phytoplankton production, thus accounting for the relatively low concentrations of chlorophyll *a* typically observed in Rhodhiss Lake.

Methods of Data Collection

Data collection began in January 1993 and continued through March 1994. Hydrologic measurements were made at three locations—the reservoir headwaters, reservoir forebay, and Lower Creek (table 1; fig. 2). Physical water-quality conditions were measured at 11 sites in Rhodhiss Lake and at one site in Lower Creek. Water samples were collected for chemical characterization at four sites—three in the reservoir and one in Lower Creek. Field measurements were made and water samples were collected monthly from January through March 1993, twice monthly from April through September 1993, and monthly from October 1993 through March 1994. Detailed sampling and analytical methodologies were described by Jaynes (1994).

Continuous-stage gages recorded water level every 15 minutes at Rhodhiss Lake site 20 and at Lower Creek site 53 (fig. 2). At site 53, streamflow (or "discharge") was computed using a stage-discharge relation developed during the investigation. Streamflow at site 20 was estimated by summing flows at two gages upstream from Rhodhiss Lake (USGS stations 0213903612, Catawba River at Calvin; and 02140991, Johns River at Arneys Store), plus estimates of flow from the ungaged portion of the watershed. Data from these two gages were published previously (Barker and others, 1994; Ragland and others, 1995). In addition to hydrologic measurements made by the USGS, Duke Power Company recorded water levels in the forebay of Rhodhiss Lake (site 27, fig. 2) and estimates of outflow from Rhodhiss Dam at hourly intervals.

Temperature, DO concentration, specific conductance, and pH were measured in the field with a multiparameter water-quality meter. At Lower Creek site 53, measurements were made 0.1 m beneath the water surface. At the 11 reservoir sites, measurements were made at 1- or 2-m vertical intervals throughout the water column to characterize patterns of stratification. Water transparency was measured at reservoir sites with a Secchi disk. In addition, automated monitors recorded water temperature every 15 minutes at headwater site 20 at three depths—near the surface, at mid-depth, and near the bottom.

To obtain composite samples at Lower Creek site 53, water was collected from several intervals across the stream using a depth-integrating sampler (Edwards and Glysson, 1988). Composite samples were analyzed for suspended sediment, total and volatile suspended solids, nutrients (total organic nitrogen plus ammonia, dissolved ammonia, dissolved nitrite plus nitrate, total phosphorus, and dissolved orthophosphate), biochemical oxygen demand (BOD), hardness, alkalinity, iron, calcium, and magnesium. Grab samples for fecal coliform bacteria and total organic carbon (TOC) were collected at midchannel by hand dipping or by using a weighted bottle holder.

Table 1. Description of data-collection network in the Rhodhiss Lake study area

[Physical water-quality characteristics and chemical constituents are listed in text. --, data not collected; x, data collected; SR, secondary road]

USGS identification number	Site location and number (fig. 2)	Type of hydrologic measurement		Type of water-quality data	
		Stage	Discharge	Physical	Chemical
0214101475	Rhodhiss Lake, site 19	--	--	x	--
0214126765	Rhodhiss Lake, Huffman Bridge, SR1501, site 20	x	x[a]	x	x
0214126800	Rhodhiss Lake, site 21	--	--	x	--
0214127125	Smoky Creek at Amherst, site 21A	--	--	x	--
0214127175	Rhodhiss Lake, site 22	--	--	x	--
0214134900	Rhodhiss Lake, site 23	--	--	x	--
0214142595	McGalliard Creek above mouth, site 23A	--	--	x	--
0214146100	Rhodhiss Lake at Castle Bridge, SR1001, site 24	--	--	x	x
0214146295	Rhodhiss Lake, site 25	--	--	x	--
0214147230	Rhodhiss Lake, site 26	--	--	x	--
0214148975	Rhodhiss Lake at Rhodhiss Dam, site 27	x[b]	x[b]	x	x
02141245	Lower Creek near Morganton, SR 1501, site 53	x	x	x	x

[a] Computed with data from upstream gages; see text for complete explanation.

[b] Measured by Duke Power Company.

The three water-chemistry sites in Rhodhiss Lake represented headwater, mid-reservoir, and forebay areas of the reservoir (sites 20, 24, and 27, respectively, fig. 2). At each site, separate samples were collected with an Alpha-type water sampler from three depths to characterize conditions near the surface (0.5 m below surface), near the bottom (1.0 m above bottom), and in the metalimnion. These samples were analyzed for BOD, hardness, alkalinity, total and volatile suspended solids, nutrients, total iron, dissolved calcium, and dissolved magnesium. Water samples for TOC analysis were collected at the same depths using a Teflon sampler. If no thermocline was apparent at the time of sampling, only surface and bottom samples were collected. A thermocline was never observed at site 20, and thermocline samples were collected only eight times at site 24 and seven times at site 27. Thermocline data were reported by Jaynes (1994) but are not discussed further in this report.

At each reservoir water-chemistry site, a depth-integrated sample of the euphotic zone was collected for chlorophyll *a* analysis. The depth of the euphotic zone was estimated by doubling the Secchi disk depth, consistent with sampling procedures used by the North Carolina Division of Water Quality (North Carolina Department of Environment, Health, and Natural Resources, 1992). Chlorophyll *a* samples were filtered immediately in the field, and filters were sealed, chilled, and protected from light until submitted for analysis. Samples for fecal coliform analysis were collected directly into sterile bottles from just below the surface by hand-dipping or using a weighted bottle holder.

Water samples were analyzed by several laboratories. Samples for fecal coliform bacteria were analyzed by the City of Hickory Water Filtration Plant using a membrane-filtration technique (American Public Health Association and others, 1992) within 6 hours of collection. Samples for BOD were analyzed by the City of Hickory Wastewater-Treatment Plant. Concentrations of suspended sediment, total suspended solids, and volatile suspended solids were determined by the USGS District sediment laboratory in Raleigh, North Carolina, using gravimetric procedures documented by Guy (1969) and Fishman and Friedman (1989).

All other constituents were analyzed at the USGS National Water-Quality Laboratory in Arvada, Colorado. Low-level analyses for nutrients resulted in laboratory reporting levels of 0.002 milligram per liter (mg/L) for dissolved ammonia, 0.005 mg/L for dissolved nitrite plus nitrate, and 0.001 mg/L for total phosphorus and dissolved orthophosphate (Fishman, 1993). Total organic nitrogen plus ammonia was reported to the nearest 0.20 mg/L (Fishman and Friedman, 1989). Chlorophyll *a* was separated by high-pressure liquid chromatography (HPLC) and measured fluorometrically (Britton and Greeson, 1987). Unlike other fluorometric or spectrophotometric methods, this procedure does not require a correction for pheophytin (American Public Health Association and others. 1992).

Acknowledgments

Sandra Gillaspie and Diane Wilburn of the North Carolina Division of Water Quality supplied compliance-monitoring data used to estimate loadings of nutrients and solids from point-source dischargers. Harold Quidley, Larry Ausley, and Ed Williams, also of the North Carolina Division of Water Quality, performed *in situ* measurements of sediment-oxygen demand at two locations in Rhodhiss Lake which aided model calibration. Tom Cole, an author of the U.S. Army Corps of Engineers CE-QUAL-W2 model, provided valuable guidance on model application.

AMBIENT CONDITIONS

Hydrologic and water-quality conditions are summarized in this section. Hydrologic data include precipitation records and measurements of water level and streamflow from Lower Creek and the Catawba River upstream from Rhodhiss Lake. Records of outflow from the reservoir were provided by Duke Power Company and were used to estimate hydraulic retention times for Rhodhiss Lake during the investigation.

Summarized water-quality conditions include measurements of temperature, DO, pH, and specific conductance, and concentrations of selected chemical and biological constituents. Spatial and temporal variations in water quality are described, and results are compared to State water-quality standards and to other published studies. Constituent concentrations and records of streamflow were used to compute loadings of suspended solids and nutrient fractions in Lower Creek and in the headwaters of Rhodhiss Lake. In addition, inputs of suspended solids and nutrients from selected point-source discharges in the Rhodhiss Lake watershed were estimated.

Hydrologic Conditions

Long-term average precipitation at the Hickory Airport ranges from 8.76 to 12.14 cm per month (National Oceanic and Atmospheric Administration, 1992). During the investigation, monthly precipitation ranged from a scant 1.42 cm during June 1993 to 21.39 cm during March 1993 (fig. 3). Overall, rainfall was below average during the summer of 1993, and above average during the winters of 1993 and 1994 (fig. 3). Monthly precipitation values shown in figure 3 represent the averages of precipitation recorded at Rhodhiss Dam and at Oxford Dam on Lake Hickory.

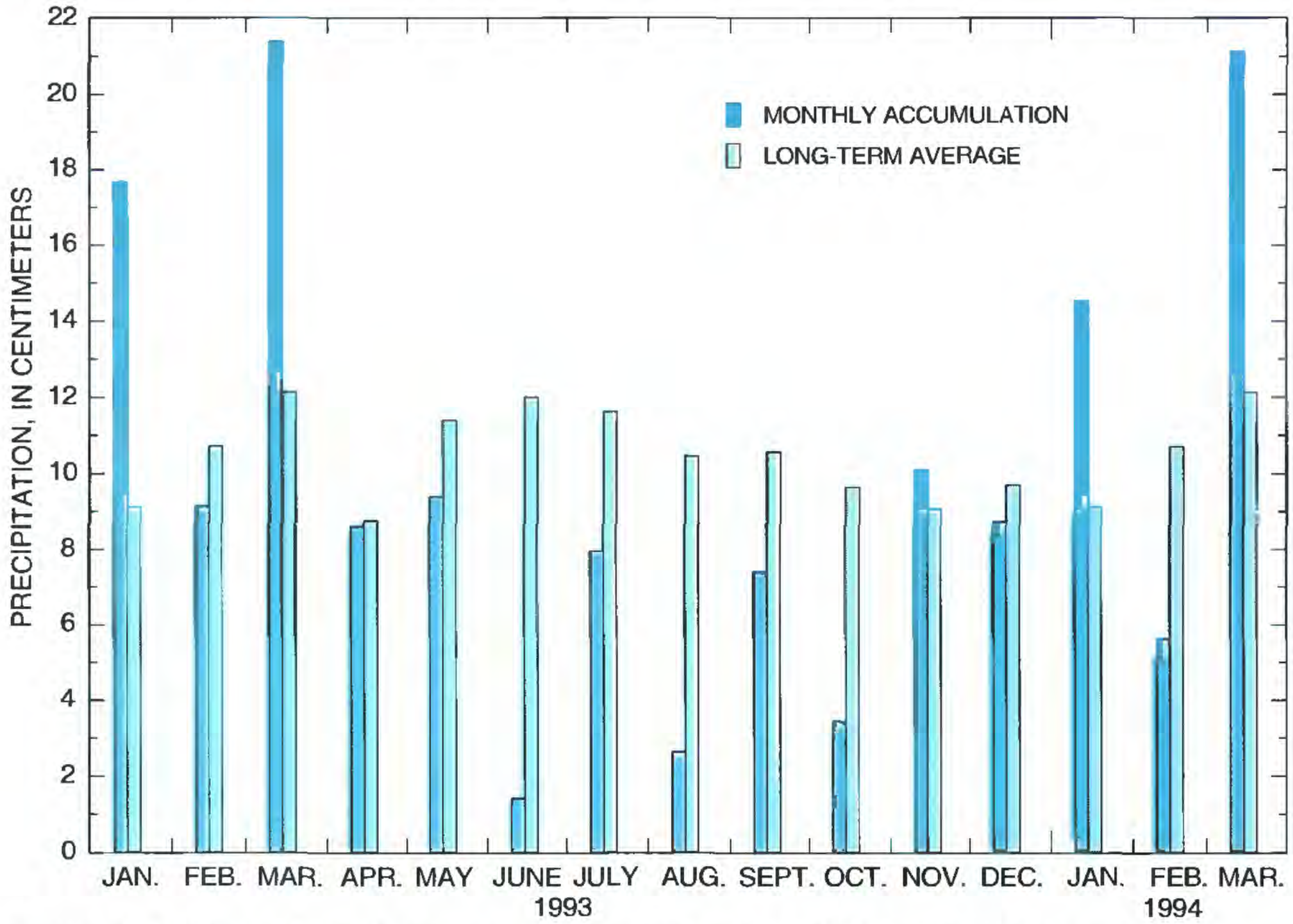

Figure 3. Relation of monthly precipitation from January 1993 through March 1994, with *long-term average precipitation near Hickory, North Carolina.*

Streamflow in Lower Creek (site 53, fig. 2) and in the upstream reach of Rhodhiss Lake (site 20, fig. 2) mirrored precipitation patterns, with peaks following heavy rainfall in March 1993 and March 1994, and low flows in the summer and fall of 1993 (fig. 4). During the period of January 1, 1993, through March 31, 1994, daily mean discharge ranged from 1.30 to 29.4 cubic meters per second (m^3/s) at Lower Creek site 53, and from 8.72 to 660 m^3/s at Rhodhiss Lake site 20.

Hydrologic data indicate that Rhodhiss Lake generally is characterized by short hydraulic retention times. Bathymetric measurements made by the USGS in June 1994 show that the current full-pond volume of Rhodhiss Lake is 59.2 million cubic meters (m^3), far less than the 89.6 million m^3 previously reported by the U.S. Environmental Protection Agency (1975). Theoretical hydraulic retention times for Rhodhiss Lake were estimated by dividing reservoir volume by outflow. From January 1993 through March 1994, retention times varied from 3 to 38 days with an average of 11 days.

Physical Water-Quality Characteristics

Temperature, DO concentration, pH, and specific conductance in Rhodhiss Lake varied seasonally and spatially and were influenced by interacting external and internal factors such as weather, inflow and outflow conditions, point-source discharges, and in-lake biochemical processes. Physical water-quality characteristics were measured at 12 sites in the study area, including nine along the reservoir mainstem, two in reservoir coves, and one in Lower Creek. The following discussion focuses on characteristics at the nine mainstem sites and, to a lesser extent, at Lower Creek site 53 (fig. 2).

Data from the nine mainstem sites are presented graphically as depth-distance diagrams (figs. 5-8). Each diagram shows conditions for one date, and represents a cross-section of the reservoir along the thalweg from the headwaters to the dam. Contours, also referred to as isopleths, delineate regions of similar value. Results for four sampling dates illustrate seasonal patterns of temperature, DO concentration, pH, and specific conductance.

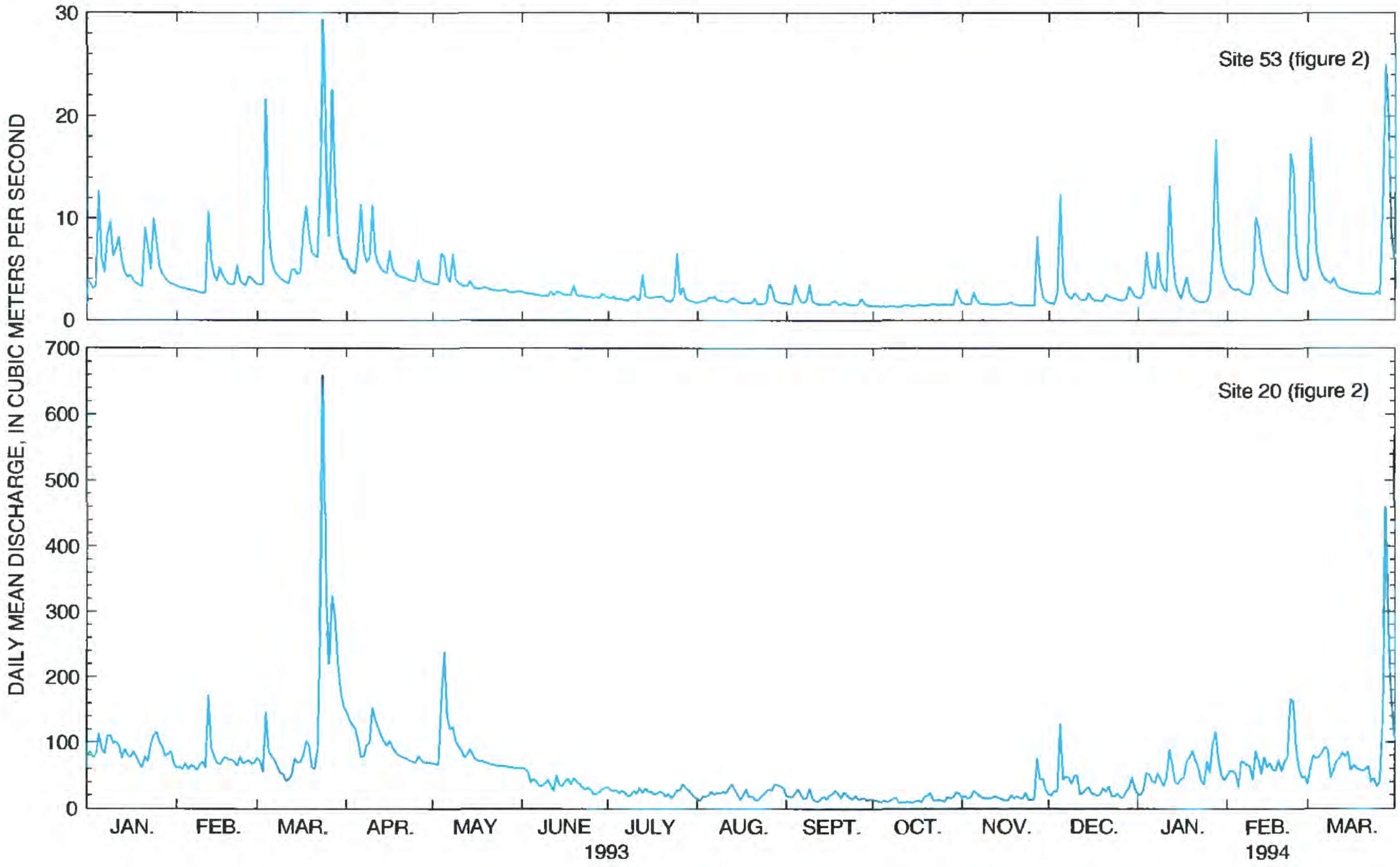

Figure 4. Daily mean discharge at Lower Creek site 53 and Rhodhiss Lake site 20, January 1993 through March 1994.

Physical water-quality characteristics indicate that longitudinal gradients exist in Rhodhiss Lake. Such gradients are common in reservoirs (Kennedy and others, 1985), and often are used to define riverine, transitional, and lacustrine zones—zones with boundaries that are dynamic (Thornton and others, 1981). For example, during extremely high-flow conditions in March 1993 (not shown), physical water-quality characteristics were uniformly distributed throughout the reservoir, indicating that Rhodhiss Lake was entirely riverine. During the summer and fall of 1993, temperature and DO distributions showed that the riverine zone extended from the headwaters through site 21; sites 22 and 23 were transitional; and sites 24 through 27 were lacustrine (figs. 5 and 6). The presence of these zones had important implications for the distribution of chemical and biological constituents in Rhodhiss Lake.

Temperature

Temperature distributions varied both seasonally and spatially, and indicated complex patterns of water circulation in Rhodhiss Lake. Riverine inflows from the Catawba River and the Johns River constitute the shallow headwater region of the reservoir. This riverine zone was consistently well-mixed (fig. 5). Continuous recorders positioned at site 20 (fig. 2) confirmed that temperature differences between the surface and the bottom never exceeded 2 degrees Celsius (°C) (Jaynes, 1994).

From mid-reservoir to the dam, temperature distributions were somewhat more lake-like. This part of Rhodhiss Lake began to stratify in May 1993, and remained stratified through the summer (fig. 5). Stratification refers to the establishment of horizontal layers of water with different temperatures and is indicated in fig. 5 by horizontal isopleths. Stratification occurs naturally as reservoir surface waters respond to warming air temperatures. The resulting vertical temperature differences establish density differences between adjacent layers of water which in turn, limit vertical mixing.

During the summer, inflows entering the upstream end of the reservoir tended to be cooler than surface waters downstream. Temperature distributions suggest that as the inflow reached the transitional zone, it sank beneath the warmer epilimnion, then continued toward the dam as an interflow (fig. 5). Effects of the interflow are most clearly seen in figure 6 for October 20, 1993. This circulation pattern likely limited mixing of the inflow with the epilimnion, as was confirmed by model simulations of material transport.

Rhodhiss Lake seldom had a clearly defined thermocline even when other physical water-quality characteristics, such as DO concentration, indicated the reservoir was stratified. During stratification, temperature decreased fairly uniformly with depth by as much as 15 °C (fig. 5, July 14, 1993).

Stratification gradually weakened during the fall as surface temperatures cooled, and the reservoir underwent complete thermal mixing in December 1993. During monthly sampling visits from December 1993 through February 1994, water temperatures were fairly uniform throughout the water column, which would indicate isothermal, or well-mixed, conditions. However, the model indicated that the winter mixing period actually comprised a series of mixing events interrupted by brief periods of stratification. Likewise, autumnal stratification was interrupted by at least one transient mixing event. Rhodhiss Lake began to restratify in March 1994 (fig. 5).

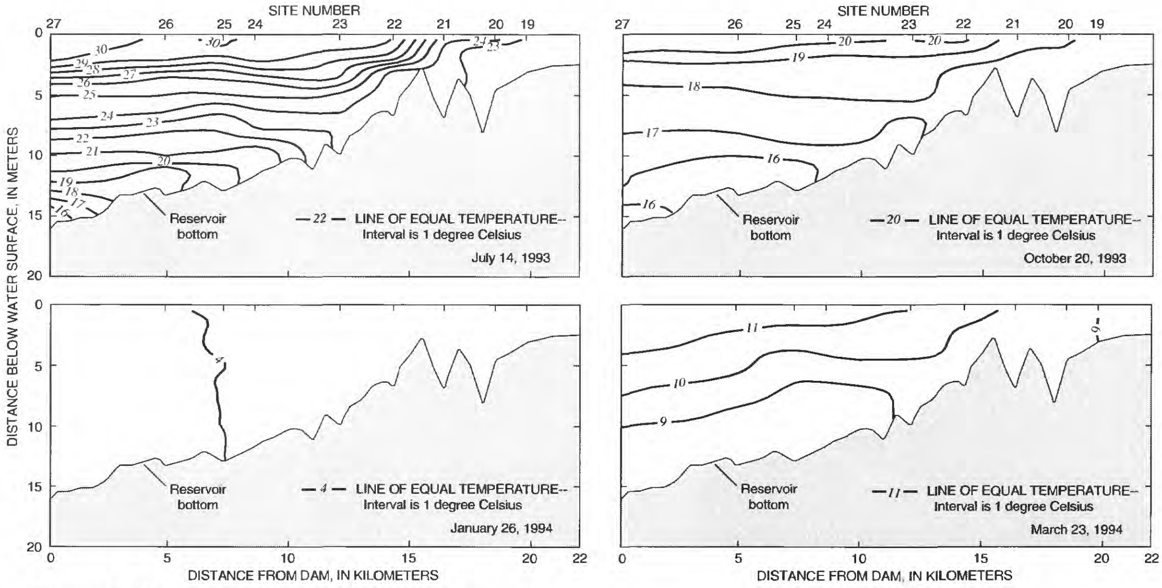

Figure 5. Water temperature in Rhodhiss Lake.

Dissolved Oxygen

Seasonal patterns of DO concentration (fig. 6) were strongly related to thermal patterns, and also were influenced by algal photosynthesis and in-lake oxygen demand. DO was at nearly saturated concentrations and well mixed in the water column from January through April 1993. With the onset of stratification in May 1993, demands from hypolimnetic waters and bed sediments began to deplete hypolimnetic oxygen in the lacustrine zone of the reservoir. Low DO concentrations persisted in the hypolimnion through the summer of 1993 (fig. 6). During the same time, DO concentrations in the upper few meters of the water column were at or above saturation levels. In general, an abrupt oxygen decline was observed 3 to 4 m below the water surface, which was just beneath the estimated depth of the euphotic zone.

In the fall of 1993, hypolimnetic anoxia began to abate, while supersaturated DO concentrations were noted in reservoir surface waters (fig. 6). In the fall, inflows were low, air temperatures were still warm, and concentrations of inorganic nutrients in the lacustrine zone increased slightly from summertime lows. These factors may have contributed to increased phytoplankton activity and thus, to elevated DO concentrations in the epilimnion. In fact, the highest chlorophyll concentrations during the investigation were from September through November. From December 1993 through early spring 1994, DO returned to nearly saturated concentrations and well mixed conditions (fig. 6).

North Carolina water-quality standards require that waters maintain a daily average DO concentration of at least 5.0 mg/L and a minimum instantaneous concentration of at least 4.0 mg/L (North Carolina Department of Environment, Health, and Natural Resources, 1994a). No violations of this standard were observed at Lower Creek site 53 or in the riverine zone of Rhodhiss Lake. Downstream from site 23 in Rhodhiss Lake, concentrations less than 5.0 mg/L typified bottom waters during summer stratification (fig. 6); however, low concentrations in lake-bottom waters are not considered violations of the State standard if caused by natural conditions (North Carolina Department of Environment, Health, and Natural Resources, 1994a).

pH

In general, pH at the upstream end of the reservoir was close to 7.0 throughout the investigation (fig. 7). During winter and early spring of 1993 and 1994, pH was fairly uniform throughout the reservoir. However, beginning in May 1993 and continuing through summer and fall, pH was elevated and strongly stratified near the surface in the lacustrine zone of Rhodhiss Lake (fig. 7). High pH coincided with supersaturated DO concentrations (fig. 6), suggesting that algal photosynthesis may have influenced these water-quality characteristics. Actively photosynthesizing algae consume dissolved carbon dioxide and shift the carbonate buffering equilibrium toward more alkaline (higher pH) conditions (Wetzel, 1983).

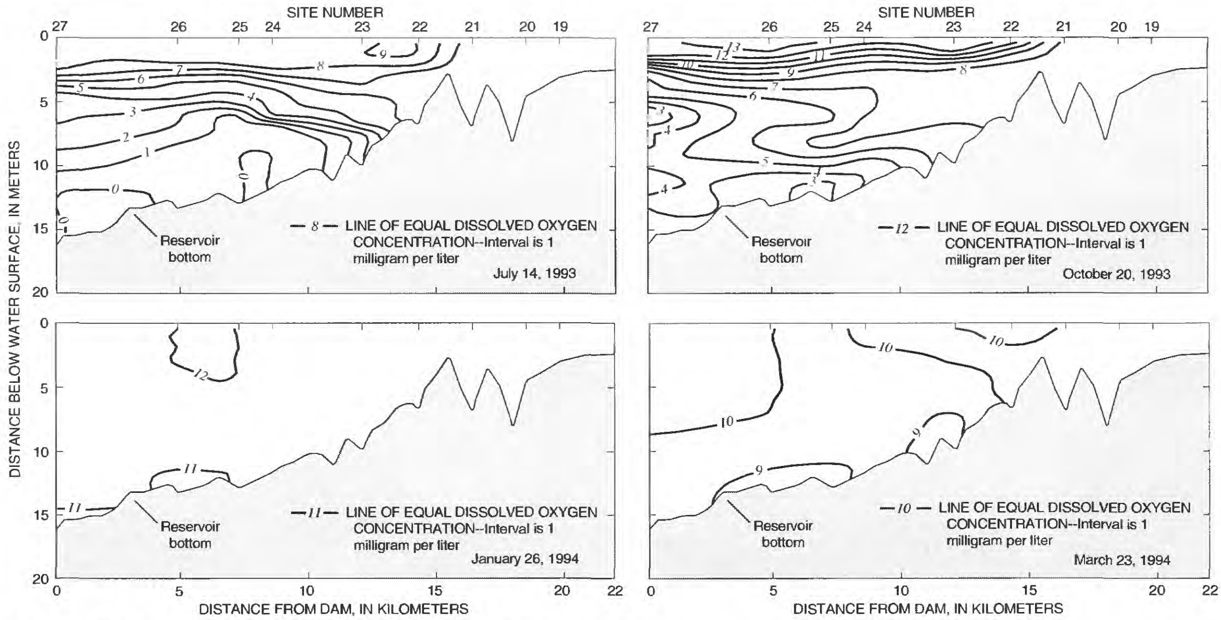

Figure 6. Dissolved oxygen concentration in Rhodhiss Lake.

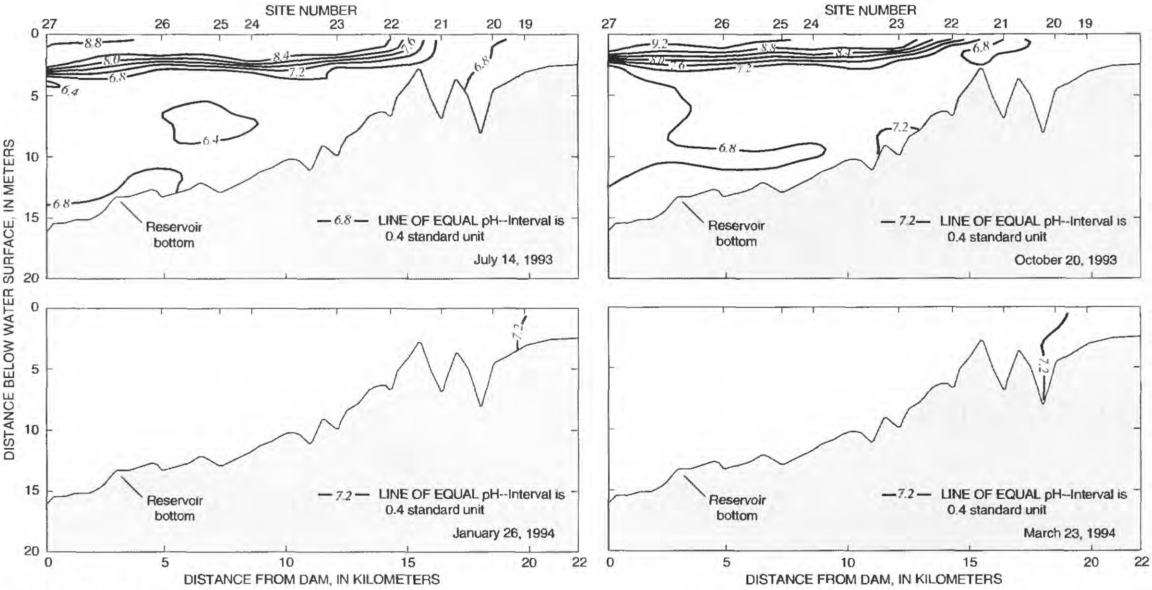

Figure 7. pH in Rhodhiss Lake.

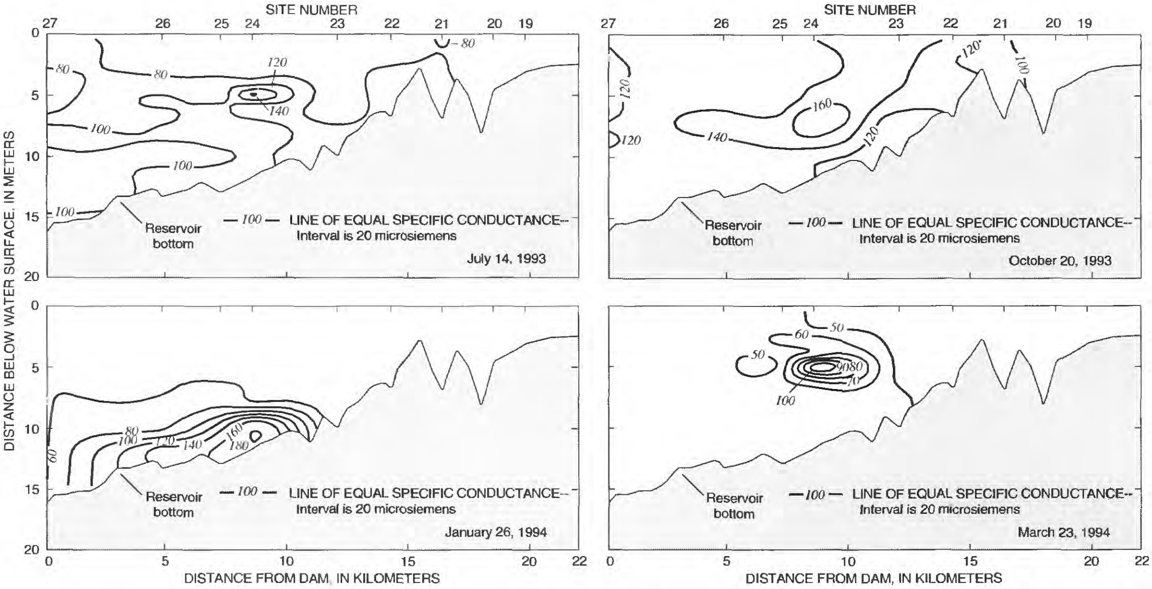

Figure 8. Specific conductance in Rhodhiss Lake.

Specific Conductance

Specific conductance in Rhodhiss Lake was influenced by headwater inflow and by inputs from wastewater-treatment plants. Specific conductance generally increases with the concentration of dissolved inorganic substances; therefore, inflows receiving wastewater effluents or nonpoint sources of pollutants typically have higher specific conductance levels than inflows undisturbed by human activity. In a minimally disturbed forested watershed near the Rhodhiss Lake study area (High Shoals Creek near Dysartsville, McDowell County), specific conductance levels ranged from 11 to 34 microsiemens per centimeter (µS/cm) (Caldwell, 1992). In contrast, concentrations at site 20 in the headwaters of Rhodhiss Lake ranged from 33 to 116 µS/cm with a median of 45 µS/cm. Specific conductance values at Lower Creek site 53 ranged from 38 to 86 µS/cm with a median of 73 µS/cm (Jaynes, 1994).

Several municipalities, industries, schools, and other facilities discharge treated wastewater into the Rhodhiss Lake watershed. The largest facility in the Lower Creek drainage is the City of Lenoir wastewater-treatment plant, which discharged an average of 0.097 m^3/s into Lower Creek upstream from site 53. The City of Morganton wastewater-treatment plant discharged an average of 0.286 m^3/s into the Catawba River upstream from Rhodhiss Lake site 19. Both Lower Creek and the Catawba River upstream from Rhodhiss Lake received effluent discharges from numerous smaller facilities as well.

In general, specific conductance values throughout Rhodhiss Lake were low during winter and spring, then gradually increased through summer to peak in the fall (fig. 8). Regression analysis showed that specific conductance was inversely related to streamflow at Rhodhiss Lake site 20 (r^2=0.78: P<0.0001) and at Lower Creek site 53 (r^2=0.95; P<0.0001). During the drier summer and fall months, point-source discharges accounted for a greater proportion of the inflow to Rhodhiss Lake and Lower Creek; thus, specific conductance was higher then than during high streamflow periods.

A peak in specific conductance frequently was observed near mid-reservoir (fig. 8), downstream from the outfall of a municipal wastewater-treatment plant. During this investigation, this facility discharged an average of 0.216 m^3/s into the mainstem of Rhodhiss Lake about 2 km upstream from site 24 at a depth of about 9 m. Elevated specific conductance values resulting from this submerged discharge were measured during 16 of the 21 sampling trips, most consistently when the reservoir was stratified. During stratified periods, a horizontal plume of high-conductivity water extended toward the dam at a depth of about 4 to 6 m, indicating that this layer tended not to mix with adjacent layers of water. In the absence of thermal stratification, such as during January 1994, peaks either were not observed or were located near the reservoir bottom (fig. 8).

Chemical and Biological Constituents

Samples were collected and analyzed for several chemical and biological constituents at four sites in the Rhodhiss Lake study area—in the headwaters, mid-reservoir, and forebay of Rhodhiss Lake (sites 20, 24, and 27, respectively; fig. 2), and in Lower Creek (site 53, fig. 2). Detailed analytical results were reported by Jaynes (1994). In this section, the spatial and temporal distributions of selected constituents are summarized, and when applicable, results are discussed in relation to water-quality criteria.

Suspended Solids

Suspended solids include inorganic particles, such as silts and clays; and organic particles, such as bacteria, algae, zooplankton, and detritus. High concentrations of suspended solids can decrease water clarity, shade rooted vegetation, clog the gills of fish and aquatic invertebrates, and increase the cost of drinking-water treatment. In this investigation, total suspended solids (TSS) and volatile suspended solids (VSS) were measured.

The highest concentrations of TSS were observed at Lower Creek site 53 (fig. 9; table 2). The maximum TSS concentration of 497 mg/L was measured at this site

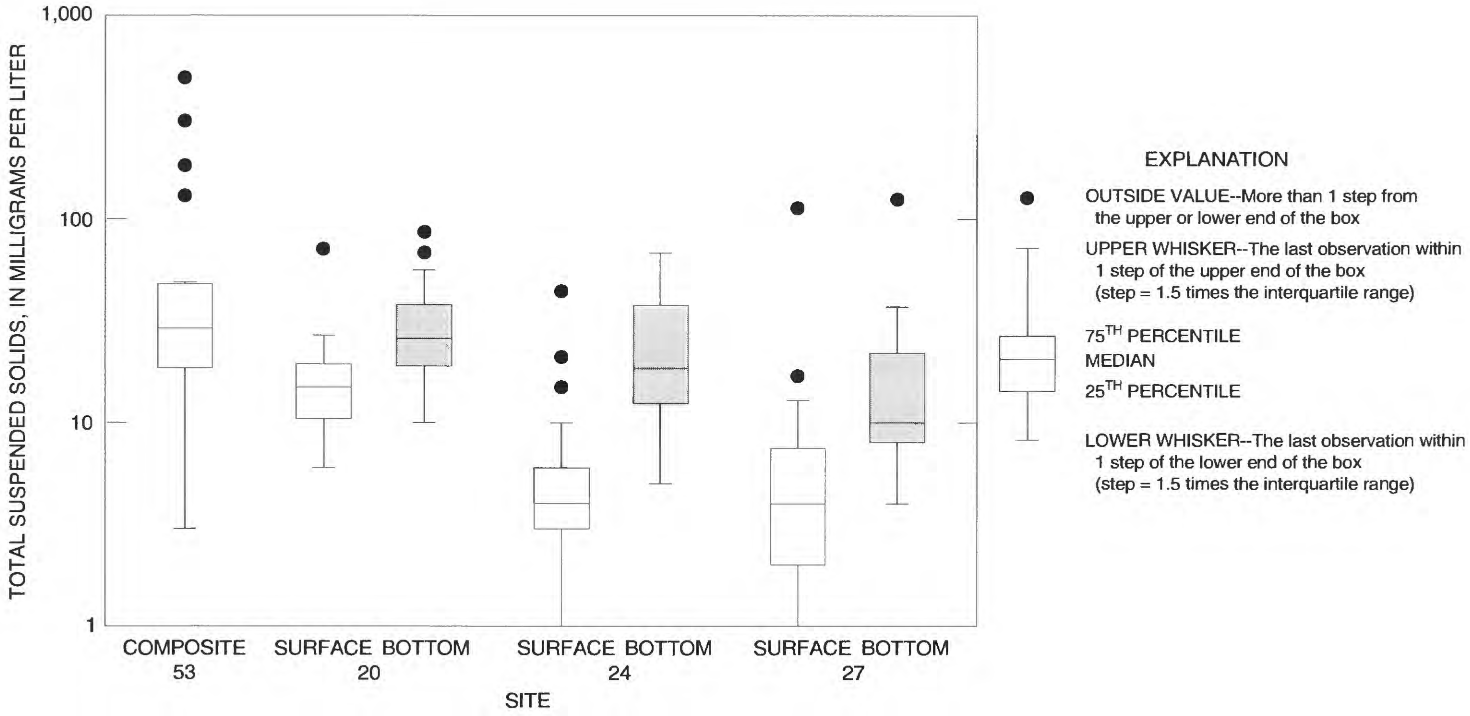

Figure 9. Total suspended solids concentrations at sites in the Rhodhiss Lake study area, January 1993 through March 1994.

Table 2. Statistical summary of selected chemical and biological constituents at Lower Creek site 53 and Rhodhiss Lake sites 20, 24, and 27, January 1993 through March 1994

[n, sample size; Max, maximum; Min, minimum; Mean, arithmetic mean; Med, median; mg/L, milligrams per liter; <, less than; N, nitrogen; P, phosphorus; µg/L, micrograms per liter; --, no data; cols/100 mL, colonies per 100 milliliters]

CONSTITUENT	Site 53					Site 20					Site 24					Site 27				
	n	Max	Min	Mean	Med	n	Max	Min	Mean	Med	n	Max	Min	Mean	Med	n	Max	Min	Mean	Med
	Water-column composite samples					Near-surface samples														
Solids, total suspended (mg/L)	21	497	3	74	29	21	71	6	18	15	21	44	1	7	4	21	114	< 1	10*	4
Solids, volatile suspended (mg/L)	21	55	2	11	6	21	9	1	4	4	21	9	1	3	2	20	9	< 1	3*	1
Nitrogen, nitrite plus nitrate dissolved (mg/L as N)	21	.802	<.005	.522*	.533	21	.306	.118	.227	.237	21	.285	<.005	.124*	.124	21	.316	<.005	.119*	.065
Nitrogen, ammonia dissolved (mg/L as N)	21	.316	.008	.099	.088	21	.248	.011	.100	.089	21	.094	<.002	.034*	.026	21	.152	<.002	.035*	.016
Nitrogen, ammonia + organic total (mg/L as N)	21	2.8	.20	.54	.40	21	.80	<.20	.30*	.30	21	.90	<.20	.29*	.20	21	.80	<.20	.31*	.30
Nitrogen, organic total (mg/L as N)	21	2.6	.11	.44	.28	21	.69	.08	.23	.18	21	.90	.11	.29	.20	21	.78	.13	.29	.24
Phosphorus, total (mg/L as P)	21	.561	.028	.117	.086	21	.154	.026	.057	.053	21	.167	.024	.050	.044	21	.113	.020	.038	.034
Phosphorus, orthophosphate dissolved (mg/L as P)	21	.045	.004	.022	.021	21	.052	.003	.019	.014	21	.029	<.001	.007*	.004	21	.022	<.001	.005*	.002
Chlorophyll *a* (photic-zone composite; µg/L)	--	--	--	--	--	21	3.8	0.2	1.0	.8	21	52	0.1	9.1	5.5	21	21	0.1	5.9	3.1
Fecal coliform bacteria (grab sample; cols/100 mL)	21	5,000	10	360†	320	19	1,300	20	130†	120	19	980	< 4	39†	40	19	2,400	< 1	26†	40
						Near-bottom samples														
Solids, total suspended (mg/L)	--	--	--	--	--	21	86	10	31	26	20	68	5	24	18	21	124	4	20	10
Solids, volatile suspended (mg/L)	--	--	--	--	--	21	12	2	6	6	20	13	1	5	5	21	16	1	4	3
Nitrogen, nitrite plus nitrate dissolved (mg/L as N)	--	--	--	--	--	21	.315	.006	.218	.238	21	.285	.118	.202	.206	21	.330	.005	.169	.193
Nitrogen, ammonia dissolved (mg/L as N)	--	--	--	--	--	21	.259	.019	.101	.085	21	.668	.031	.176	.135	21	1.10	.002	.343	.147
Nitrogen, ammonia + organic total (mg/L as N)	--	--	--	--	--	21	.60	<.20	.28*	.20	21	.90	<.20	.364*	.30	21	1.6	<.20	.54*	.40
Nitrogen, organic total (mg/L as N)	--	--	--	--	--	21	.38	.07	.19	.16	21	.53	.02	.20	.17	21	.50	.10	.21	.19
Phosphorus, total (mg/L as P)	--	--	--	--	--	21	.129	.023	.065	.062	21	.170	.033	.088	.086	21	.288	.034	.080	.052
Phosphorus, orthophosphate dissolved (mg/L as P)	--	--	--	--	--	21	.052	.001	.017	.015	21	.085	.007	.023	.014	21	.130	.001	.025	.014

* Value is estimated by using a log-probability regression to predict the values of data below the detection limit.
† Geometric mean

during a high-flow event in March 1993. As expected, concentrations of solids were directly related to streamflow at Lower Creek site 53 and Rhodhiss Lake site 20 throughout this investigation, because swiftly moving water can carry more solids in suspension.

Among the three sites in Rhodhiss Lake, median concentrations of TSS in surface samples decreased from 15 mg/L at site 20 to 4 mg/L at sites 24 and 27 (table 2). At each reservoir site, bottom samples had higher TSS than surface samples (fig. 9). Median concentrations of TSS in bottom samples decreased in a downstream direction from 26 mg/L at site 2, to 18 mg/L at site 24 and 10 mg/L at site 27 (table 2).

Volatile suspended solids represent the organic component of total suspended solids. VSS constituted a relatively small proportion of the TSS at most sampling sites. Median concentrations were 6 mg/L in Lower Creek (site 53), 1 to 4 mg/L in reservoir surface samples, and 3 to 6 mg/L in reservoir bottom samples (table 2).

Nitrogen

Nitrogen and phosphorus are the primary nutrients that regulate algal productivity in lakes and reservoirs (Wetzel, 1983). During this investigation, three nitrogen components were measured—nitrite plus nitrate (NO_2+NO_3); ammonia (NH_4); and total organic nitrogen plus ammonia. NO_2+NO_3 and NH_4 are inorganic forms that are readily assimilated by phytoplankton. Total organic nitrogen (TON), computed by subtracting NH_4 from TON plus ammonia, is present in water as dissolved amino acids and polypeptides and as living or detrital particulate matter.

NO_2+NO_3 concentrations were higher in Lower Creek (site 53) than in Rhodhiss Lake (sites 20, 24, and 27) (fig. 10). With one exception, all samples from site 53 contained at least 0.300 mg/L of NO_2+NO_3 as N,

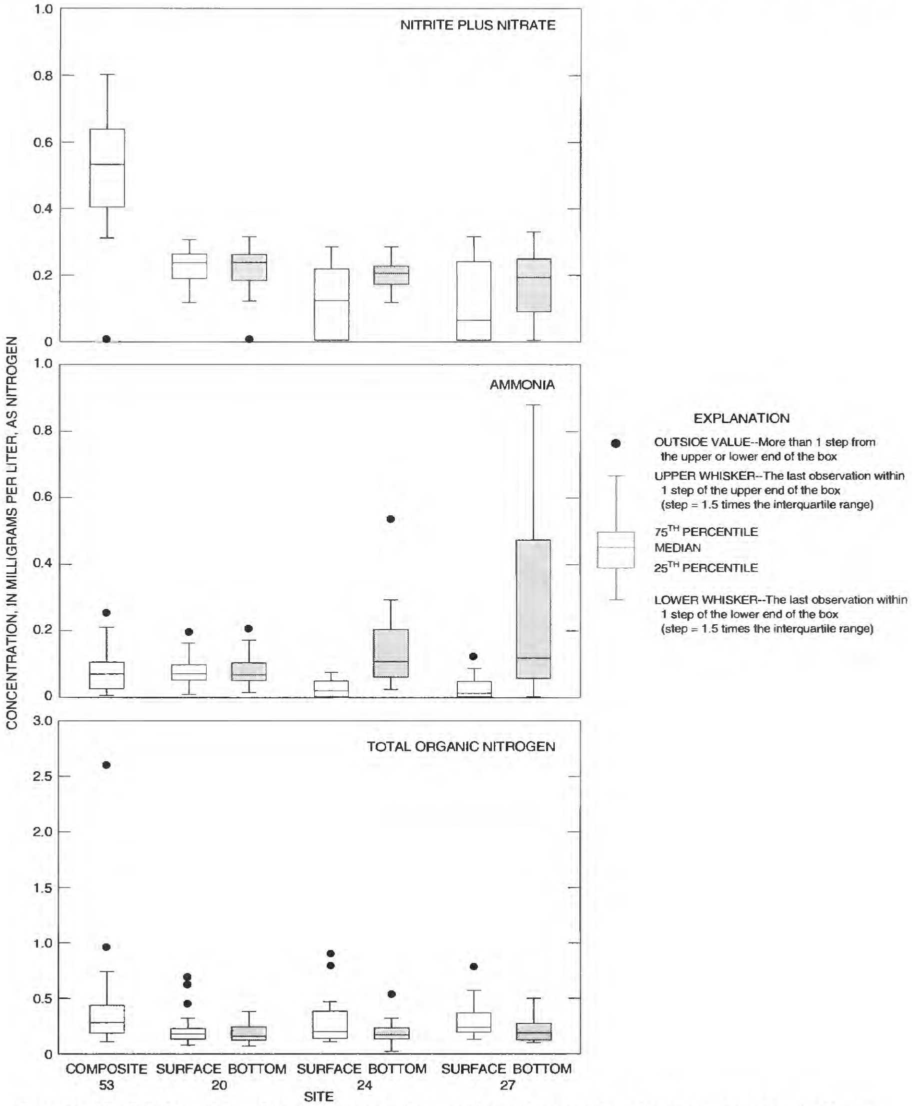

Figure 10. Nitrite plus nitrate, ammonia, and total organic nitrogen concentrations at sites in the Rhodhiss Lake study area, January 1993 through March 1994.

and the median concentration was 0.533 mg/L. In contrast, all reservoir samples had concentrations less than or equal to 0.330 mg/L (table 2). Concentrations of NO_2+NO_3 in Lower Creek were elevated compared to two nearby tributaries to Lake Hickory, each of which had a median concentration of 0.254 mg/L during the same period of time (Jaynes, 1994). Elevated concentrations in Lower Creek reflected inputs from numerous domestic and industrial discharges and from nonpoint sources including urban and agricultural runoff.

In Rhodhiss Lake, median concentrations of NO_2+NO_3 in near-surface samples were highest in the headwaters (site 20), then decreased at the two downstream sites (fig. 10; table 2). The three sites had similar surface concentrations of NO_2+NO_3 during cool months when the reservoir was unstratified. However, surface NO_2+NO_3 concentrations declined at sites 24 and 27 with the onset of warm weather and stratification in May and remained low through mid-September 1993 (fig. 11). In fact, most of the surface samples collected from sites 24 and 27 during this period contained less than the laboratory reporting level of

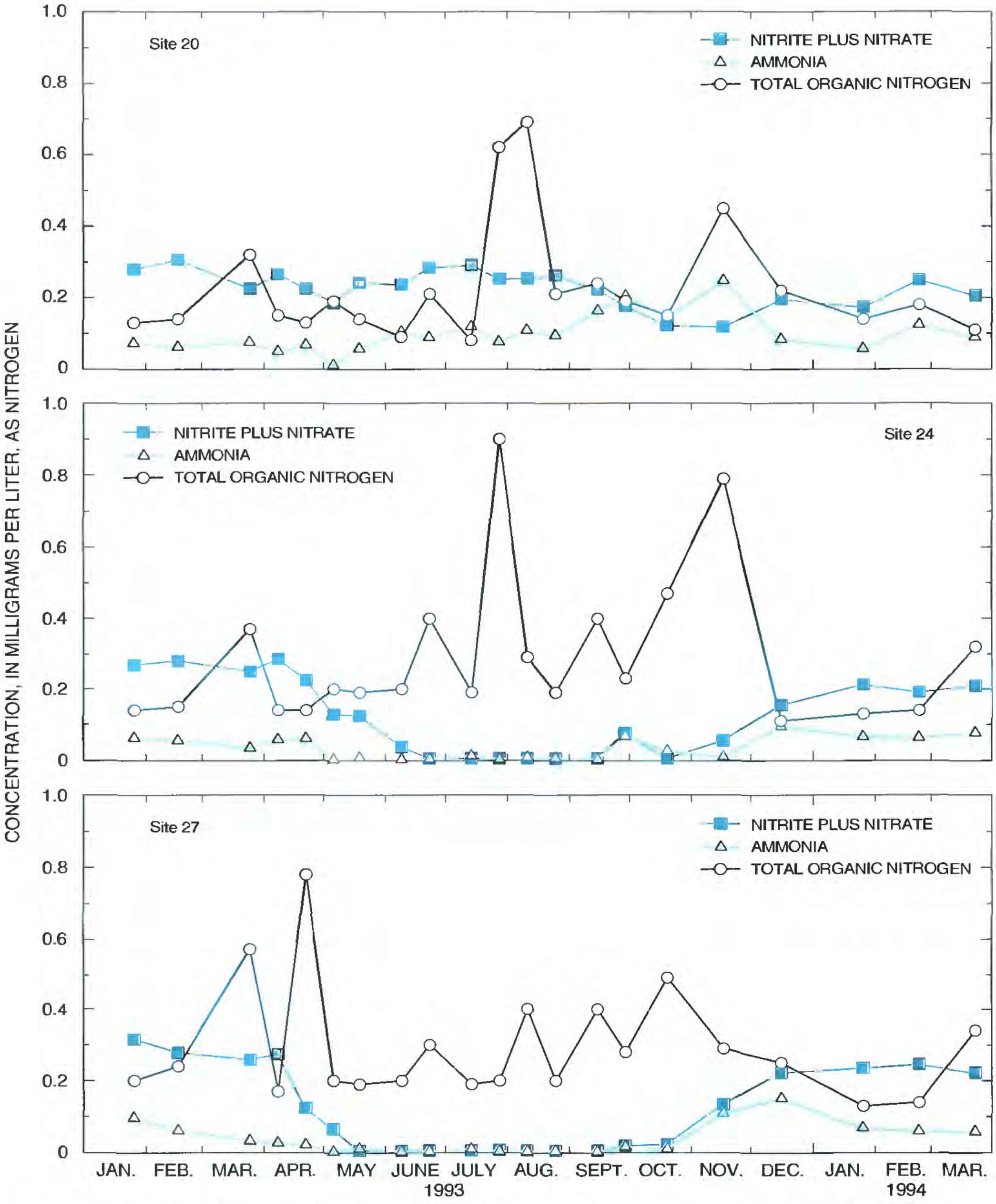

Figure 11. Nitrite plus nitrate, ammonia, and total organic nitrogen in Rhodhiss Lake surface samples, January 1993 through March 1994.

0.005 mg/L of NO_2+NO_3. This period coincided with the growing season. when actively growing phytoplankton likely depleted nitrate from the epilimnion.

Seasonal differences between reservoir sites also were observed in bottom-water concentrations of NO_2+NO_3 (fig. 12), but another mechanism probably was responsible. Low concentrations of NO_2+NO_3 at site 27 were associated with low DO concentrations (<1.0 mg/L) from late June through September. Under anaerobic conditions, such as in the hypolimnion of Rhodhiss Lake, nitrite and nitrate are rapidly converted to nitrogen gas by bacterial denitrification. The hypolimnion at site 24 had transient anoxia during the summer of 1993, which may have accounted for the intermediate levels of NO_2+NO_3 observed in bottom waters at this location.

Median concentrations of NH_4 were similar at Lower Creek site 53 (0.088 mg/L as N) and headwaters site 20 (0.089 mg/L, surface; 0.085 mg/L, bottom) during the investigation (table 2). Median surface concentrations were lower at mid-lake and at the forebay than in the headwaters of Rhodhiss Lake (fig. 10; table 2). Concentrations less than or equal to 0.010 mg/L characterized sites 24 and 27 from May through mid September 1993 (fig. 11). Low concentrations coincided with the growing season and well-oxygenated conditions in the epilimnion, suggesting that phytoplanktonic assimilation and biochemical oxidation

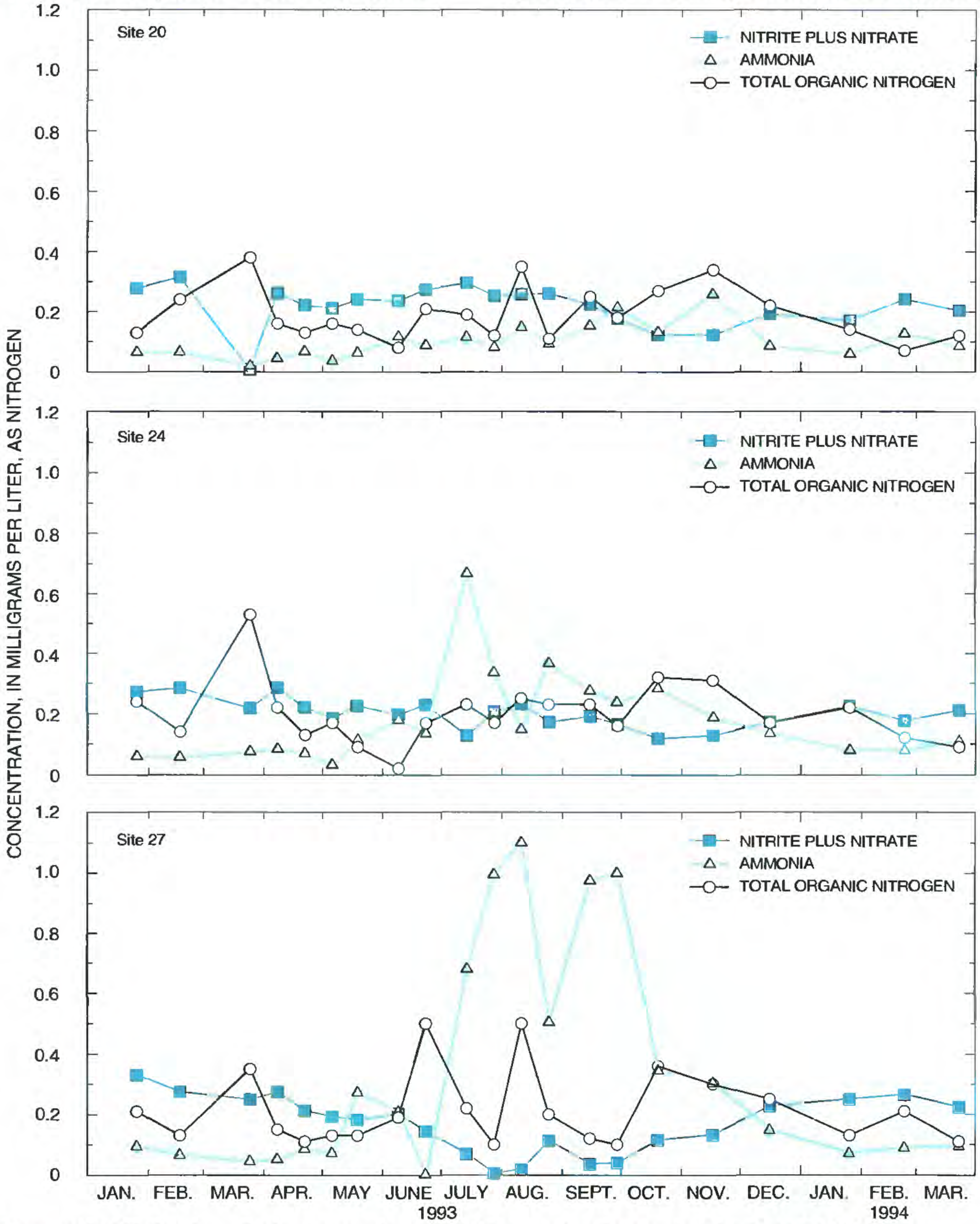

Figure 12. Nitrite plus nitrate, ammonia, and total organic nitrogen in Rhodhiss Lake bottom samples, January 1993 through March 1994.

processes reduced the amount of NH_4 present in the epilimnion.

At site 27, the hypolimnion had high concentrations of NH_4 from mid-June through September 1993 (fig. 12), the same period that NO_2+NO_3 and DO decreased in the hypolimnion. Elevated concentrations of NH_4 likely resulted from decomposition of organic matter in both hypolimnetic waters and bed sediments under anaerobic conditions. The maximum NH_4 concentration of 1.10 mg/L was observed on August 11, 1993. Hypolimnetic concentrations of NH_4 also increased during the summer at site 24, but to a lesser degree than at site 27 (fig. 12). Results showed this reduced form of nitrogen predominated during anaerobic conditions.

Total organic nitrogen was calculated by subtracting NH_4 concentrations from total organic nitrogen plus ammonia concentrations for each location and sampling date. Concentrations of total organic nitrogen plus ammonia less than the laboratory reporting level of 0.20 mg/L were considered to be equal to 0.20 mg/L for this computation. Likewise, concentrations of NH_4 less than the laboratory reporting level of 0.002 mg/L were considered to be equal to 0.002 mg/L for computational purposes.

Total organic nitrogen ranged from 0.11 to 2.6 mg/L as N at Lower Creek site 53 (table 2); however, all but one sample had less than 1.0 mg/L (fig. 10). The median concentration of 0.28 mg/L in Lower Creek was slightly higher than at the three reservoir sites (table 2).

Concentrations of TON in Rhodhiss Lake ranged from 0.02 to 0.90 mg/L (table 2). Although TON varied more between dates than between sites, concentrations did not follow a strong seasonal pattern (figs. 11 and 12). The maximum reservoir concentration of 0.90 mg/L occurred near the surface at site 24 on July 28, 1993. At sites 24 and 27, concentrations of TON generally were higher in surface waters than in bottom waters and constituted the majority of the total nitrogen (TN) present during summer stratification (figs. 11 and 12).

Phosphorus

Phosphorus components measured during this investigation include orthophosphate (PO_4) and total phosphorus (TP). PO_4 is a dissolved, inorganic form of phosphorus that is readily available to phytoplankton.

Concentrations of TP in Lower Creek ranged from 0.028 to 0.561 mg/L, with a median concentration of 0.086 mg/L (fig. 13; table 2). The maximum, which also was the highest concentration at any site during the investigation, was observed in conjunction with heavy rainfall in March 1993 (fig. 14). TP was much higher in Lower Creek than in two nearby tributaries to Lake Hickory, where median concentrations were 0.016 and 0.027 mg/L during the same period (Jaynes, 1994). In an earlier investigation, researchers reported a mean TP concentration of 0.01 mg/L in an unpolluted stream located in the same geochemical zone as Lower Creek (Simmons and Heath, 1982). Furthermore, median TP concentrations of 0.01 and 0.02 mg/L characterized minimally disturbed streams in forested watersheds near the Rhodhiss Lake study area (Caldwell, 1992).

Concentrations of PO_4 ranged from 0.004 to 0.045 mg/L as P at Lower Creek site 53. The median concentration of 0.021 mg/L in Lower Creek was higher than median concentrations at the three sites in Rhodhiss Lake (fig. 13; table 2) and in two tributaries to Lake Hickory, where medians were less than or equal to 0.001 mg/L (Jaynes, 1994).

Relations between streamflow and phosphorus concentrations indicate that Lower Creek was influenced by both point and nonpoint sources. PO_4 was inversely related to streamflow in Lower Creek; that is, PO_4 concentrations generally were highest during low-flow periods, such as the summer of 1993 (fig. 14), when point-source inputs constituted a larger proportion of the streamflow. TP concentrations also decreased as streamflow increased from low to moderate flow (about 4.2 m^3/s) conditions. However, when heavy rainfall caused streamflow to rise above this level, such as during March 1993 and January and February 1994 (fig. 14), TP concentrations increased, presumably as a result of inputs from nonpoint sources and resuspension of bed sediments. This assumption was supported by the fact that PO_4 concentrations in Lower Creek remained low during rainfall events, indicating that most of the increase in TP was associated with particulate matter or dissolved organic compounds.

A similar relation between phosphorus and streamflow was observed at site 20 in the headwaters of Rhodhiss Lake. Concentrations of TP decreased as streamflow increased from low-flow to moderately high-flow conditions. When substantial rainfall caused streamflow to exceed approximately 77 m^3/s, TP increased while PO_4 remained low. Like Lower Creek, the Catawba River upstream from Rhodhiss Lake received nutrient inputs from both point and nonpoint sources.

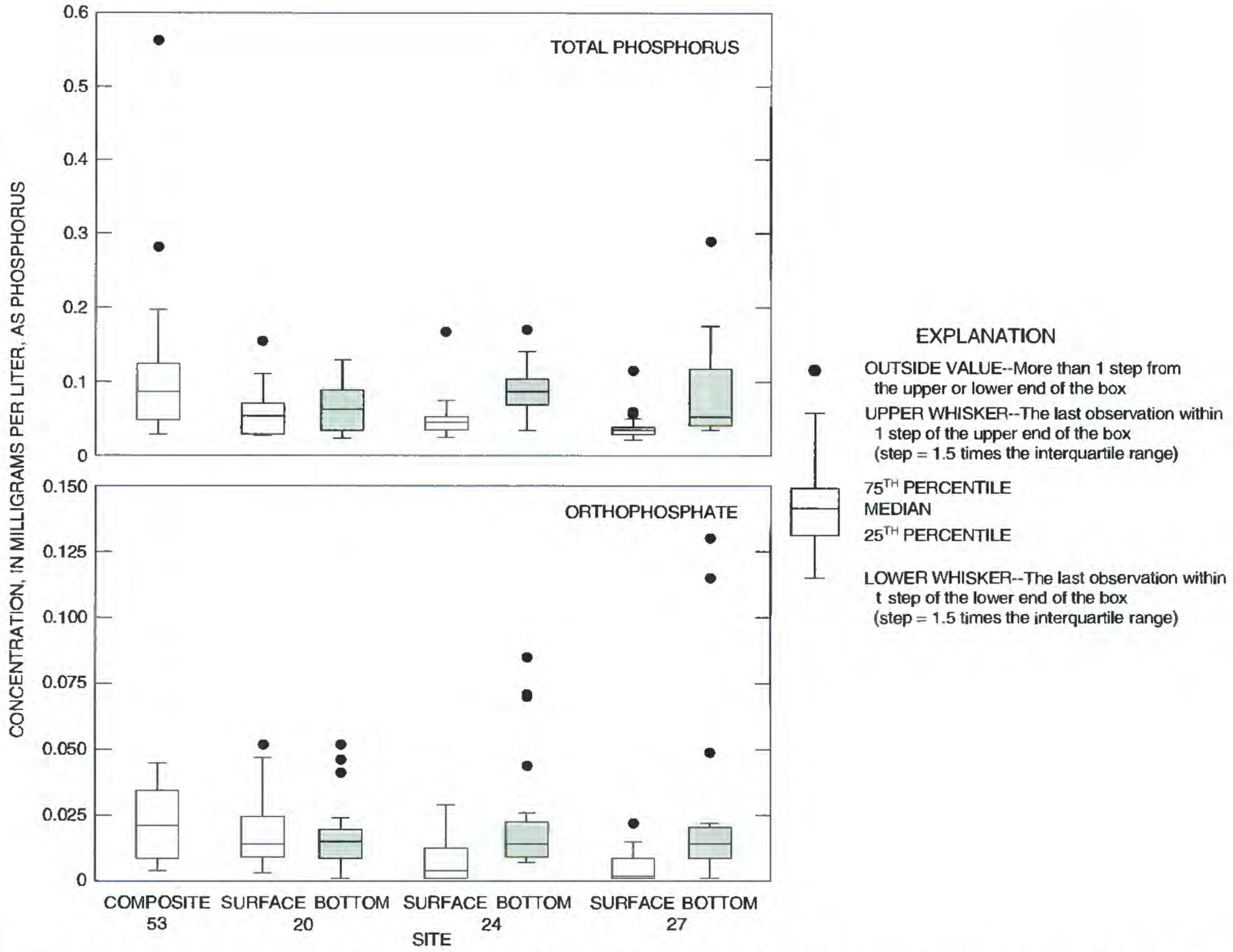

Figure 13. Box plots showing the distribution of total phosphorus and orthophosphate concentrations at sites in the Rhodhiss Lake study area, January 1993 through March 1994.

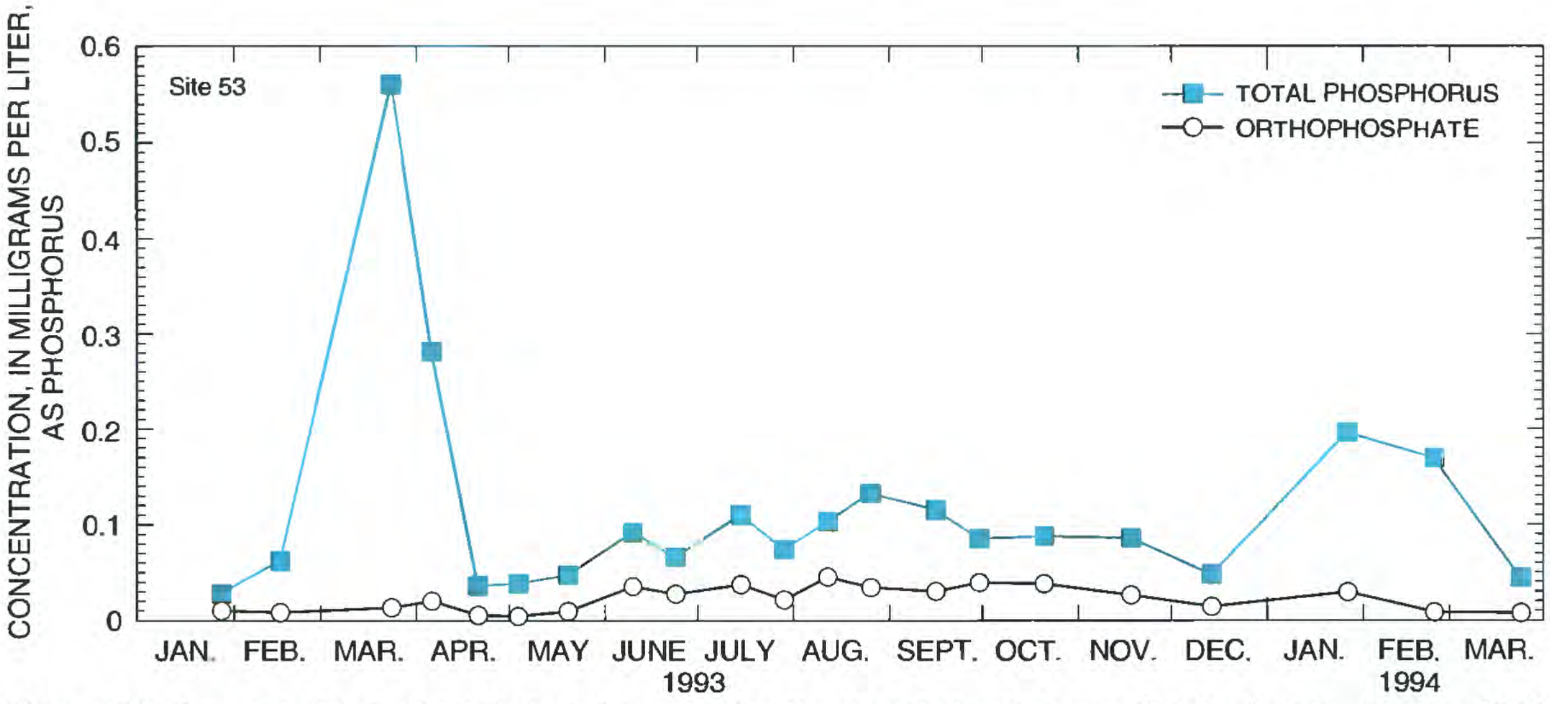

Figure 14. Concentrations of total phosphorus and orthophosphate at Lower Creek site 53, January 1993 through March 1994.

In Rhodhiss Lake, median concentrations of TP in near-surface samples decreased from 0.053 mg/L in the headwaters, to 0.044 mg/L at mid-reservoir and 0.034 mg/L in the forebay (fig. 13; table 2). At mid-reservoir site 24 and forebay site 27, near-surface concentrations of TP did not follow seasonal patterns; however, elevated concentrations were observed at all three reservoir sites following very heavy rainfall in March 1993 (fig. 15).

Surface concentrations of PO_4 also were higher in the headwaters than at the other two reservoir sites (fig. 13; table 2). Epilimnetic PO_4 concentrations decreased to very low levels at sites 24 and 27 during the summer of 1993. At mid-reservoir, PO_4 concentrations were less than or equal to 0.002 mg/L from June through mid-September 1993. In the forebay, PO_4 concentrations were less than or equal to 0.001 mg/L from mid-May through September 1993 (fig. 15).

At sites 24 and 27, concentrations of TP and PO_4 in bottom waters were higher and more variable than in surface waters (fig. 13). Maximum concentrations of TP (0.288 mg/L) and of PO_4 (0.130 mg/L) were observed in the hypolimnion at site 27 on August 11, 1993 (fig. 16; table 2). It is common for the hypolimnion of a productive lake to contain elevated phosphorus levels, particularly during the later stages of stratification (Wetzel, 1983).

Based on average concentrations of TP observed in the epilimnion, Rhodhiss Lake would be classified eutrophic (Wetzel, 1983) and would be expected to support abundant algal biomass. However, as discussed in the next section, nuisance levels of algae were not frequently observed. In reservoirs like Rhodhiss Lake, factors other than nutrient availability, such as short hydraulic residence time and high nonalgal turbidity, frequently limit phytoplankton production and thus suppress the manifestation of eutrophic symptoms (Kimmel and Groeger, 1984; Walker, 1984).

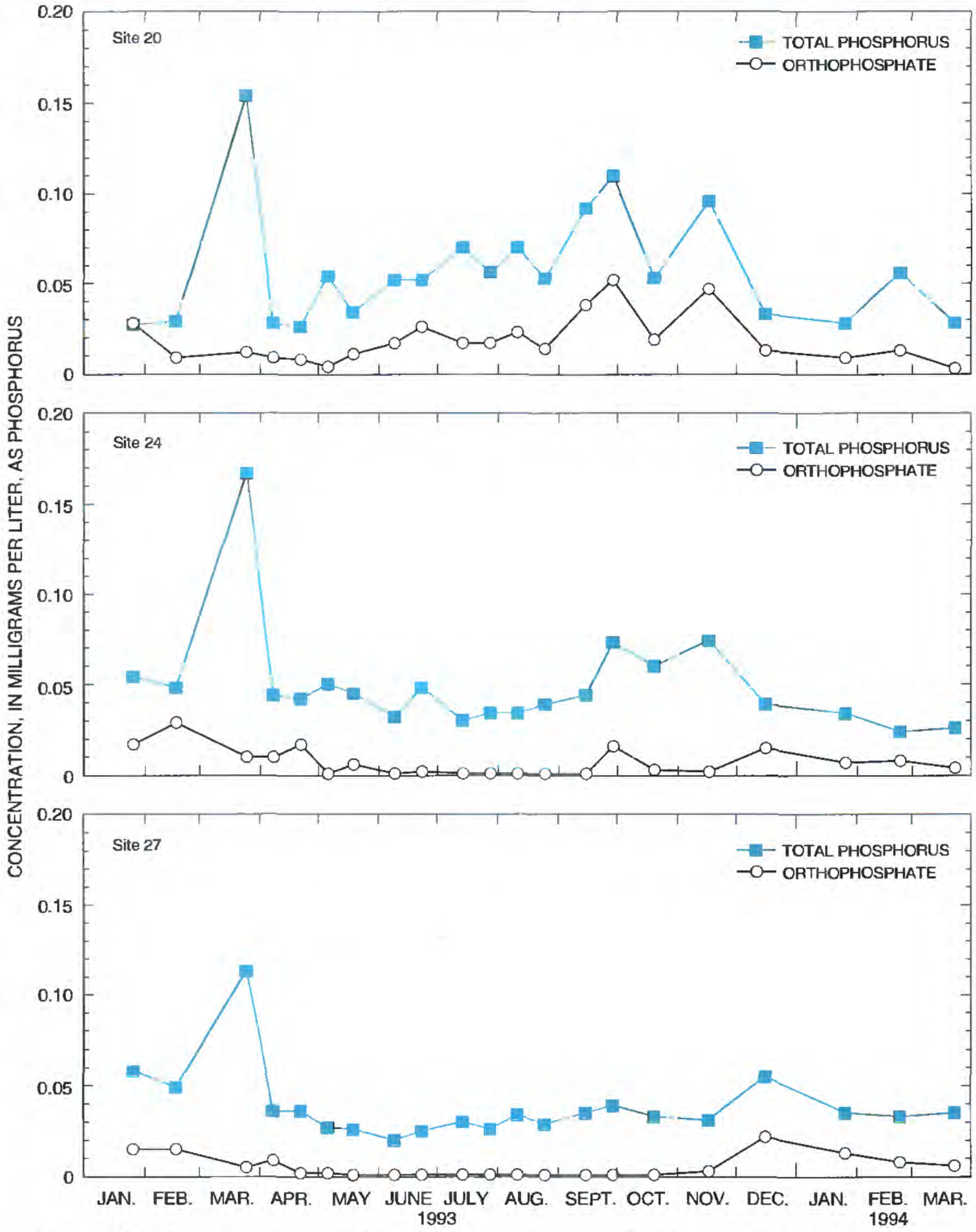

Figure 15. Concentrations of total phosphorus and orthophosphate over time in Rhodhiss Lake surface samples, January 1993 through March 1994.

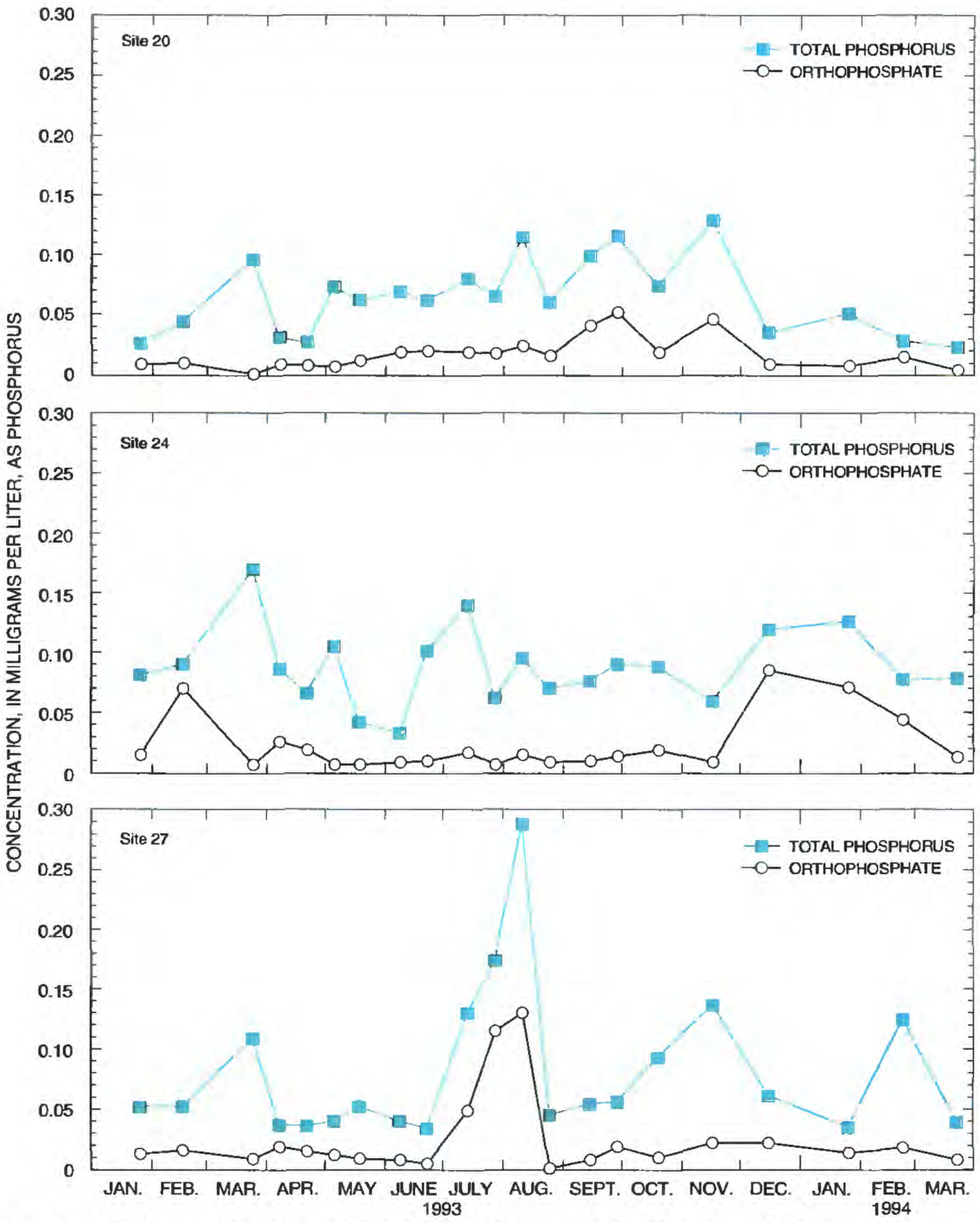

Figure 16. Concentrations of total phosphorus and orthophosphate in Rhodhiss Lake bottom samples, January 1993 through March 1994.

Chlorophyll *a*

Chlorophyll *a* often is used as a measure of algal biomass in lakes and reservoirs. Concentrations of chlorophyll *a* were measured at reservoir sites 20, 24, and 27 (fig. 2). Throughout the investigation, concentrations were low in the riverine headwaters of Rhodhiss Lake, never exceeding 3.8 micrograms per liter (µg/L) (fig. 17; table 2). Concentrations of chlorophyll *a* were higher in the lacustrine part of the reservoir, and tended to be higher at mid-reservoir site 24 than at forebay site 27 (fig. 17). Mid-reservoir peaks in chlorophyll *a* concentrations have been observed at other southeastern reservoirs (Kennedy and others, 1982; Radtke, 1986; Pickett and Harvey, 1988). In this zone, slower water velocity, increased water clarity (from particle settling), and abundant nutrients permit plankton communities to establish and flourish. In addition to these expected features, Rhodhiss Lake also receives a discharge of treated wastewater approximately 2 km upstream from site 24. This source delivers additional nutrients that may fuel phytoplankton growth.

As expected, concentrations of chlorophyll *a* were very low during winter months at all sampling locations. Higher concentrations were observed at sites 24 and 27 from spring through fall (fig.17). From May through September 1993, the mean chlorophyll *a* concentration was 10 µg/L at site 24 and 7.9 µg/L at site 27. A maximum concentration of 52 µg/L was observed at site 24 on November 17, 1993. This value exceeded the State water-quality standard for chlorophyll *a* of 40 µg/L (North Carolina Department of Environment, Health, and Natural Resources, 1994a), and occurred when tributary inflows to the reservoir were low and air temperatures were unseasonably warm (Jaynes, 1994). No other samples contained concentrations of chlorophyll *a* greater than the State standard.

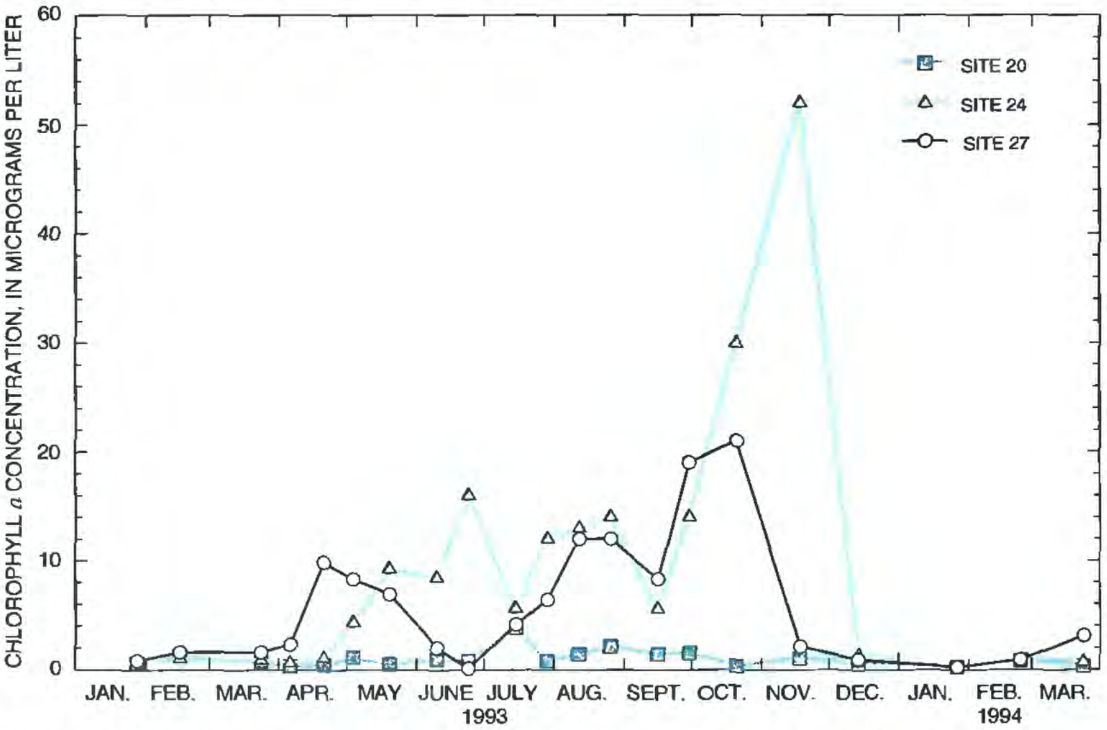

Figure 17. Concentrations of chlorophyll *a* in photic-zone samples from Rhodhiss Lake sites 20, 24, and 27, January 1993 through March 1994.

Bacteria

Fecal coliform bacteria occur in the digestive tract of warm-blooded animals, and therefore are used to indicate the presence of human and other animal wastes in the aquatic environment. North Carolina has adopted a water-quality standard for freshwater which states that fecal coliform bacteria are not to exceed a geometric mean of 200 colonies per 100 milliliters (200 cols/100 mL) of water, based on at least five samples collected during a 30-day period (North Carolina Department of Environment, Health, and Natural Resources, 1994a). Samples were collected less frequently during this investigation of Rhodhiss Lake; therefore, the results could not be interpreted to meet or violate the State water-quality standard. Even so, repeated exceedances of this numerical criterion generally are undesirable and indicate a potential threat to human health.

At Lower Creek site 53, fecal coliform concentrations exceeded 200 cols/100 mL in 76 percent of the samples. The two highest values occurred in conjunction with elevated streamflow, but values in excess of 200 cols/100 mL were observed across a wide range of flow conditions. Sources of the high bacterial concentrations in Lower Creek were not investigated but could include inputs of inadequately treated sewage and wastes from cattle or other animal operations in the watershed.

Concentrations of fecal coliform bacteria greater than 200 cols/100 mL also were observed at the three sites in Rhodhiss Lake—in the headwaters 37 percent of the time, and at mid-reservoir and in the forebay 16 percent of the time. Maximum concentrations at all three reservoir sites occurred in conjunction with heavy rainfall in March 1993.

Loadings of Solids and Nutrients

Loadings of total suspended solids, nitrogen, and phosphorus were calculated for Lower Creek site 53, Rhodhiss Lake site 20 (fig. 2), and for selected point-source discharges in the Rhodhiss Lake watershed. Rhodhiss Lake site 20 received inputs from Lower Creek and the Catawba River upstream from Rhodhiss Lake (fig. 2). Simultaneous measurements of streamflow and constituent concentrations collected by the USGS during the investigation were used to develop regression relations for sites 53 and 20. Continuous records of streamflow were then used to compute hourly, daily, and monthly loads. Daily loadings for site 20 were used as input to the CE-QUAL-W2 model.

Monthly loadings of all constituents to Lower Creek and the headwaters of Rhodhiss Lake peaked in March 1993 in conjunction with the highest streamflow recorded during the investigation (tables 3 and 4). It was estimated that 26 percent of the TSS load and 21 percent of the TP load to the upper end of Rhodhiss Lake for the study period occurred during one storm event during March 23-25, 1993. In contrast, this event delivered only 6 percent of the TN load to the headwaters. If only the period of January through December 1993 is considered, the same event contributed 34 percent of the 1993 annual TSS load, 27 percent of the annual TP load, and 8 percent of the annual TN load at site 20.

Table 3. Monthly loadings of total suspended solids and nutrients at Lower Creek site 53, January 1993 through March 1994

[kg, kilogram]

Month	Total suspended solids (kg)	Nitrite plus nitrete, as N (kg)	Ammonia, as N (kg)	Total nitrogen, as N (kg)	Ortho-phosphate, as P (kg)	Total phosphorus, as P (kg)
			1993			
January	2,040,000	7,880	2,220	18.800	186	1,810
February	793,000	5,350	1.350	10.900	107	730
March	7,540,000	10,900	3,050	32.500	341	6,350
April	1,620,000	7,470	2,190	17.200	170	1,450
May	643,000	5,460	1,270	10,600	108	630
June	241,000	3,990	598	6.630	112	426
July	326,000	3,810	541	6,560	135	550
August	173,000	2,910	351	5.080	147	463
September	140,000	2,340	296	4,460	154	440
October	84,000	1,720	198	3,660	176	428
November	273,000	2,420	349	5,020	162	554
December	565,000	3,960	624	7,400	143	740
			1994			
January	1,910,000	6,230	1,480	14,600	183	1,790
February	2,250,000	6,340	1,590	15,300	164	1,990
March	4,230,000	7,900	1,950	21,300	237	3,680
Total	22,828,000	78,680	18,057	180,010	2,525	22,031

Table 4. Monthly loadings of total suspended solids and nutrients at Rhodhiss Lake site 20, January 1993 through March 1994

[kg, kilogram]

Month	Total suspended solids (kg)	Nitrite plus nitrate, as N (kg)	Ammonia, as N (kg)	Total nitrogen, ss N (kg)	Ortho-phosphate, as P (kg)	Total phosphorus, as P (kg)
			1993			
January	3,170,000	57,500	16,400	109,000	2.020	7,770
February	2,540,000	43,300	11,600	78,600	1.580	5,980
March	21,300,000	86,800	32,100	202,000	3.850	46.100
April	3,780,000	60,100	17,600	117,000	2,160	9.530
May	4,000,000	55,800	16,100	107,000	2,090	9.770
June	1,340,000	24,400	8340	50,800	1,430	4.350
July	951,000	15,700	7,220	39,900	1,390	3,890
August	891,000	14,300	7,200	37,700	1,370	3,770
September	679,000	9,480	6,730	30,400	1,280	3,330
October	524,000	6,120	6,880	25,100	1,260	3,070
November	769,000	11,300	7,050	33,100	1,290	3,440
December	1,250,000	20,500	8,310	47,900	1,470	4,470
			1994			
January	1,990,000	36,300	10,900	71,000	1,630	5,460
February	2,580,000	41,200	11,800	77,500	1,630	6,440
March	8,130,000	58,200	19,000	121,000	2,430	18,500
Total	53,894,000	541,000	187,230	1,148,000	26,880	135,870

Dissolved, inorganic forms of nitrogen—NO_2+NO_3 and NH_4—accounted for approximately 63 percent of the TN load at Rhodhiss Lake site 20, and 54 percent at Lower Creek site 53. Loadings of PO_4 accounted for 20 percent of the TP loads in the headwaters of the reservoir and 11 percent in Lower Creek (tables 3 and 4).

Seven permitted point sources, including one industry, four municipal wastewater-treatment plants, and two drinking-water treatment plants, each discharged, on average, a total of 0.904 m^3/s into the Rhodhiss Lake watershed. Although numerous additional point sources were present, these seven accounted for more than 99 percent of the permitted wastewater flow in the Rhodhiss Lake watershed downstream from the Lake James dam (fig. 1). Five discharges were located upstream from site 20—the Marion wastewater-treatment plant (WWTP), Lenoir WWTP, Great Lakes Carbon, the Morganton WWTP, and the Morganton water-treatment plant. Two facilities discharged directly into Rhodhiss Lake—the Valdese WWTP, which discharged 2 km upstream from site 24, and the Lenoir-Rhodhiss water-treatment plant, which discharged 3.5 km downstream from site 24 (fig. 2).

These seven facilities monitored the volume of discharge and TSS concentrations in their effluents. In addition, the four municipal wastewater-treatment plants monitored concentrations of TN, NH_4, and TP. The USGS obtained these data from the State Compliance Monitoring System (Sandra Gillaspie, North Carolina Department of Environment, Health, and Natural Resources, written commun., 1995) and used them to estimate total loads from each facility for the study period (table 5). These estimated loads should be considered approximate, because monthly or quarterly data were used for the computations.

Comparison of results in tables 4 and 5 indicated that almost all of the suspended solids and the majority of the nitrogen and phosphorus entering the headwaters of the reservoir originated from nonpoint sources. Nonetheless, point sources contributed significant amounts of nitrogen and phosphorus. For example, although the Lenoir-Lower Creek WWTP accounted for less than 1 percent of the TSS load measured at Lower Creek site 53, it accounted for 34 and 25 percent of the TN and TP loads, respectively. Likewise, the five point sources upstream from Rhodhiss Lake site 20 accounted for less than 1 percent of the TSS load, but 27 percent of the TN load and 22 percent of the TP load. Because it is unlikely that 100 percent of the loads from the facilities upstream from Rhodhiss Lake actually reached the reservoir because of biochemical transformations and losses, these estimated point-source contributions to Rhodhiss Lake are somewhat liberal. In addition to these five point sources, the city of Valdese WWTP added approximately 80,900 kilograms (kg) of nitrogen and 30,500 kg of phosphorus directly to the mid-section of Rhodhiss Lake during the study (table 5).

Table 5. Estimated total loadings of total suspended solids and nutrients from selected point-source discharges in the Rhodhiss Lake watershed for the period January 1993 through March 1994

[Facilities with daily mean discharges of at least 0.002 m^3/s (1 m^3/s = 22.83 Mgal/d) were included. kg, kilogram; WWTP, wastewater treatment plant; --, no data]

Permitted discharging facility	Average discharge (m^3/s)	Total suspended solids (kg)	Ammonia, as N (kg)	Total nitrogen, as N (kg)	Total phosphorus, as P (kg)
Marion-Corpening Creek WWTP	0.109	8,000	649	42,000	3,280
Great Lakes Carbon	.163	26,100	--	--	--
Lenoir-Lower Creek WWTP	.097	64,600	20,400	60,600	5,550
Morganton WWTP	.286	388,000	131,000	204,000	21,100
Morganton Water Plant	.029	6,000	--	--	--
Valdese WWTP	.216	353,000	6,500	80,900	30,500
Lenoir-Rhodhiss Water Plant	.004	1,100	--	--	--
Total	0.904	846,800	387,500	158,549	60,430

SIMULATION OF HYDRODYNAMICS, CONSTITUENT TRANSPORT, AND WATER QUALITY

A two-dimensional, laterally averaged hydrodynamic and water-quality model was constructed for Rhodhiss Lake. The model was calibrated and tested using data collected during April 1993-March 1994. The model has the capability to simulate water level, currents, heat transport and temperature distribution, nonconservative material transport, and the transport and transformation of 11 chemical constituents. Transport during stratified and unstratified conditions, the effects of changing wind and air temperature on reservoir response, and the effects of changes in orthophosphate loadings on dissolved oxygen and algal concentrations in Rhodhiss Lake were simulated by using the calibrated model.

Modeling Approach

The modeling approach described in this section includes information on the hydrodynamic and water-quality model used to simulate flow, transport, and water-quality conditions in Rhodhiss Lake. The theoretical foundations of the model are briefly summarized. Implementation of the model or configuration of the model for the Rhodhiss Lake application is documented. Implementation included development of the computational grid, establishment of boundary and initial conditions, and preliminary selection of model parameters.

Model Description

Objectives of the reservoir modeling were to develop the capability to simulate (1) movement and dispersion of spills or releases in the reservoir and (2) reservoir water quality (nutrient, algal, and DO concentrations) in response to possible changes in external loadings and flows. Reservoir water-quality degradation often is first evident in coves. Hence, the capability for simulation of hydrodynamic and water-quality processes in coves and embayments, including exchange with the mainstem, was needed. Rhodhiss Lake is relatively narrow, with a maximum width of less than 600 m. Temperature and DO data (fig. 18) suggest that a laterally averaged model formulation is appropriate for Rhodhiss Lake. These functional requirements for the model were considered when selecting the model CE-QUAL-W2 for application to Rhodhiss Lake.

The model CE-QUAL-W2 has been under continuous development since 1975 (Edinger and Buchak, 1975). Complete details on model theory and structure, and an extensive bibliography for theoretical development and application are given by Cole and Buchak (1995).

Finite-difference forms of the complete unsteady, laterally averaged equations of conservation of mass, conservation of momentum, and transport (one equation for each constituent modeled) are solved by the model. Heat transport and the momentum equations are coupled through the density gradient terms—the water movement affects the heat distribution through advection and diffusion, and the heat distribution affects water movement through the presence of longitudinal and vertical density gradients. Likewise, heat transport and water quality are dynamically coupled because of the effects of suspended solids on light penetration and heat adsorption.

An efficient and accurate numerical scheme (Leonard, 1979) is used to solve the system of partial differential equations. The appropriate maximum computational time interval to ensure numerical stability is automatically computed by the model for each time step in the simulation, so that the time step is variable throughout the simulation. The time step can be controlled somewhat, by setting a maximum allowable value for the simulation.

The reservoir is divided into a series of longitudinal segments, each of which may have a unique length (fig. 19). Branches also can be included in the model so that complex reservoir geometries, including coves and embayments, can be realistically represented by the computational grid (fig. 19). Each segment is further subdivided into layers. All layers within a segment must have the same length, but each layer can have a unique width and thickness (fig. 19). Conditions within each computational cell are assumed to be uniform, so that the governing equations represent conditions throughout the cell. The cell-averaged longitudinal velocity, vertical velocity, density, temperature, and constituent concentrations are determined for each cell through the numerical solution of the governing equations; water-surface elevation is determined for the surface layer in each segment.

Multiple inflows (tributaries and point sources) and outflows (withdrawals) can be included in the model. A selective withdrawal algorithm computes the withdrawal zone for an outlet structure at the dam by using information on outflows, outlet geometry, and water density.

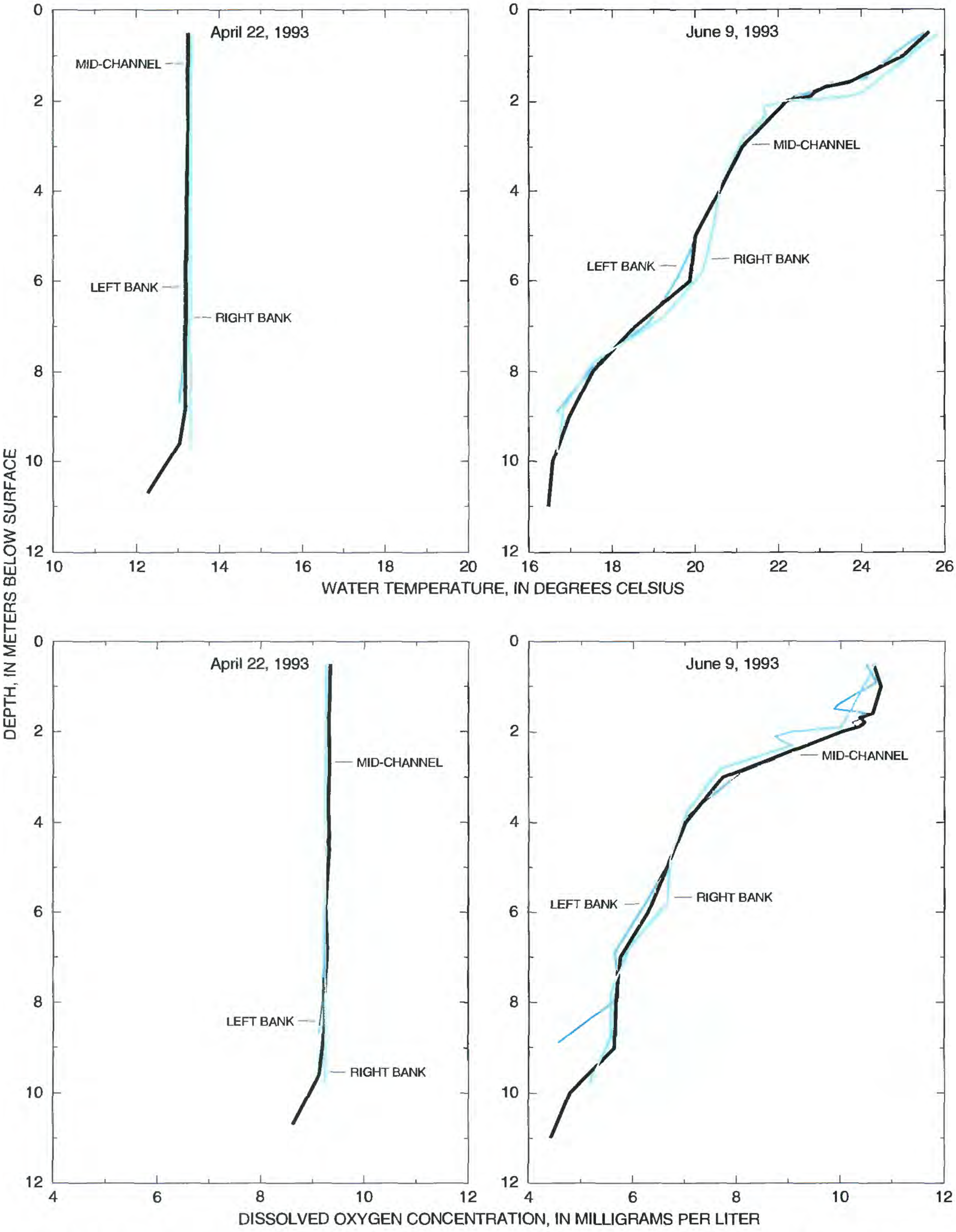

Figure 18. Vertical profiles of water temperature and dissolved oxygen concentration at Rhodhiss Lake site 24, near left bank, mid-channel, and near right bank, for stratified and unstratified conditions.

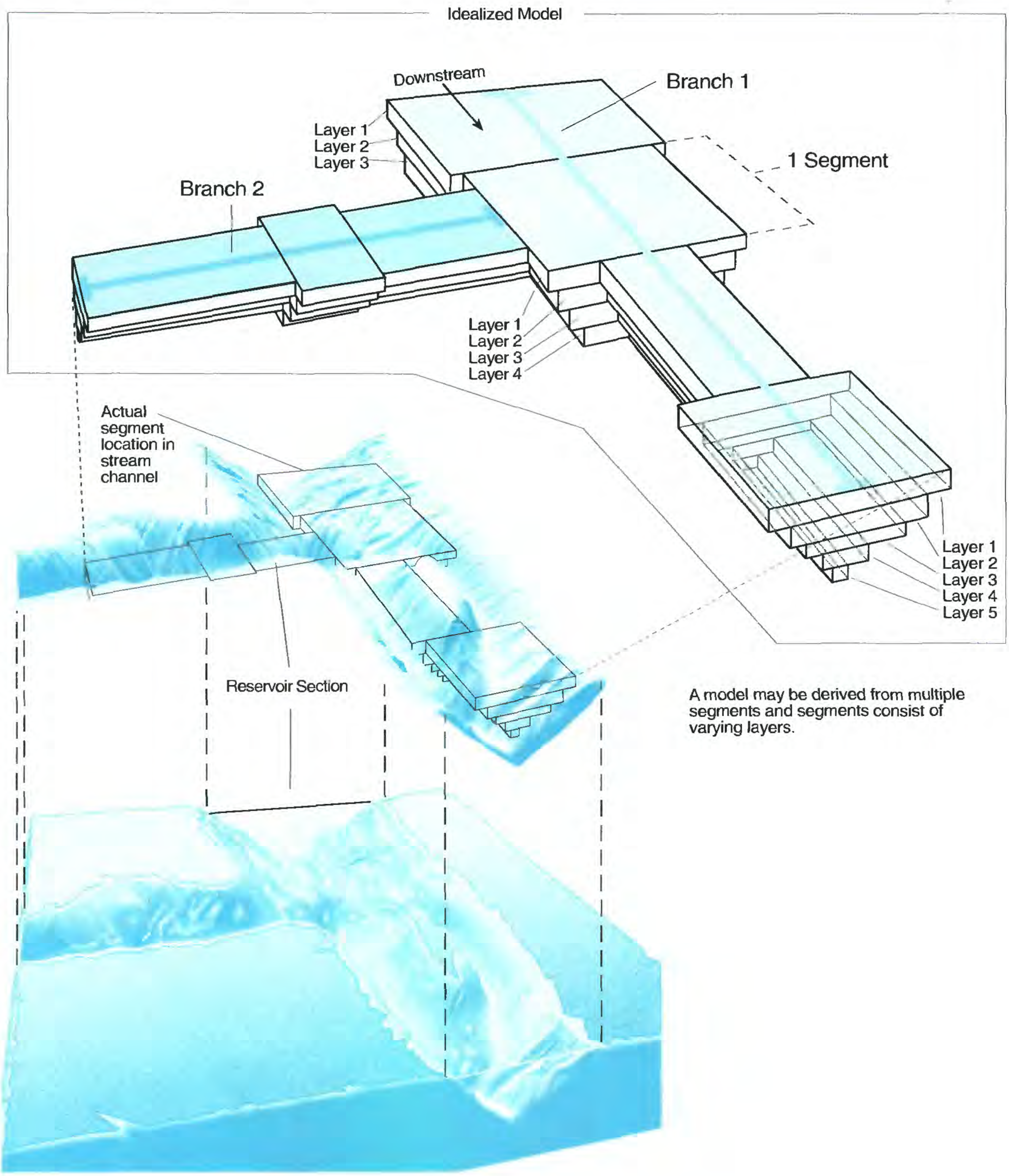

Figure 19. Idealized model segments, layers, and branches for Rhodhiss Lake.

As many as 21 separate constituents can be included in the water-quality simulations (table 6); temperature is always included because of the dependence of the flows on temperature gradients. Of the remaining 20 constituents, all or only selected constituents can be included, but the inter-dependence of the various processes must be considered when selecting constituents for simulation. Because water-quality conditions generally vary much more slowly than the hydrodynamics, water-quality calculations can be performed less frequently than hydrodynamic calculations during simulations.

Although CE-QUAL-W2 is quite rigorous in the treatment of hydrodynamics, heat transport, and chemical kinetics, the model does have limitations. These limitations need to be considered when interpreting model results. The reservoir is assumed to be well-mixed laterally at all points. Consequently, inflows from tributaries entering the reservoir along one shoreline are assumed to be instantaneously mixed across the reservoir at the point of entry to the reservoir. The model may not reproduce flow patterns in regions where vertical accelerations are large, such as in the immediate vicinity of an outlet structure at a dam. The model includes only one algal component, so algal succession cannot be simulated. The model also does not simulate the effects of zooplankton and macrophytes on water-quality conditions. These limitations on algal succession, zooplankton, and macrophytes should be of little consequence for Rhodhiss Lake simulations. Finally, chemical processes at the sediment-water interface are modeled rather simplistically, using either a zero- or first-order approximation. These processes may be particularly important, however, when evaluating the long-term effects of changes in external (from point sources, tributaries, and runoff) nutrient loadings on reservoir water quality. Processes at the sediment-water interface control internal loadings of nutrients to the reservoir. These internal loadings may be a significant source of nutrients to the reservoir, but the loadings from bottom sediments can change as a function of the external loading rate. The model does not simulate the effects of changes in external loadings on internal nutrient loads.

Model Implementation

Implementation of CE-QUAL-W2 for Rhodhiss Lake included (1) development of the computational grid, (2) specification of boundary and initial conditions, and (3) preliminary selection of model parameter values. This section of the report describes the model domain and computational grid. Procedures and associated assumptions for specification of boundary and initial conditions are given, and values of model parameters are identified.

Computational Grid

The model domain extends from Huffman Bridge (site 20, fig. 2) to Rhodhiss Dam, or a distance of 18.5 km. The domain encompasses five embayments—Stafford Creek, McGalliard Creek, Hoyle Creek, Freemason Creek, and Nolden Creek (fig. 20). There are 37 computational segments along the mainstem of the reservoir and each embayment is represented by 3 segments. Volumes associated with smaller embayments not explicitly included in the model, such as the cove directly to the west of Rhodhiss Dam, were added to nearby mainstem volumes so that the total volume of the reservoir was preserved in the computational grid. The mathematical formulation of the model suggests that length of a particular segment should generally exceed the maximum reservoir width in that segment. All segments were 500 m long, which was about the minimum allowable length; segment widths for some of the upper layers downstream from Freemason Creek exceeded l500 m, but most computational cell widths were less than 500 m. Each layer was 1 m thick. Distances from the elevation of the spillway crest to the bottom of the channel ranged from about 3 m near Huffman Bridge to slightly more than 16 m at the dam; thus, the number of active computational layers ranged from 3 to 16. The orientation of the longitudinal axis of each segment relative to north was determined for each segment. This information is used in the computation of surface wind stress in each segment.

Reservoir bathymetry was determined from surveys made by the USGS on June 9, 1994, when 38 cross sections were measured along the mainstem of the reservoir using a recording fathometer. Cross sections were measured at 0.5-km longitudinal intervals, and measured sections coincided approximately with mainstem computational segment boundaries (fig. 20). Additional cross sections were measured in the coves and embayments around the reservoir. The water surface was 1 m below the spillway crest when the cross sections were measured. The measured cross sections were extrapolated to the top of the spillway so that the entire reservoir volume could be included in the computational grid. Widths used in the computational grid were determined from these measured cross sections.

Table 6. Water-quality constituents included in the Rhodhiss Lake model

[BOD, biological oxygen demand]

Constituent	Constituents in Rhodhiss model	Description
Conservative tracer	Yes	Neutrally buoyant, nonreactive material used for tracking water movement and determining water age.
Inorganic suspended solids	No	Affects light penetration and nutrient concentrations through adsorption and settling.
Coliform bacteria	No	Indicator of pathogen contamination; modeled as a first-order temperature-dependent decay.
Total dissolved solids	No	Affects ionic strength and, thus, pH and carbonate species; Rhodhiss Lake has very low dissolved solids concentrations.
Labile dissolved organic matter	Yes	Part of biological oxygen demand; consists of relatively fast decaying algal excretion and algae.
Refractory dissolved organic matter	Yes	Part of biological oxygen demand; slowly decaying compounds produced from decay of labile dissolved organic matter.
Algae	Yes	All phytoplankton are represented by one algal compartment; algae produce and use dissolved oxygen, utilize inorganic carbon, nitrate, ammonium, and phosphorus, and contribute to labile and refractory dissolved organic matter.
Detritus	Yes	Particulate organic material; decaying detritus exerts an oxygen demand in the water column and on the bed, and is a source of nitrogen, phosphorus, and carbon.
Phosphorus	Yes	Assumed to be completely available as orthophosphate for use by algae; released by sediments under anoxic conditions; released by algal respiration and decay of organic matter.
Ammonium	Yes	Used by algae in photosynthesis; converted to nitrate under oxic conditions; released by sediments under anoxic conditions.
Nitrate-nitrite	Yes	Converted from ammonium (nitrification) under oxic conditions; lost from system by denitrification under anoxic conditions; used by algae.
Dissolved oxygen	Yes	Both aerobic and anaerobic processes are simulated; dissolved oxygen lost by (1) decay of detritus and dissolved organic matter (2) sediment oxygen demand, (3) nitrification processes and (4) algal respiration; input by reaeration and algal photosynthesis.
Organic bottom sediments	Yes	Settling algae and detritus accumulate as organic bottom sediments; sediments decay, producing inorganic carbon. ammonium. and phosphorus, and use dissolved oxygen.
Total inorganic carbon	No	Used by algae in photosynthesis; produced by decay of organic materials; affects pH through carbon dioxide concentration.
Alkalinity	No	Indicates buffering capacity and resistance to pH changes from acidic or alkaline loadings; treated as conservative constituent.
pH	No	Computed from water temperature, total dissolved solids concentration, alkalinity, and total inorganic carbon concentration.
Carbon dioxide	No	Used in pH computation.
Bicarbonate	No	Used in pH computation.
Carbonate	No	Used in pH computation.
Iron	Yes	Affects nutrient concentrations through adsorption and settling; released from anoxic sediments.
Carbonaceous biological oxygen demand	Yes	Used in evaluation of effects on dissolved oxygen of loadings expressed in terms of BOD.

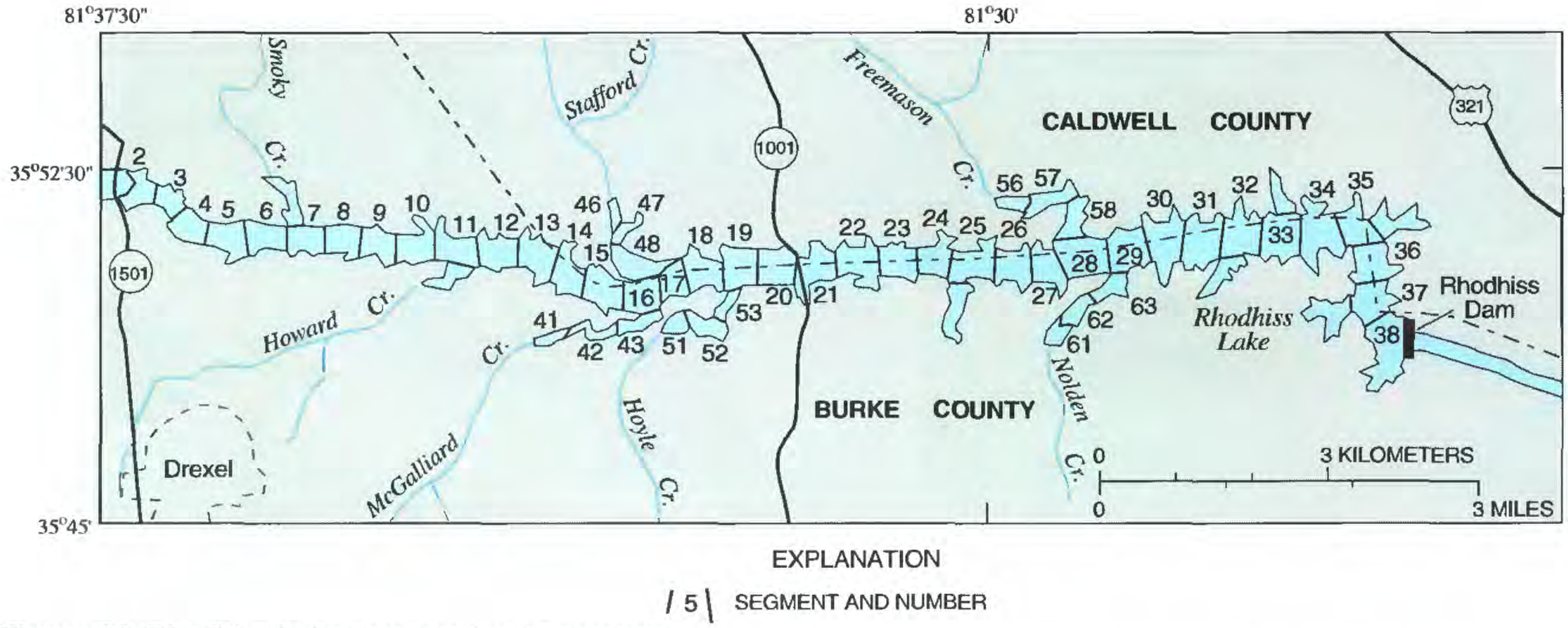

Figure 20. Rhodhiss Lake computational segments.

An elevation-volume table (table 7) was developed for Rhodhiss Lake from the computational grid, which is an approximation of the actual volume. The full-pool volume of Rhodhiss Lake was estimated to be 59.2 million m^3. This volume is significantly less than the previously published values of 83 million m^3 (Jayne, 1994) and 89.6 million m^3 (U.S. Environmental Protection Agency, 1975).

Table 7. Volume-elevation relation for Rhodhiss Lake computational grid

Distance below spillway crest (meters)	Elevation (meters above sea level)	Volume (million cubic meters)
0	303.4	59.206
1	302.4	51.063
2	301.4	43.829
3	300.4	37.181
4	299.4	31.022
5	298.4	25.499
6	297.4	20.583
7	296.4	16.272
8	295.4	12.620
9	294.4	9.415
10	293.4	6.689
11	292.4	4.256
12	291.4	2.422
13	290.4	1.099
14	289.4	.403
15	288.4	.155

In 1996, Duke Power Company personnel used a recording fathometer coupled with a global positioning system to record more than 1 million discrete depths in Rhodhiss Lake (Tim Leonard, Duke Power Company, oral commun., June 6, 1996). A geographic information system with an elevation contouring package was used to develop a detailed bathymetric map of the reservoir. Using these data, the full-pond volume of the reservoir was computed as 57.429 million m^3. This volume, determined by using detailed data and sophisticated data interpretation techniques, generally agrees with the full-pond volume used in the Rhodhiss Lake model.

Boundary and Initial Conditions

Boundaries of the Rhodhiss Lake model include the reservoir bottom, the shoreline and tributary streams, the upstream boundary at Huffman Bridge, the downstream boundary at Rhodhiss Dam, and the water surface. Both hydraulic, which includes temperature, and chemical boundary conditions are required.

Hydraulic Boundary Conditions

The reservoir bottom is assumed to be an impermeable boundary so that there is no discharge of ground water to the reservoir within the model domain or loss of water from the reservoir to the ground-water system. The magnitude of ground-water recharge or discharge from Rhodhiss Lake is unknown, but is probably small relative to the flow in the reservoir. The reservoir bottom also is assumed to be immobile so that sediments are not resuspended by the flow. The reservoir bottom causes resistance to the flow, thereby extracting energy from the mean flow. The Chezy formulation is used in CE-QUAL-W2 to describe this phenomenon, which varies with the magnitude of the flow. A single resistance coefficient, which is an empirical value and cannot be measured directly, is applied to all of the bottom computational cells (table 8).

Table 8. Hydraulic and thermal parameters specified as model input

[$m^{0.5}/s$, meter to the half power per second; watts/$m^2/^oC$, watts per square meter per degree Celsius; m^2/s, square meter per second]

Parameter	Purpose	Value in Rhodhiss model	Constant or time variable
Chezy resistance coefficient	Represents turbulent exchange of energy at reservoir bottom	70 $m^{0.5}/s$	Constant
Coefficient of sediment-water heat exchange	Computes heat exchange between reservoir bottom and overlying water	8×10^{-7} watts/$m^2/^oC$	Constant
Wind sheltering coefficient	Reduces measured wind speed to effective wind speed at water surface	0.7 - 0.9 (dimensionless)	Time variable
Longitudinal eddy viscosity	Represents laterally averaged longitudinal turbulent transport of momentum	1 m^2/s	Constant
Longitudinal eddy diffusivity	Represents laterally averaged longitudinal turbulent transport of mass and heat	1 m^2/s	Constant

Heat exchange between the reservoir bottom and the overlying water is computed from (1) the reservoir bottom temperature, (2) the simulated water temperature, and (3) a coefficient of sediment-water heat exchange (table 8). The reservoir bottom temperature and the coefficient of heat exchange are assumed to be constant in space and time. A reasonable estimate of the reservoir bottom temperature is the annual average bottom-water temperature; a value of 10 °C was used in this application. Heat exchange at the reservoir bottom is typically quite small—about two orders of magnitude less than surface heat exchange.

The reservoir shoreline is defined as a boundary across which there is no flow. The exact position of the shoreline changes during a model simulation because of the changing water level.

The drainage area of Rhodhiss Lake increases less than 10 percent (from 2,650 to 2,830 km^2) between Huffman Bridge and Rhodhiss Dam. For the purposes of the model, all of the tributary stream inflows from this 180-km^2 area into Rhodhiss Lake were assumed to occur in the five branches representing Stafford, McGalliard, Hoyle, Freemason, and Nolden Creeks (fig. 20). These tributaries are spatially distributed along the length of the reservoir and generally represent the largest tributary inflows to the reservoir.

The 180-km^2 contributing drainage area between Huffman Bridge and Rhodhiss Dam was apportioned among the five branches (branches 2-6, fig. 20) based on the measured drainage areas of the streams. Inflows from the branches were estimated by using measured daily mean flows at Lower Creek (site 53) in cubic meters per second per square kilometer of drainage area, multiplied by the estimated drainage of the individual branch. Inflows from the tributary streams were generally quite small, and there was little justification for estimating inflows at a time interval of less than 1 day. The mean tributary inflow (total for all five branches) for the period April 1, 1993, through March 31, 1994, was estimated to be 2.54 m^3/s. The median total tributary inflow for this period was 1.9 m^3/s, and the total tributary inflow exceeded 5.5 m^3/s 5 percent of the time.

Estimated hourly inflows were used as the upstream hydraulic boundary condition at Huffman Bridge. Measured hourly flows at the Catawba River at Calvin (drainage area of 1,322 km^2) and Johns River at Arney's Store (drainage area of 523 km^2) were used to determine flows at Huffman Bridge. The ungaged drainage area between these two gages and Huffman Bridge is 805 km^2. Flow at Huffman Bridge was estimated as the sum of (1) measured flow at the Catawba River at Calvin, (2) measured flow at Johns River at Arney's Store, and (3) ungaged flow (estimated as measured Johns River flow per unit drainage area multiplied by the ungaged area). Measured flows in the Catawba River at Calvin were not used to estimate downstream ungaged flows because of the regulation effects of Bridgewater Dam on flows at Calvin. These estimated flows were subsequently adjusted during model calibration, as described in the following section. Adjusted hourly flows at Huffman Bridge during the period April 1, 1993, through March 31, 1994, ranged from 9.31 to 448 m^3/s, with a median flow of 39.2 m^3/s (fig. 21).

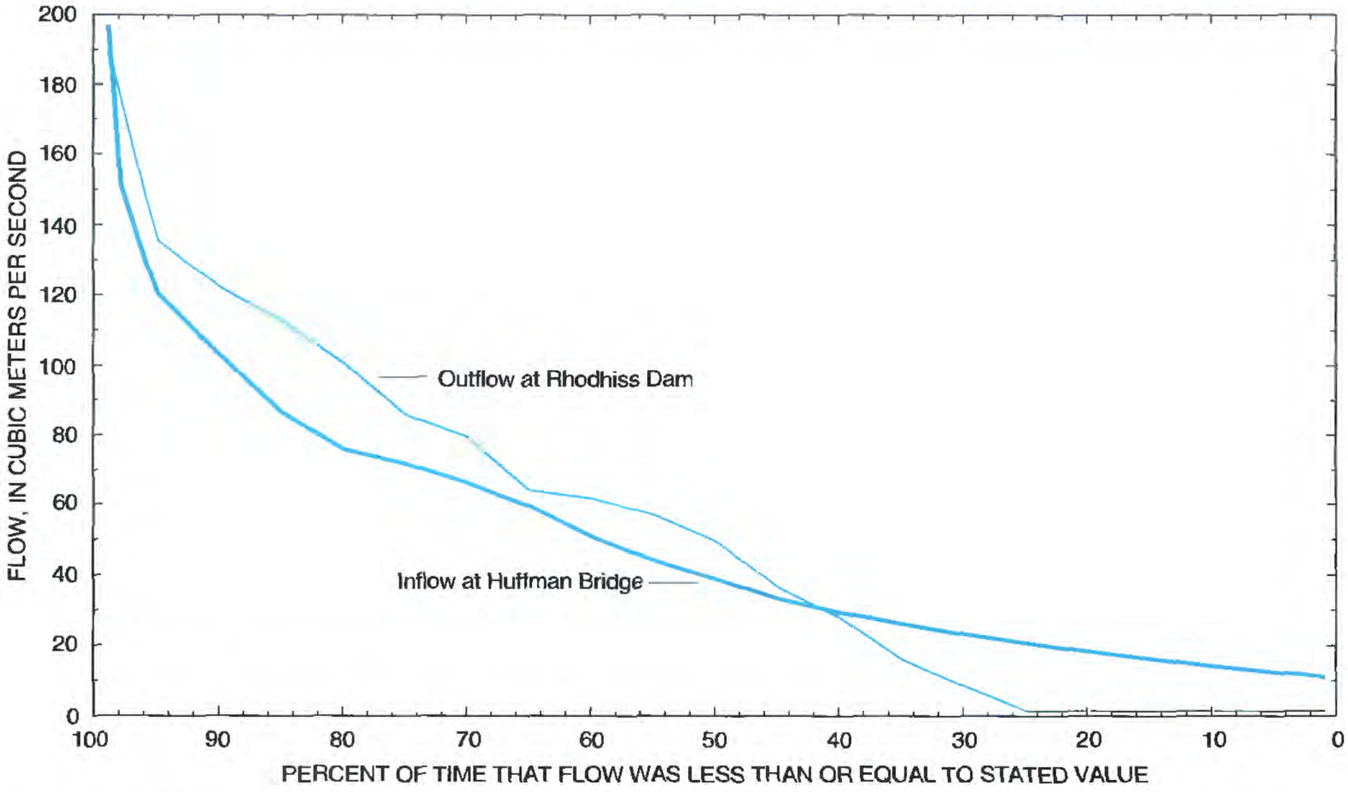

Figure 21. Flow durations for the Catawba River at Huffman Bridge and at Rhodhiss Dam, April 1, 1993, through March 31, 1994.

Water temperature measured at Huffman Bridge at hourly intervals (fig. 22) was used as the upstream thermal boundary condition. Daily mean water temperature measured at Huffman Bridge was used as the required upstream temperature boundary condition for each of the 5 tributary branches.

Outflows from Rhodhiss Dam were obtained from Duke Power Company records of hourly power generation, and a relation between power generation and flow. During the period April 1, 1993, through March 31, 1994, hourly flows from Rhodhiss Dam ranged from 1.7 to 243 m^3/s, with a median flow of 50 m^3/s. Hourly flows were greater than 101 m^3/s at least 20 percent of the time, and were less than or equal to 10 m^3/s 30 percent of the time (fig. 21). Discharge measurements for two flow rates were made downstream from Rhodhiss Dam on April 21, 1994. Measured flows were within 5 percent of flows determined from the relation between power generation and flow.

The vertical extent of and the flow distribution in the withdrawal zone (the region near the dam directly affected by flow through the turbines) near Rhodhiss Dam were simulated by using information on the intake structure (which directs flow from the reservoir to the turbines), the outflow rate, and the simulated density gradient upstream in the reservoir. The withdrawal structure was assumed to be narrow (less than 10 percent of the total width of dam) relative to the total width of the dam. The bottom of the withdrawal structure is at an elevation of 290.9 m above sea level, and the top of the structure is at 301.8 m mean sea level (msl) (Jon Knight, Duke Power Company, written commun., September 1994). The top of the spillway is at an elevation of 303.4 m msl. The intake structures at Rhodhiss Dam are located adjacent to the north bank of the reservoir, which likely results in a lateral asymmetry in the withdrawal zone. These lateral variations in flow conditions are not simulated by the model because of the laterally averaged assumption.

Boundary conditions at the water surface include (1) wind stress and (2) surface heat exchange. All meteorological data required for these computations were measured at the Hickory Airport, located about 5 km southeast of Rhodhiss Lake, where data generally were recorded at hourly intervals.

Wind stress is computed from a time series of wind speed and direction, the orientation of the computational segment, and a wind-sheltering coefficient (table 8). The temporally variable wind-sheltering coefficient reduces the effects of wind on the reservoir because of topographic or vegetation sheltering of the water surface.

Surface heat exchanges are computed from reservoir latitude and longitude, and from a time series of measured air temperature, dewpoint temperature, cloud cover, and wind speed and direction. Simulated surface-water temperature also is required for the computation of surface heat exchange. Surface heat exchange can be simulated from the heat exchange equation as the sum of seven separate terms, or by using a more simple linearization of the heat exchange

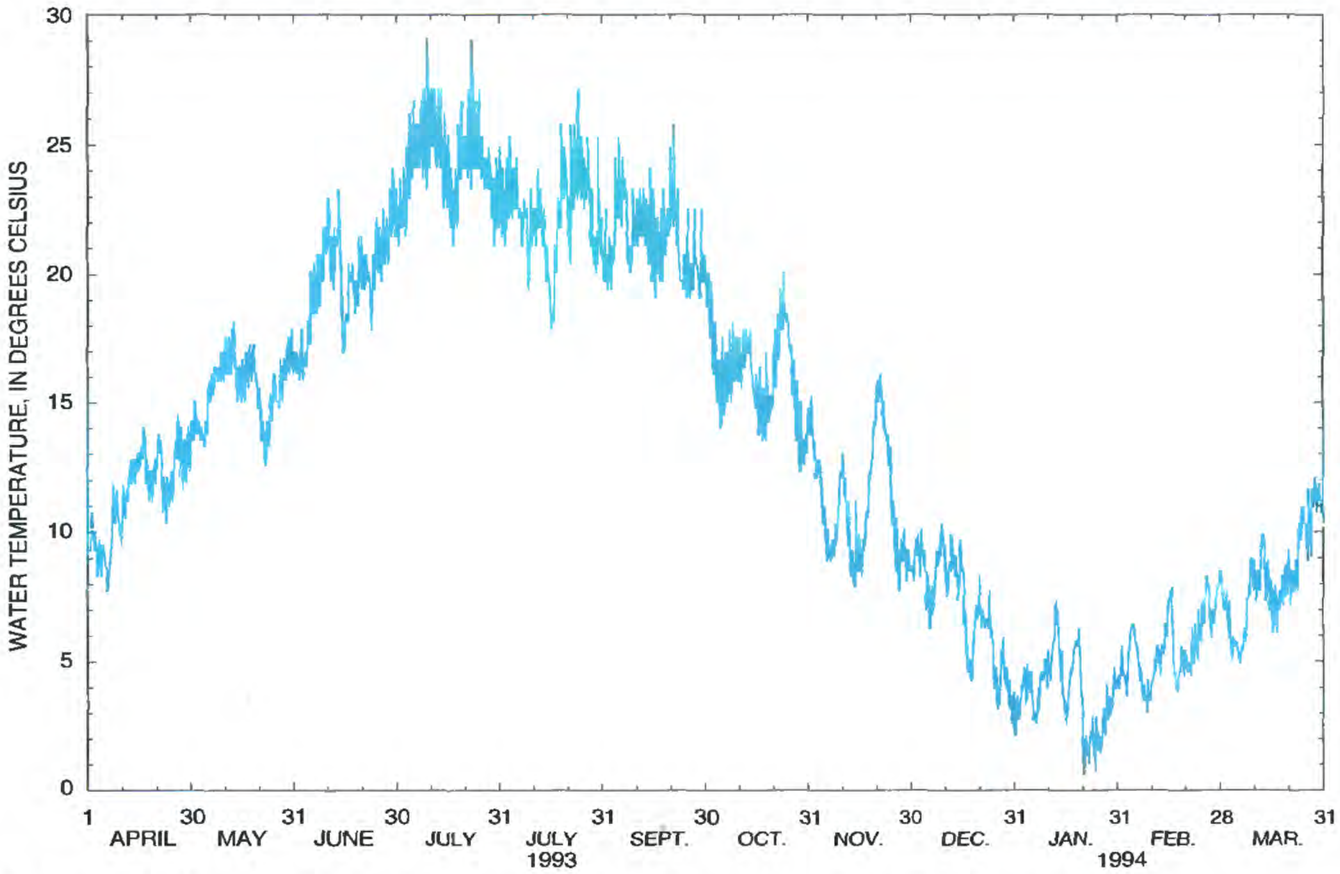

Figure 22. Measured water temperature at Huffman Bridge, April 1, 1993, through March 31, 1994.

equation and an estimated equilibrium temperature (the temperature at which incoming radiation heat rates are balanced by outgoing water-surface temperature-dependent processes). The full heat exchange equation was used in this application.

Other hydraulic boundary conditions included water-supply withdrawals and point-source inputs. Water-supply withdrawals were located in model segments 14 (Valdese intake), 28 (Lenoir intake), and 35 (Granite Falls intake). Withdrawal rates were assumed to be constant throughout the simulation period at 0.28 m^3/s (Valdese), 0.33 m^3/s (Lenoir), and 0.06 m^3/s (Granite Falls), respectively. These withdrawal rates were based on reported 1993 intakes (Terry Gross, North Carolina Department of Environment, Health, and Natural Resources, Mooresville, oral commun., August 1994). The only significant permitted discharge directly into Rhodhiss Lake is from the Valdese wastewater-treatment facility. The outlet is located in model segment 17. The reported 1993 average flow of 0.21 m^3/s (Sandra Gillaspie, North Carolina Department of Environment, Health, and Natural Resources, oral commun., August 1994) was used for the entire simulation period. The temperature of the discharge was assumed to vary monthly and be equal to the monthly mean air temperature. The discharge point was located about 9 m below the water surface, or about 3 m above the bottom of the reservoir.

The difference between precipitation inputs to and evaporative losses directly from the reservoir was assumed to be small relative to other inflows and outflows. On an annual basis, this difference is about 0.1 m^3/s, or 0.2 percent of the April 1993 - March 1994 median outflow rate. Consequently, time series of precipitation to and evaporation from the reservoir were not included as model boundary conditions. The loss of heat from the reservoir resulting from the evaporation process was, however, included in the heat balance.

Chemical Boundary Conditions

A time series of concentrations of selected constituents at all inflow boundaries is required for model operation. Boundary conditions need not be supplied for all constituents which are included in model simulations.

Chemical boundary conditions at Huffman Bridge were based on estimated daily loadings, as described earlier in this report. Boundary data at Huffman Bridge and the tributaries included algae, detritus (particulate organic matter), PO_4, nitrate, DO, iron, and carbonaceous biological oxygen demand (CBOD). Concentrations of total suspended solids, TN, ammonia, TP, and DO were available from the Valdese wastewater-treatment plant at monthly intervals. Because no orthophosphate data from the Valdese plant were available, PO_4 concentrations in the discharge were initially assumed to be zero.

Exchange of carbon dioxide and DO occurs at the air-water interface. Exchange rates are governed by wind speed, water temperature, reservoir elevation above sea level, and the molecular diffusivity of the gas. Because initial results indicated that simulated reaeraton rates were too small, the CE-QUAL-W2 surface-exchange algorithm was modified for this application to increase reaeration. Atmospheric inputs of all simulated constituents to the reservoir were assumed to be negligible. Constituent inputs from the bottom boundary generally were computed within the model, and were based on the value of selected parameters (table 9) and the concentration of the constituent in the overlying waters.

Initial Conditions

Initial water level and velocity, temperature, and constituent concentrations for each computational cell are required prior to initiating a simulation. Initial water level was set to the measured value at Rhodhiss Dam for the day simulations were to begin. Velocities were assumed to be zero. Initial water temperature was assumed to be uniform throughout the reservoir, and equal to the measured water temperature at Huffman Bridge for the day simulations were to begin. Initial constituent concentrations also were assumed to be uniform throughout the reservoir and equal to measured values at Huffman Bridge.

The time required for the effects of the estimated initial conditions to disappear from the model domain, (model spin-up time) was generally 3-10 days. Spin-up time generally is less if initial model conditions closely represent actual reservoir conditions.

Model Parameters

Parameters are used to describe physical and chemical processes which are not explicitly modeled and also to provide chemical kinetic rate information. Many parameters cannot be measured directly and are often adjusted during the model calibration process until simulated results agree with observations.

Most of the relevant hydrodynamic and thermal processes are modeled in CE-QUAL-W2 so there are relatively few adjustable hydraulic and thermal model parameters (table 8). The resistance, bottom heat exchange, and wind-sheltering coefficients were previously discussed. The longitudinal eddy viscosity describes horizontal turbulent exchange of momentum, and the eddy diffusivity describes horizontal turbulent mixing of mass and heat. (Formally, both of these coefficients include the effects of mathematical terms resulting from the lateral integration of the governing three-dimensional equations.) Simulation results for Rhodhiss Lake were relatively insensitive to all of these parameters, with the exception of the wind-sheltering coefficient.

There are 57 chemical kinetic rate coefficients required for application of CE-QUAL-W2 (table 9). Most of the coefficients were selected based on published values, and all of the coefficients are temporally and spatially invariant.

Other Model Options

The maximum computational time step was limited to 1 hour so that the computational time step would not exceed the interval at which boundary data were supplied to the model; the model-selected computational interval generally was about 5 minutes. The "QUICKEST" numerical scheme (Leonard, 1979) was used for solving the transport equations. A Crank-Nicholson scheme (Roache, 1982) was used to solve the vertical advection equation.

Model calculations occur at time steps smaller than the time increments at which boundary data are provided. Boundary data can be assumed to vary linearly between measured values or remain constant between measured values. For this application, all boundary data were assumed to vary linearly between measured values, with the exception of releases from Rhodhiss Dam, which remained constant between measured values.

Model Calibration And Testing

Model calibration was achieved through adjustment of model parameters and, in some cases, estimated input data, for the period April 1, 1993, through March 30, 1994, when data collection ended. Although data collection began in January 1993, upstream boundary water-temperature data at Huffman Bridge (site 20) were not available until March 11, 1993. Moreover, high flows during late March resulted in flows over the spillway; on March 24, the water level at Rhodhiss Dam reached a maximum elevation of 304.36 m, or more than one meter above the crest of the spillway. Downstream boundary flow data were not available during the time when water was flowing over the spillway.

Table 9. Chemical kinetic rate coefficients specified as model input

[m, meter; $m^3/m/g$, cubic meter per meter per gram; *, dimensionless parameter; m/d, meter per day; d, day; watts/m^2, watts per square meter; °C, degrees Celsius; m^2/d, square meter per day; BOD, biological oxygen demand; g/m, gram per meter; mg/L, milligram per liter]

Parameter	Computational purpose	Value in Rhodhiss model
Light extinction coefficient for water	Amount of solar radiation absorbed in the surface layer	0.5/m
Light extinction coefficient for organic solids	Amount of solar radiation absorbed in the surface layer	0.2 $m^3/m/g$
Fraction of incident solar radiation absorbed at water surface	Amount of solar radiation absorbed in the surface layer	0.3*
Suspended solids settling rate	Settling rates and sediment accumulation on reservoir bottom	2 m/d
Algal growth rate	Maximum gross algal production rate, uncorrected for respiration, mortality, excretion, or settling; temperature dependent	1.9/d
Algal mortality rate	Maximum algal mortality rate; temperature dependent	0.09/d
Algal excretion rate	Maximum algal photorespiration rate, which becomes labile dissolved organic matter	0.005/d
Algal dark respiration rate	Maximum algal dark respiration rate	0.005/d
Algal settling rate	Representative settling velocity for algal assemblages	0.1 m/d
Saturation light intensity	Saturation light intensity at maximum algal photosynthesis rate	150 watts/m^2
Fraction of algal biomass lost by mortality to detritus	Detritus and dissolved organic matter concentrations; remaining biomass becomes labile dissolved organic matter	0.8*
Lower temperature for algal growth	Algal growth rate as a function of water temperature	10 °C
Fraction of algal growth at lower temperature	Algal growth rate as a function of water temperature	0.1*
Lower temperature for maximum algal growth	Algal growth rate as a function of water temperature	22 °C
Fraction of maximum growth at lower temperature	Algal growth rate as a function of water temperature	0.99*
Upper temperature for maximum algal growth	Algal growth rate as a function of water temperature	22.5 °C
Fraction of maximum growth at upper temperature	Algal growth rate as a function of water temperature	0.95*
Upper temperature for algal growth	Algal growth rate as a function of water temperature	35 °C
Fraction of algal growth at upper temperature	Algal growth rate as a function of water temperature	0.1*
Labile dissolved organic matter decay rate	Dissolved oxygen loss and production of inorganic carbon, ammonium, and phosphate from algal decay; temperature dependent	0.04/d
Labile to refractory decay rate	Transfer of labile to refractory dissolved organic matter	0.005/d
Maximum refractory dissolved organic matter decay rate	Dissolved oxygen loss and production of inorganic carbon, ammonium, and phosphate from decay of refractory dissolved organic matter; temperature dependent	0.001/d
Detritus decay rate	Dissolved oxygen loss and production of inorganic carbon, ammonium, and phosphate from decay particulate organic matter; temperature dependent	0.002/d
Detritus settling velocity	Loss of particulate organic matter to bottom sediment	2.5 m/d
Lower temperature for organic matter decay	Organic matter decay as a function of temperature	5 °C
Fraction of organic matter decay at lower temperature	Organic matter decay as a function of temperature	0.05*
Lower temperature for maximum organic matter decay	Organic matter decay as a function of temperature	25 °C
Fraction of maximum organic matter decay at lower temperature	Organic matter decay as a function of temperature	0.95*

Table 9. Chemical kinetic rate coefficients specified as model input--Continued

[m, meter; $m^3/m/g$, cubic meter per meter per gram; *, dimensionless parameter; m/d, meter per day; d, day; watts/m^2, watts per square meter; °C, degrees Celsius; m^2/d, square meter per day; BOD, biological oxygen demand; g/m, gram per meter; mg/L, milligram per liter]

Parameter	Computational purpose	Value in Rhodhiss model
Sediment decay rate	Decay rate of organic matter in bed sediments	0.015/d
Sediment oxygen demand	Zero-order sediment oxygen demand for each computational segment	0 g/m^2/d
5-day biological oxygen demand decay rate	Effects of BOD loading on dissolved oxygen	0.15/d
BOD temperature rate coefficient	Adjusts 5-day BOD decay rate at 20 °C to ambient temperature	1.0147*
Ratio of 5-day BOD to ultimate BOD	Effects of BOD loading on dissolved oxygen	1.20*
Release rate of phosphorus from bottom sediments	Phosphorus balance; computed as a fraction of the sediment oxygen demand	0.005*
Phosphorus partitioning coefficient	Describes sorption of phosphorus on to suspended solids	3.0
Algal half-saturation constant for phosphorus	The phosphorus concentration at which the uptake rate is one-half the maximum uptake rate; upper concentration at which algal growth is proportional to phosphorus concentration	0.005 g/m
Release rate of ammonia from bottom sediments	Nitrogen balance; computed as a fraction of the sediment-oxygen demand	0.003*
Ammonia decay rate	Rate at which ammonia is oxidized to nitrate	0.2/d
Algal half-saturation constant for ammonia	Nitrogen concentration at which the algal uptake rate is one half the maximum uptake rate	0.014*
Lower temperature for ammonia decay	Ammonia nitrification as a function of temperature	5 °C
Fraction of nitrification at lower temperature	Ammonia nitrification as a function of temperature	0.1*
Lower temperature for maximum ammonia decay	Ammonia nitrification as a function of temperature	25 °C
Fraction of maximum nitrification at lower temperature	Ammonia nitrification as a function of temperature	0.99*
Nitrate decay rate	Rate at which nitrate is denitrified; temperature dependent	0.15/d
Lower temperature for nitrate decay	Denitrification as a function of temperature	5.0 °C
Fraction of denitrification at lower temperature	Denitrification as a function of temperature	0.1*
Lower temperature for maximum nitrate decay	Denitrification as a function of temperature	25 °C
Fraction of maximum denitrification at lower temperature	Denitrification as a function of temperature	0.99*
Iron release rate from bottom sediments	Iron balance; computed as fraction of sediment oxygen demand	1.0*
Iron settling velocity	Particulate iron settling velocity under oxic conditions	2.0 m/d
Oxygen stoichiometric equivalent for ammonia decay	Relates oxygen consumption to ammonia decay	4.0*
Oxygen stoichiometric equivalent for organic matter decay	Relates oxygen consumption to decay of organic matter	1.5*
Oxygen stoichiometric equivalent for dark respiration	Relates oxygen consumption to algae dark respiration	0.9*
Oxygen stoichiometric equivalent for algal growth	Relates oxygen production to algal growth	3.0*
Stoichiometric equivalent between organic matter and phosphorus	Relates phosphorus release to decay of organic matter	0.009*
Stoichiometric equivalent between organic matter and nitrogen	Relates nitrogen release to decay of organic matter	0.08*
Dissolved oxygen limit	Dissolved oxygen concentration below which anaerobic processes, such as nitrification and sediment nutrient releases occur	0.1 mg/L

Hydrodynamics and Heat Transport

Initial simulation results indicated somewhat poor agreement between measured and simulated water levels at Rhodhiss Dam. The disagreement could have been because of errors in (1) measurements of water level at the dam, (2) estimates of outflow at the dam, (3) representation of reservoir bathymetry by the computational grid, or (4) estimates of inflow. Elevations relative to sea level were checked by leveling at the water-level gage, and the water-level measurements were considered reliable. Two flow measurements were made in the river downstream from the dam. The differences between measured flows in the river and the release rates recorded by the dam operators were less than 5 percent. There is likely some error in the representation of reservoir bathymetry in the model, but there was no indication of a consistent bias in differences between measured and simulated water levels in the initial results. This, along with the close agreement between the USGS and Duke Power Company estimates of reservoir volume, suggests that the computational grid adequately represents the reservoir bathymetry. As previously described, inflows from about one-third of the Rhodhiss Lake drainage area were estimated. Consequently, it is most likely that poor estimates of the inflow (both volume and timing) boundary condition were the primary source of the initial disagreement between measured and simulated water levels.

Original estimates of inflow, therefore, were adjusted to achieve agreement between measured and simulated water levels. Simulated water levels were quite sensitive to inflow rates so that only relatively small adjustments in the original estimates generally were required. Ninety-nine percent of the adjustments were less than about 20 percent of the original estimated inflow.

After these adjustments, there was good agreement between measured and simulated water levels at Rhodhiss Dam. The root mean square difference between measured and simulated water levels was 0.085 m, the maximum positive (water level under-predicted) difference was 0.15 m on November 26, 1993, and the maximum negative difference was 0.38 m during the high flow event on May 4, 1993 (fig. 23). Eighty percent of the differences between measured and simulated water levels were between 0.02 and 0.12 m. The total range in measured water level during the simulation period was 1.32 m.

The heat budget is computed from information on inflow water temperature, air temperature, dewpoint temperature, cloud cover, wind speed and direction, organic solids concentration, and heat exchange between bottom sediments and the water column. Organic solids affect light penetration and consequently, heat distribution in the reservoir. Hence, water temperature calibration cannot be performed independently from water chemistry calibration. If no water chemistry computations are made or if algae and detritus are not included in the simulations, water temperature can still be simulated, but the effects of solids on heat distribution are not included.

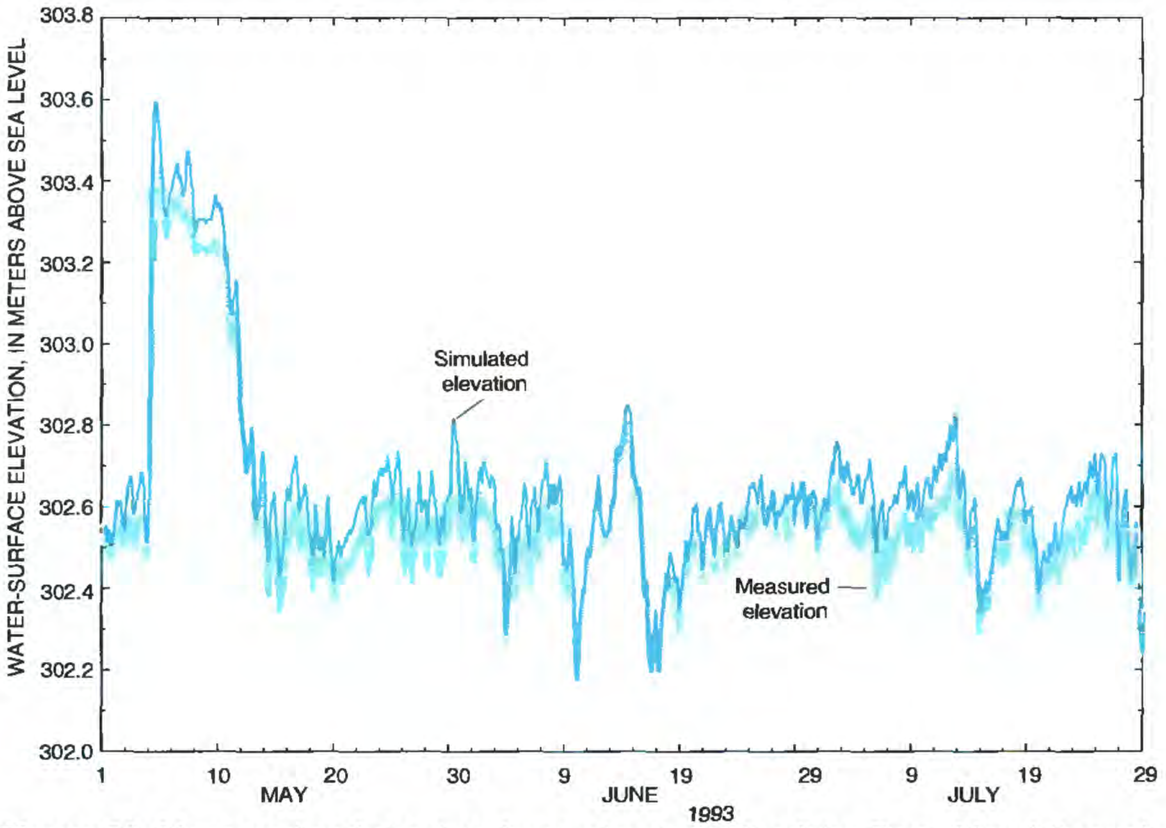

Figure 23. Measured and simulated water levels at Rhodhiss Dam, May 1 through July 29, 1993.

Simulated water temperatures reproduced the general seasonal variations in near-surface and near-bottom water temperature throughout Rhodhiss Lake (fig. 24). Simulated near-surface water temperatures generally were within 1 °C of measured values. Near-bottom water temperatures were under-predicted from mid-May through August, and larger differences between measured and simulated values occurred in the deeper waters.

The difference between near-surface and near-bottom temperature is a measure of the thermal stratification in the reservoir. Top-to-bottom temperature and the associated density differences affect the rate of transport and mixing of dissolved and suspended matter in the reservoir. Measured and simulated top-to-bottom water temperature differences were in good agreement for site 22 (fig. 25A), which is located in the upstream part of Rhodhiss Lake.

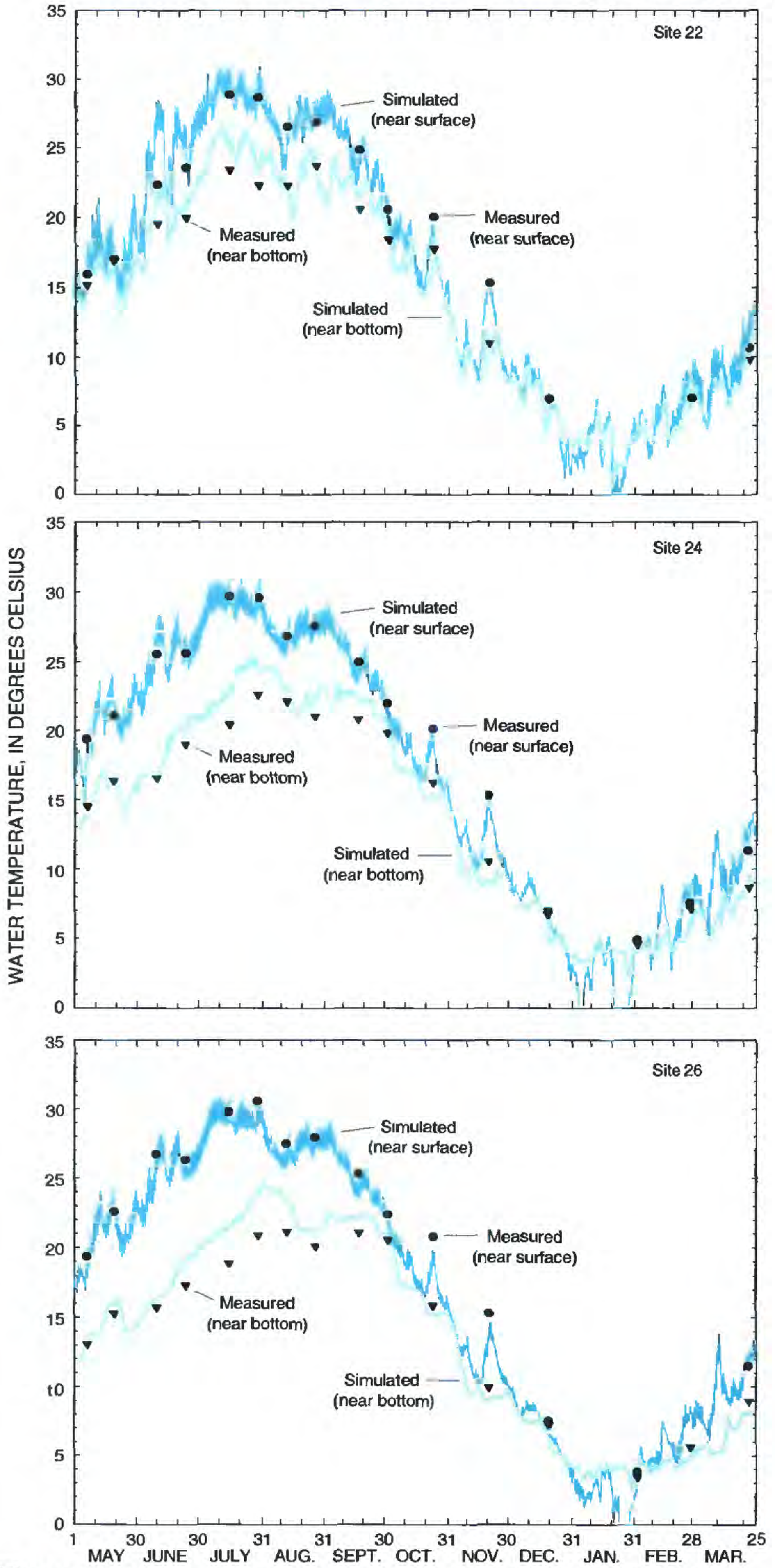

Figure 24. Measured and simulated water temperatures, May 1, 1993, through March 25, 1994, at Rhodhiss Lake sites 22, 24, and 26.

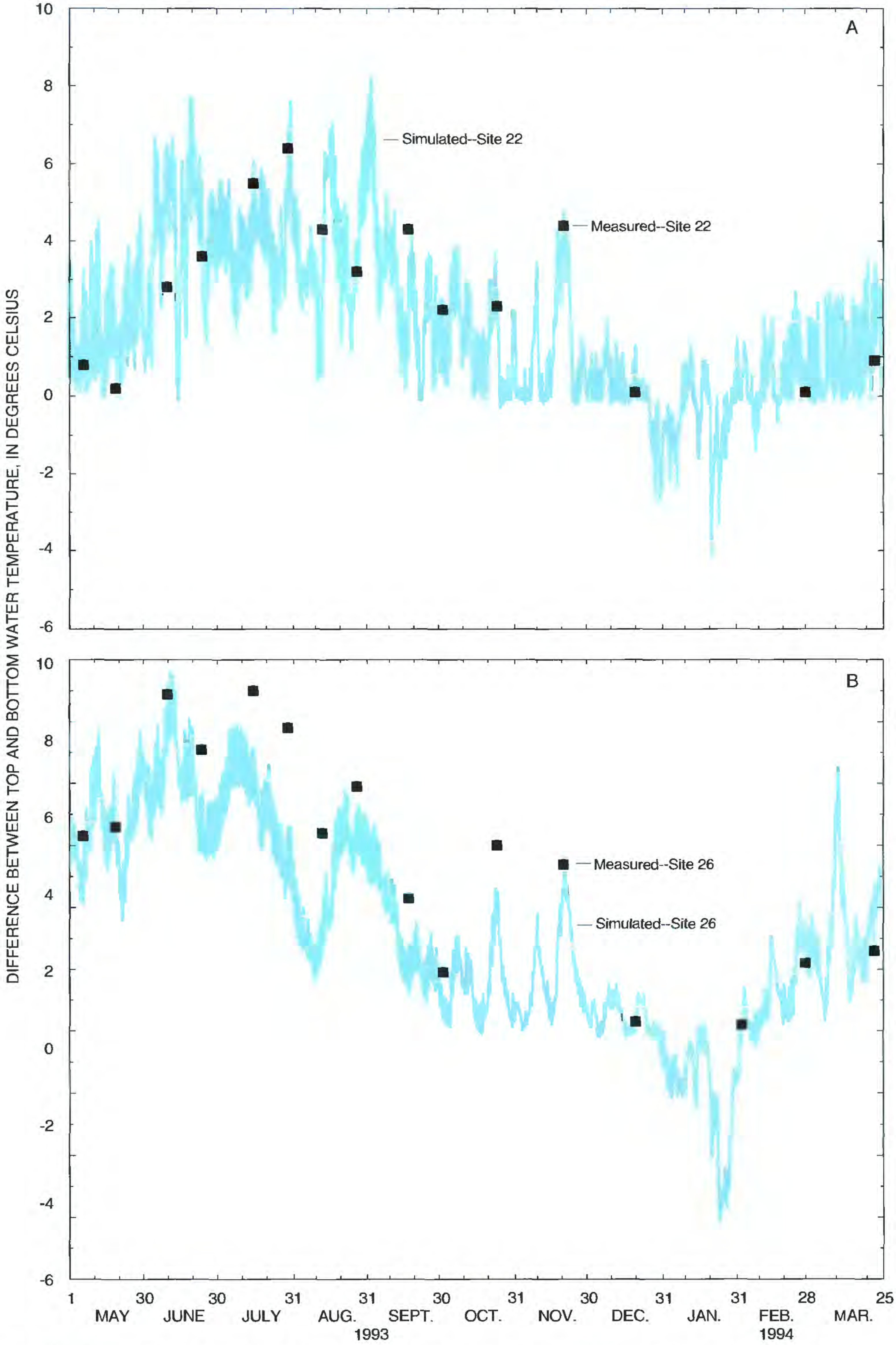

Figure 25. Measured and simulated differences between near surface and near bottom water temperatures, May 1, 1993, through March 25, 1994, at Rhodhiss Lake (A) site 22 and (B) site 26.

Measured and simulated top-to-bottom water-temperature differences at site 26 (fig. 25B), in the downstream part of the reservoir, were in general agreement, although the under-predicted near-bottom water temperatures resulted in an under-prediction of the top-to-bottom water-temperature difference.

The vertical distribution of water temperature affects vertical mixing of dissolved and suspended materials and can be used to define the general location of the epilimnion and hypolimnion of the reservoir. The epilimnion and hypolimnion typically are separated by a thermocline, in which there is a large change in temperature over a small depth. Although top-to-bottom water temperature differences can exceed 15 °C near the dam, a strong thermocline is rare in Rhodhiss Lake. Typical vertical profiles of measured and simulated water temperatures at site 24 (mid-reservoir) and site 26 (downstream segment) are shown in figure 26 A and B to reflect strongly stratified (June 9, sites 24 and 26) and weakly stratified (October 20, site 24; and September 15, site 26) conditions. For some cases (for example, June 9), both the simulated temperatures and the general shape of vertical distribution of simulated temperatures agree with measured values. In other cases, the general vertical distribution of measured water temperature is reproduced (for example August 28, site 24; fig. 26A), although some differences between measured and simulated water temperatures exist.

All of the measured water temperatures (177 observations) for the period April 1993 through March 1994 were compared with corresponding simulated values for site 24. Measured water temperature during this period ranged from about 4 to about 30 °C, although simulations indicate a greater range (fig. 24). The mean difference between simulated and measured values was -0.24 °C, 80 percent of the differences were between 1.26 and -1.80 °C, and 95 percent of the differences were between 1.7 and -2.4 °C.

For site 24, simulated water temperatures generally were too high relative to measured values when the measured water temperature exceeded 20 °C (fig. 27A); however, some of the higher water temperatures were under-predicted. Most of the simulated water temperatures under-predicted measured water temperature when the measured water temperature was less than about 8 °C, but the simulated values generally were within about 1 °C of the measured value. All of the simulated water temperatures for a particular date typically were biased in the same direction (fig. 27B). In other words, all temperatures for a given day were either over- or under-predicted. As previously suggested (fig. 24), temperatures were under-predicted during mid-May through August 1993, and generally over-predicted during the remainder of the simulation period. Simulated water temperatures were equally over-predicted as under-predicted at a particular measurement depth (fig. 27C); although simulation errors were smallest near the water surface, and the greatest range in errors was about 3 m above the reservoir bottom.

All of the results of the water temperature simulations suggest that (1) the calibrated model provides a reasonable simulation of water temperature in Rhodhiss Lake, with most of the simulated values within about 2 °C of the actual value; and (2) near-bottom water temperature, particularly in the deeper parts of the reservoir, is typically over-predicted from 1 to 3 °C during the warmer months. The simulated near-surface water temperatures generally agree more closely with corresponding measured values than simulated near-bottom water temperatures. (Accurate simulation of near-surface water temperature is important to obtain reasonable simulations of algal concentrations.) This, along with the simulated vertical profiles of water temperature (fig. 26), suggests that the model over-predicts vertical mixing, resulting in excessive transport of warmer surface waters to the cooler, deeper waters of the reservoir.

Results from the water temperature simulations provide information on physical characteristics and processes in the reservoir—information which might not be obtained from monthly or semi-monthly measurements. For example, the measurements of water temperature from mid-September to mid-November suggest that the reservoir was thermally stratified during the entire period (fig. 24). The simulations, however, indicate that the reservoir was continually mixing and stratifying during the period. A thermal inversion (near-surface water temperature less than near-bottom water temperature), which occurred in January 1994 because of extremely cold air temperatures (fig. 24), was simulated by the model, whereas no data were available to document the event. Simulation results also suggest that near-bottom temperatures in the deeper part of the reservoir vary more gradually than those in the shallower reaches (fig. 24). Finally, the reservoir appears to thermally stratify and destratify fairly quickly and often in the upstream reaches of the reservoir (fig. 25A); likewise, stratification and destratification appear to occur fairly often in the downstream reaches of the reservoir in the fall and late winter (fig. 25B).

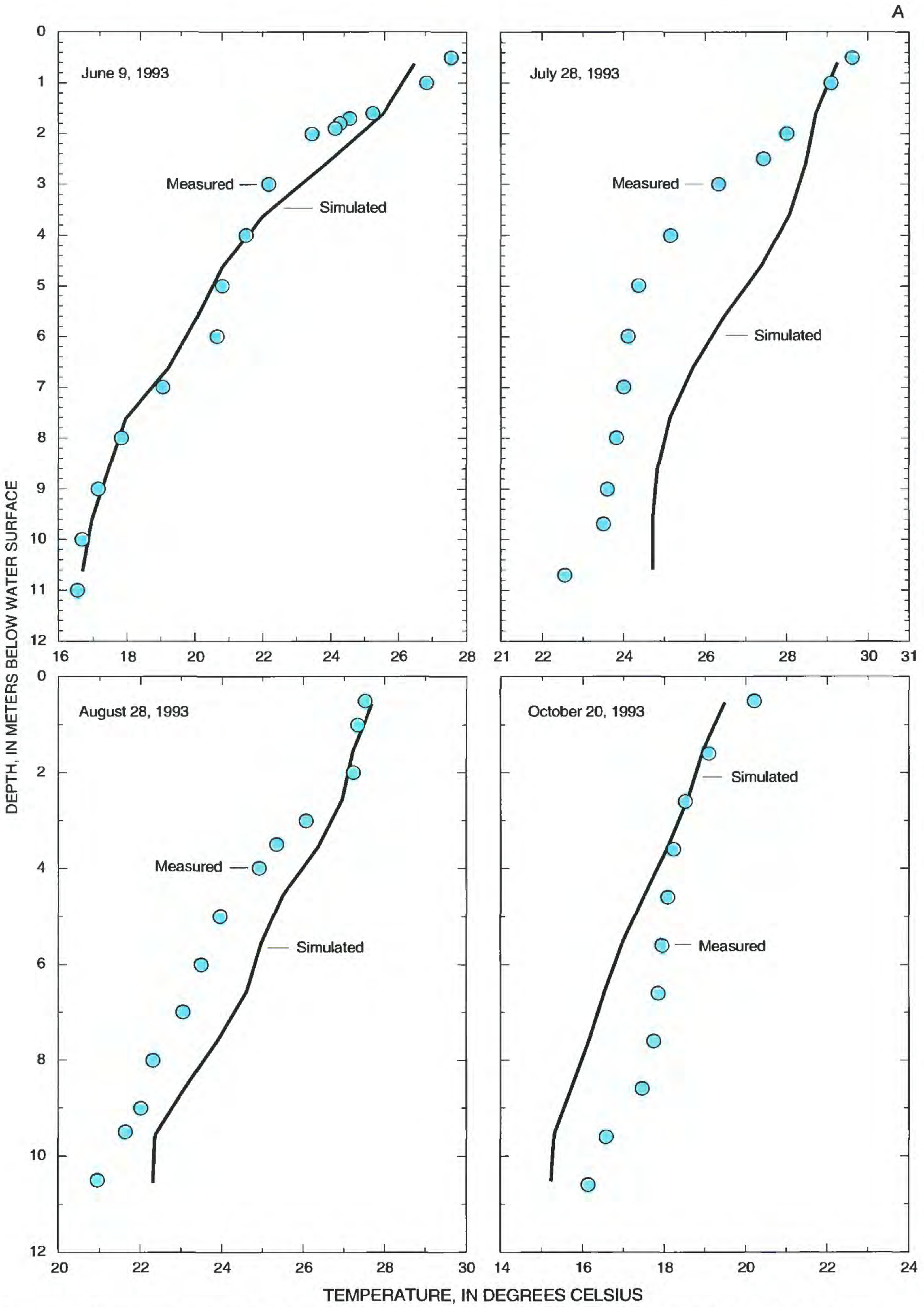

Figure 26. Measured and simulated vertical profiles of water temperature at Rhodhiss Lake for (A) site 24 on June 9, July 28, August 28, and October 20, 1993.

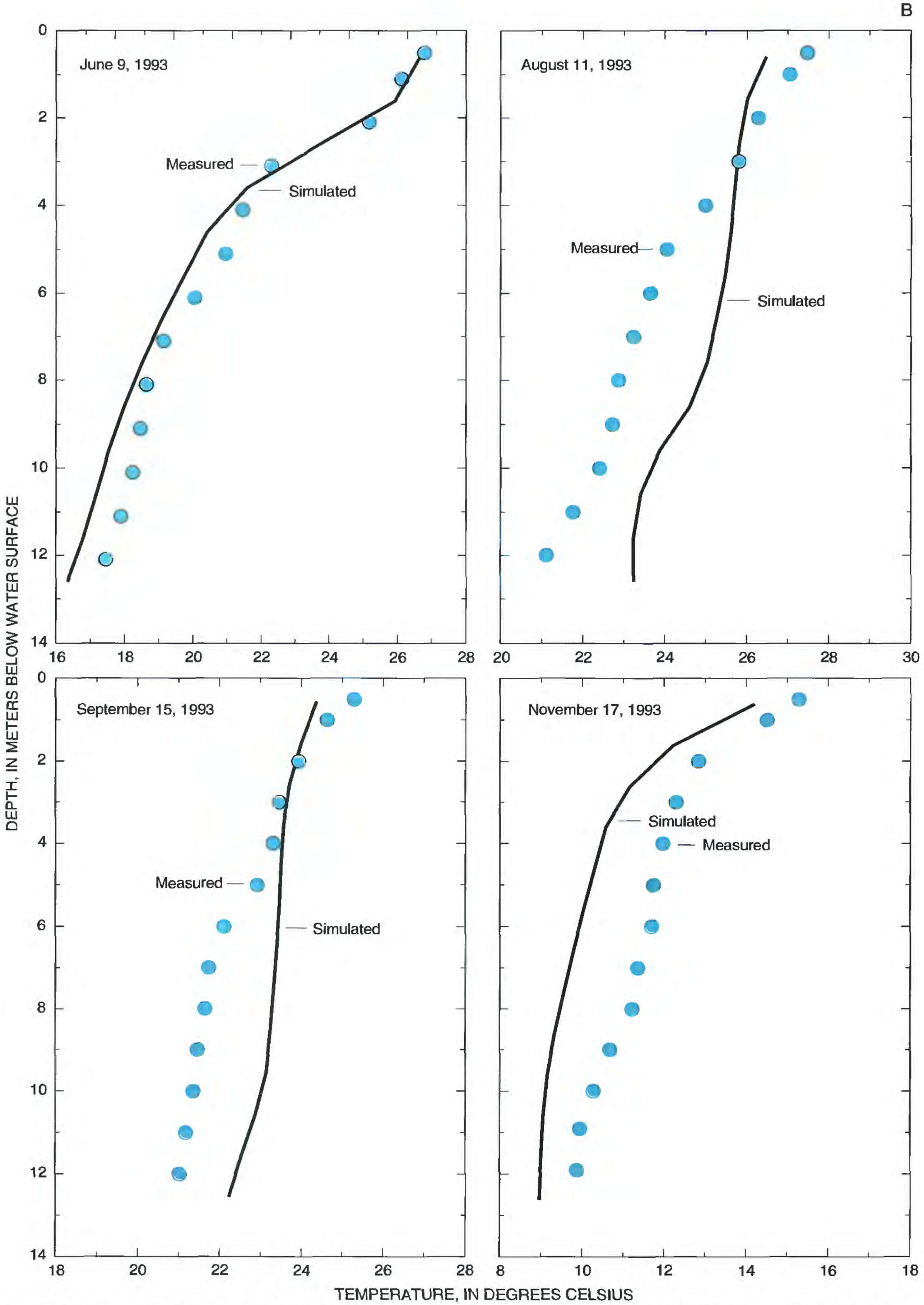

Figure 26. Measured and simulated vertical profiles of water temperature at Rhodhiss Lake for (B) site 26 on June 9, August 11, September 15, and November 17, 1993.

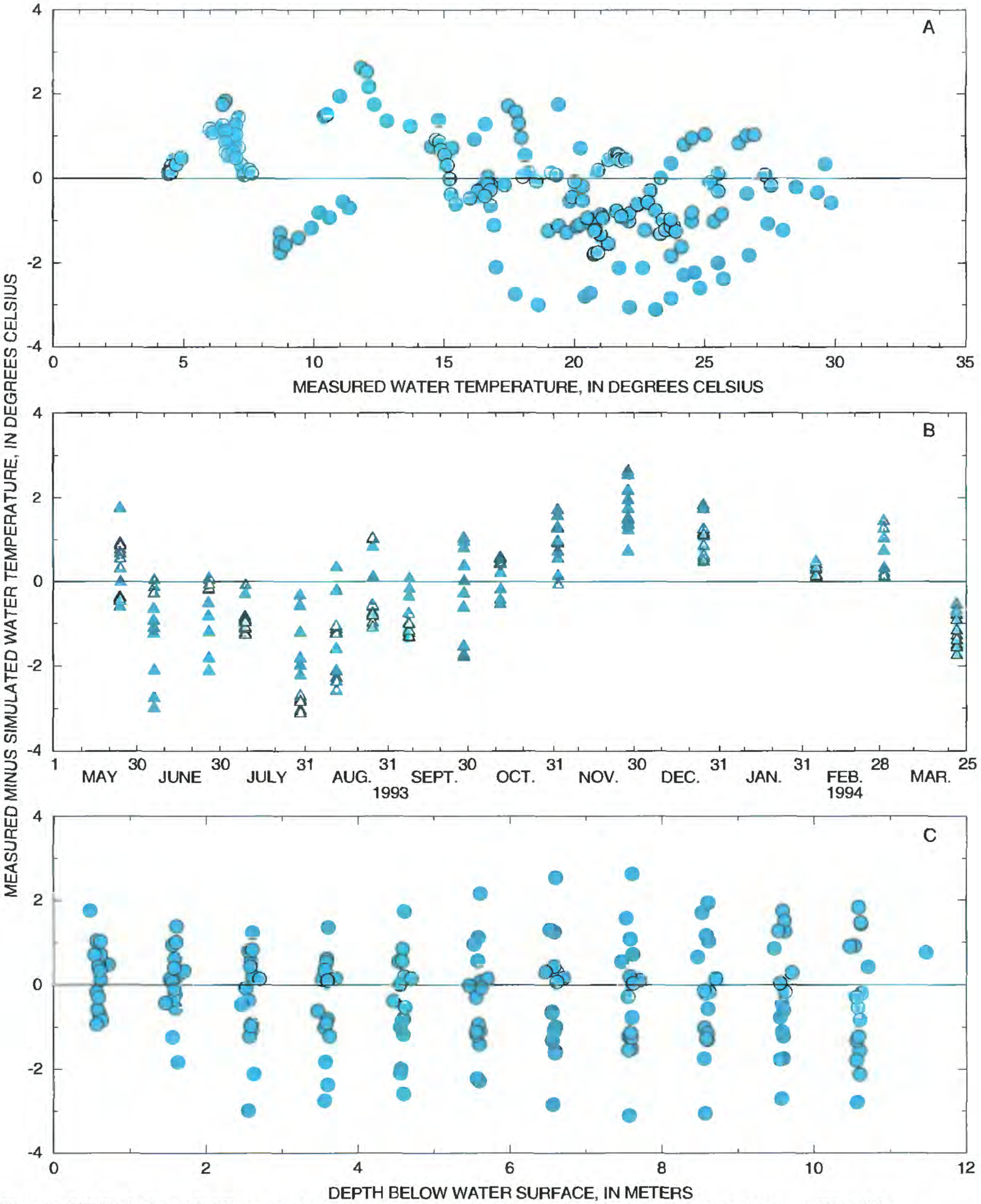

Figure 27. Relation of difference between measured and simulated water temperatures at Rhodhiss Lake site 24 to (A) measured water temperature, (B) date, and (C) water depth, May 1, 1993, through March 25, 1994.

Dissolved Oxygen

The simulated complex biochemical reactions affecting the 11 water-chemistry parameters (table 6) and physical transport processes in the Rhodhiss Lake model are aggregated in the simulated DO concentrations. Thus, results of the DO simulations are emphasized in the water-chemistry discussions in this report. The North Carolina ambient water-quality standard for DO is a daily average of greater than 5 mg/L, and an instantaneous value greater than 4 mg/L (North Carolina Department of Environment, Health, and Natural Resources, 1994a), which is another reason for focusing on DO simulation results. Nitrate and ammonia concentrations appear to have little effect on DO and algal concentrations in Rhodhiss Lake.

Simulated DO concentrations throughout the reservoir exhibited the same general patterns and magnitudes as measured values (fig. 28). The one major exception is the near-surface DO on September 29, 1993, when DO was significantly over-predicted at all sites. It is possible that there was an undetected, large influx of oxygen-demanding material sometime after September 15, when DO concentrations were high (fig. 28). Consequently, the simulations may not reflect the effects of this likely influx of oxygen-demanding material, because no information on the influx was available to be used as model boundary conditions.

Many of the differences between measured and simulated DO concentrations, particularly for near-bottom conditions (fig. 28) may be related to simulating the timing of the occurrence of increasing or decreasing DO. A good example of the shift in timing between measured and simulated DO concentrations is on October 20, at site 25 (fig. 28). The difference between the measured and simulated DO value for the time at which the measurement was made is about 3.5 mg/L. However, another way to view the difference between the measured and simulated values is that the measured low DO occurred about 2 days before the simulated DO of like magnitude. Similarly, the occurrence of a decrease in near-bottom DO concentration in early June at site 25 was simulated to have occurred at least two days before the actual decrease (fig. 28). Factors affecting simulation of the timing of low DO events include temperature-dependent rate multipliers for organic-matter decay and vertical mixing. Hence, accurate simulation of hydraulic and thermal conditions is required to adequately simulate DO dynamics.

Some examples of simulated vertical distributions of DO concentration and associated measured values are presented in figure 29, as examples of varying degrees of agreement between measured and simulated DO. The distributions shown in figure 29 do not account for any time shifting associated with the simulations. Consequently, simulated vertical distributions of DO at some time slightly before or after the actual time of the field measurements may exhibit better agreement with measured values than simulated distributions corresponding to the exact time of the measurements. The agreement between measured and simulated vertical distributions of DO concentrations (fig. 29) was not as good as for vertical distributions of water temperature (fig. 26). In general, simulations of vertical distributions of DO were better when simulated and measured vertical distributions of water temperature were in good agreement (for example, June 9 at sites 24 and 26; figs 26 and 29). The poorest agreement between measured and simulated DO concentrations generally was at mid-depths, and the more complex vertical distributions (for example, August 11 and September 15 at site 26; fig. 29B) were more difficult to simulate accurately.

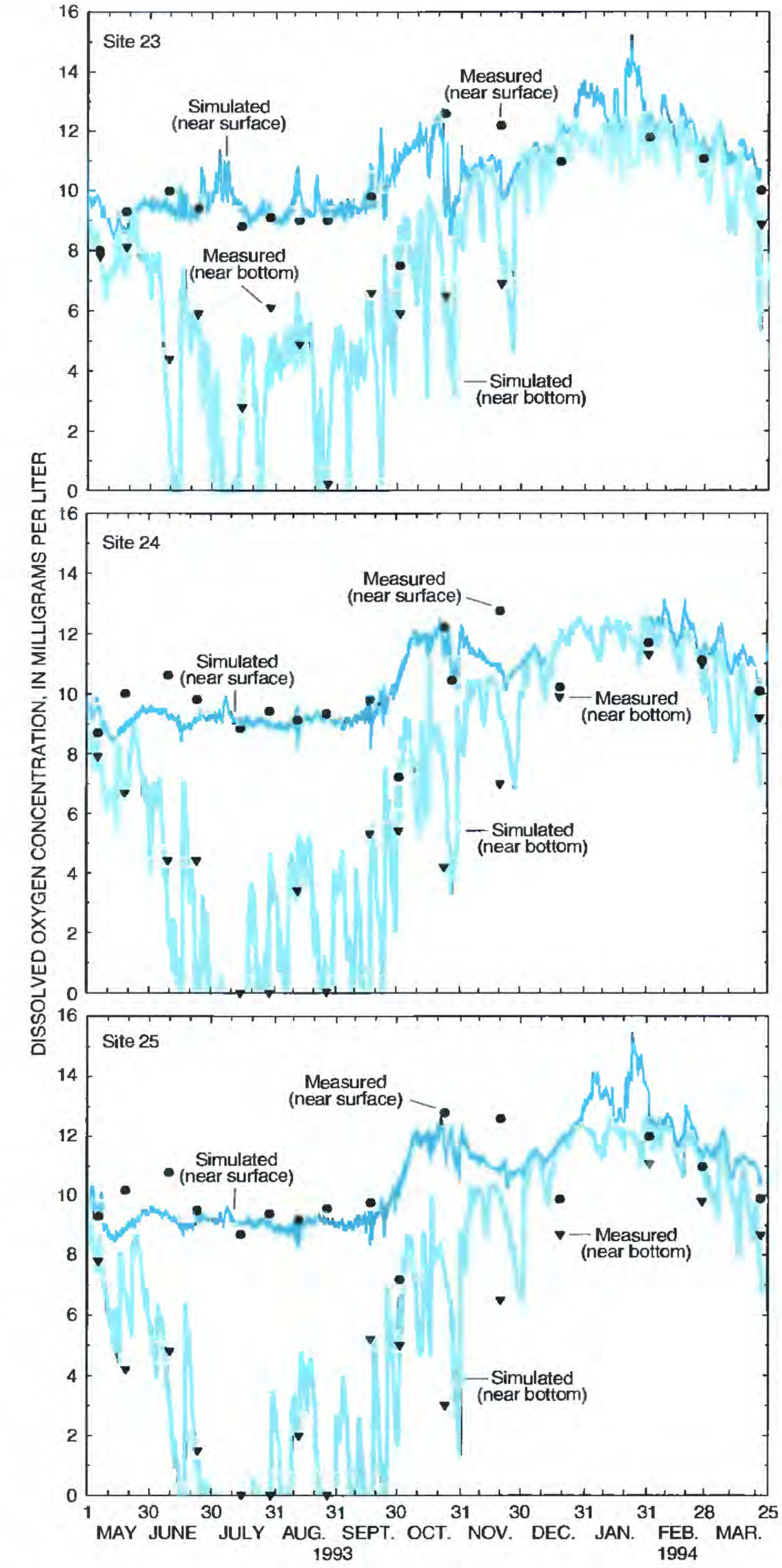

Figure 28. Measured and simulated dissolved oxygen concentrations, May 1, 1993, through March 25, 1994, at Rhodhiss Lake sites 23, 24, and 25.

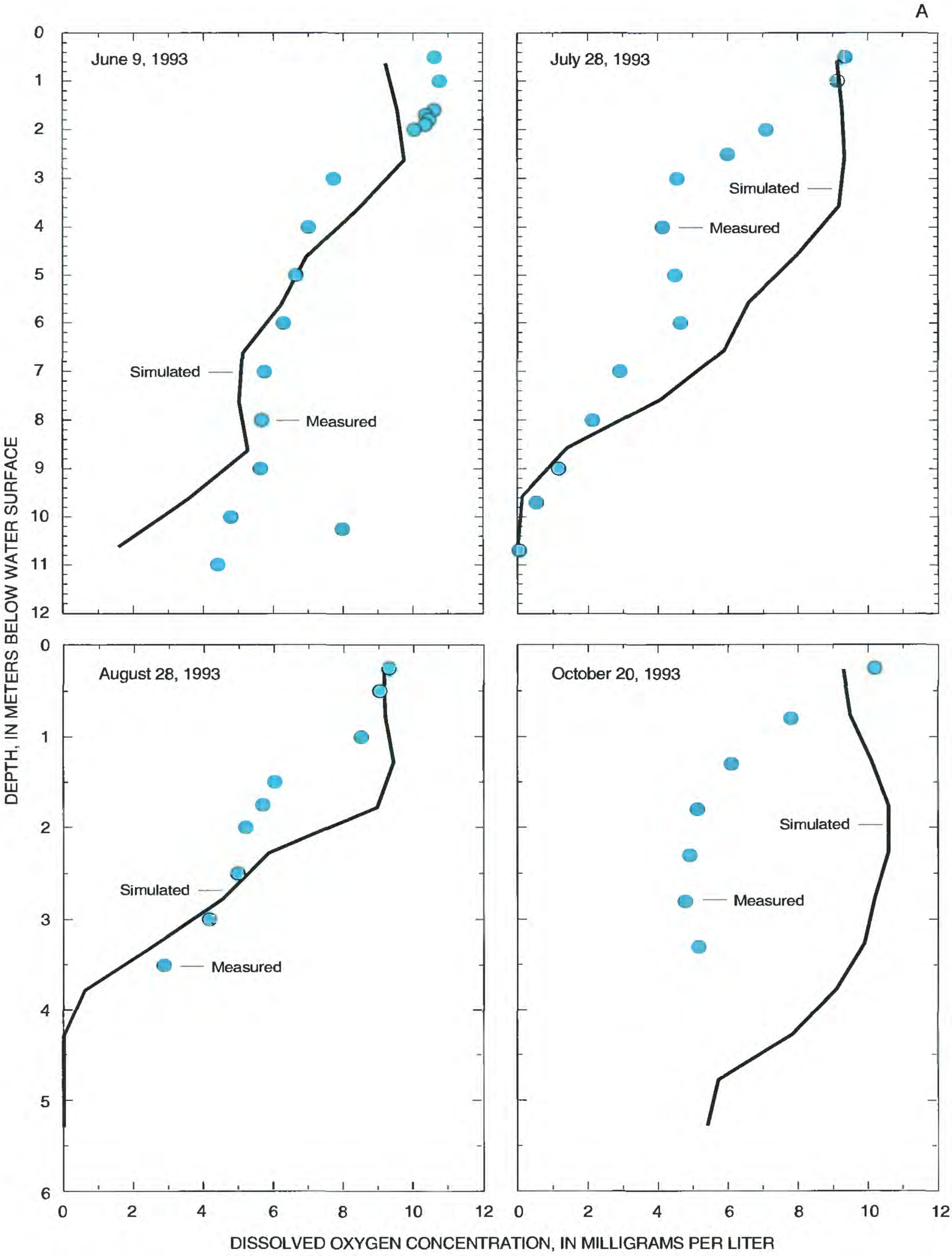

Figure 29. Measured and simulated vertical profiles of dissolved oxygen concentration at Rhodhiss Lake for (A) site 24 on June 9, July 28, August 28, and October 20, 1993.

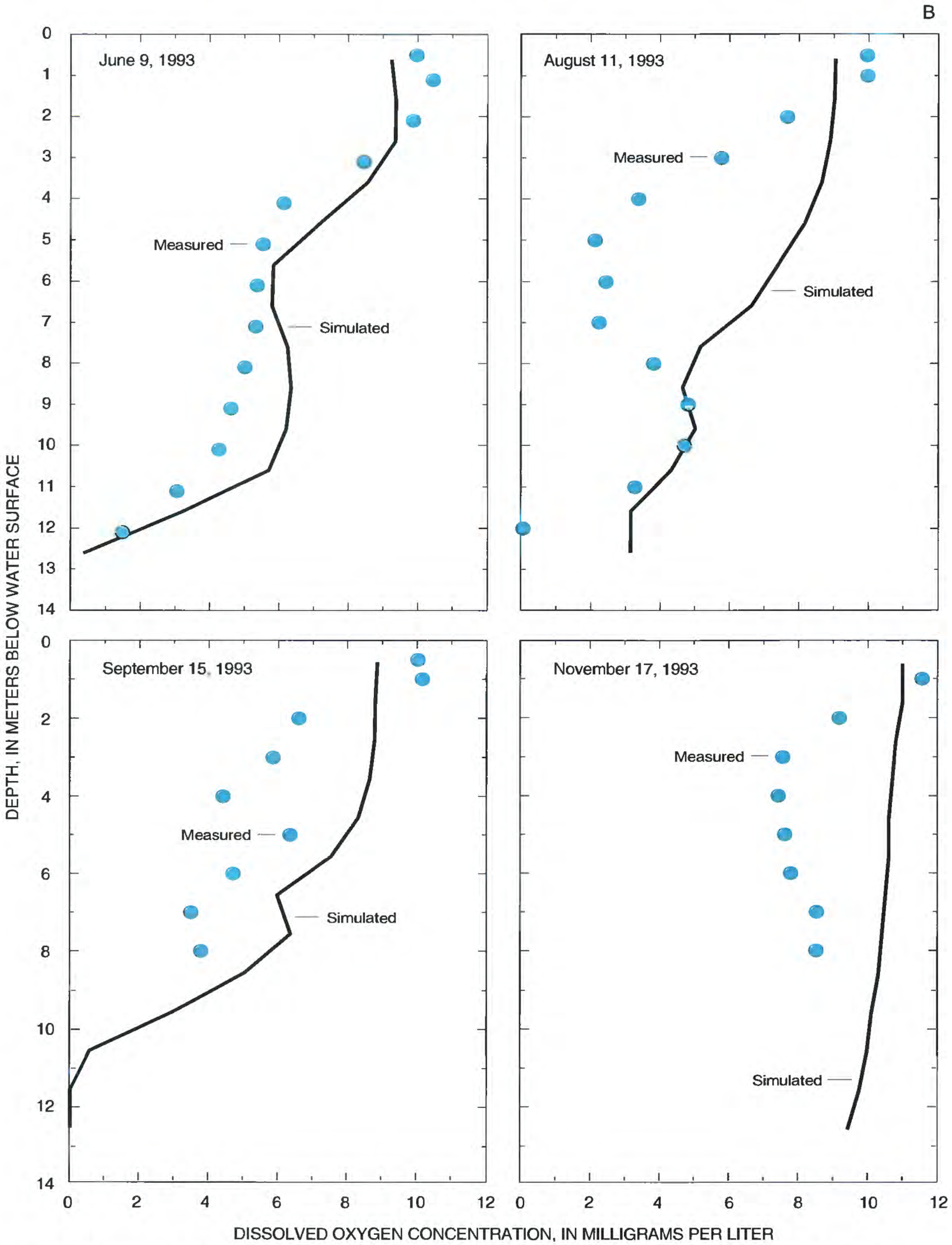

Figure 29. Measured and simulated vertical profiles of dissolved oxygen concentration at Rhodhiss Lake for (B) site 26 on June 9, August 11, September 15, and November 17, 1993.

The difference between each measured DO value at site 24 and the simulated DO at the corresponding time and depth was calculated. These differences were compared to the corresponding measured DO concentrations, to the measurement date, and to the depth of measurement (fig. 30). As in fig. 29, no consideration is given to the simulated timing of a DO event in this analysis. Measured and simulated DO closely agree when the ambient DO concentrations are greater than about 8 mg/L (fig. 30). When the measured DO is between about 5 and 8 mg/L, the simulated DO is typically greater than the corresponding value. Differences between measured and simulated DO for measured concentrations less than about 5 mg/L vary, but these differences are likely due to poor simulation of timing of events. The most biased results and most of the large differences between measured and simulated DO concentrations occurred on four dates—July 14, July 28, September 29, and October 20—when almost all simulated DO values were greater than measured values (fig. 30). For other dates, simulated values agreed fairly closely with measured values, and(or) errors were relatively evenly distributed between over- and under-predictions. Most of the over-predictions at site 24 were at mid-depth or depths between about 2 and 7 m below the water surface (fig. 30). These are the depths where the DO values between 5 and 8 mg/L, which were over-simulated, most often occur (fig. 30).

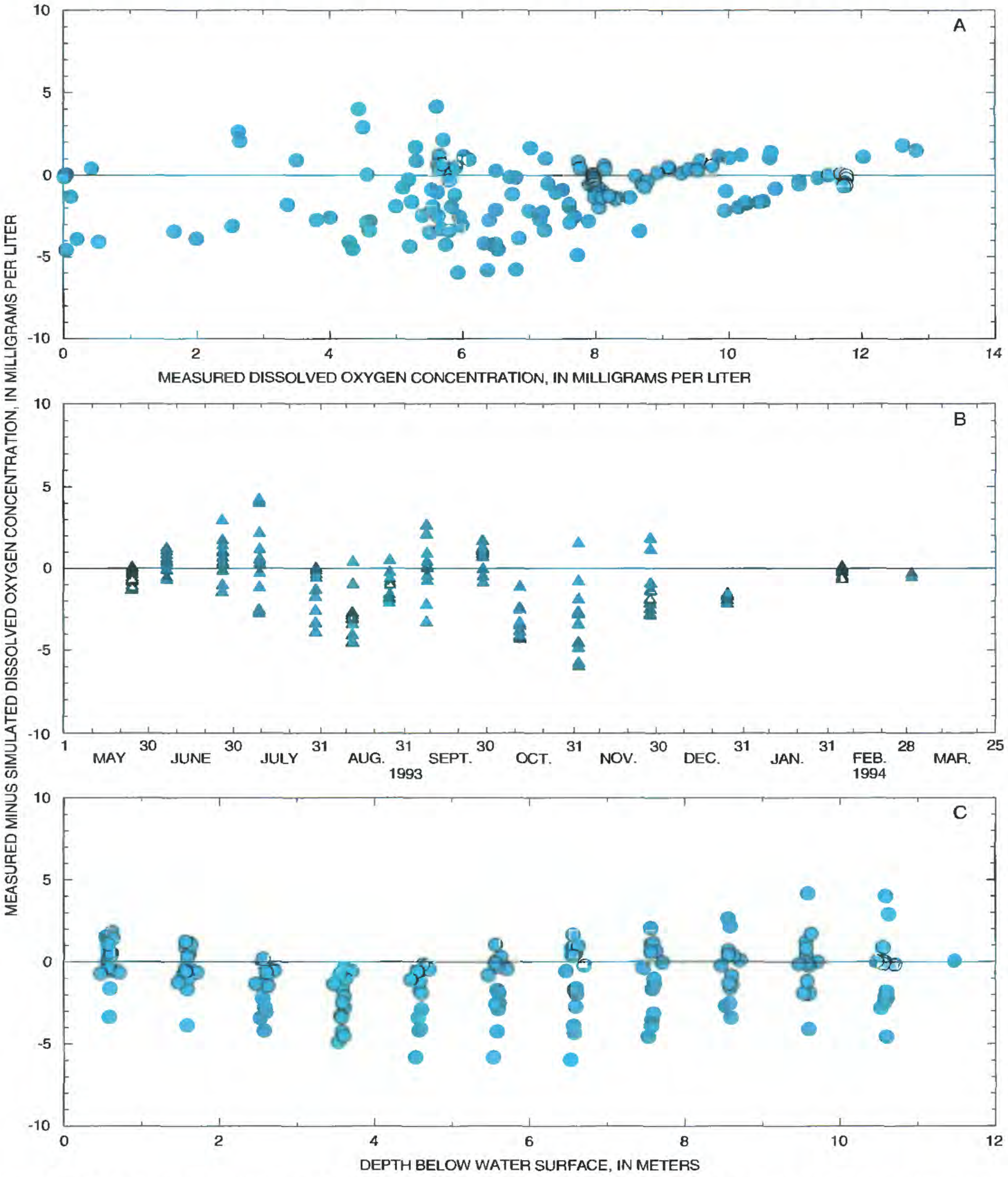

Figure 30. Relation of difference between measured and simulated dissolved oxygen concentrations at Rhodhiss Lake site 24 to (A) measured dissolved oxygen concentration, (B) date, and (C) water depth, May 1, 1993, through March 25, 1994.

The cumulative frequency of occurrence of measured DO concentrations at site 24 was determined and plotted (fig. 31). A similar analysis was performed using the simulated DO values which correspond to times at which DO measurements were made. The frequency of occurrence of DO concentrations less than 5 mg/L was almost the same for measured and simulated DO (fig. 31). For DO concentrations between about 5 and 12 mg/L, simulated DO values were greater than measured DO values. For example, about 50 percent of the time simulated DO values were less than or equal to about 9.5 mg/L, but measured DO values were less than or equal to about 8.5 mg/L (fig. 31). Regulators and resource managers generally are more interested in the occurrence of DO concentrations less than 5 mg/L than in higher values.

The overall results of the DO simulations indicate that the calibrated model provides a reasonable simulation of DO concentration in Rhodhiss Lake. Near-surface and near-bottom DO concentrations appear to be predicted better than DO concentrations at mid-depth, where DO typically is over-predicted. There is good agreement between simulated and measured frequency of occurrence of DO concentrations less than 5 mg/L in the reservoir; simulation of the exact timing of the low DO events appears to be within about 5 days of the actual occurrence. The simulated DO profiles do not always exhibit the complex variations of DO with depth which are seen in the measured profiles, which again may be because vertical mixing is over-predicted, resulting in excessive smoothing of the vertical DO gradients.

The simulated DO time series presents a clearer picture of DO dynamics in Rhodhiss Lake than do monthly to semi-monthly measurements. At sites 23, 24, and 25 (fig. 28), simulated near-bottom DO concentrations did not fall below 5 mg/L from late October 1993 through March 1994. However, near-bottom DO concentrations at site 25 were less than 4 mg/L from late June through mid-September 1993.

During an approximately 35-day period from late June through July 1993, simulated DO was less than 1 mg/L at site 25; extended periods during which DO was less than 1 mg/L also occurred in the reservoir downstream from site 25, but were less frequent and shorter in duration upstream from site 25. From an analysis of simulated hourly near-bottom DO concentrations, near-bottom DO of less than 4 mg/L occurred only 2 percent of the time during May 1993 through March 1994 at site 22, but occurred about 40 percent of the time at site 27 (fig. 32). Similarly, about 6 percent of the time bottom waters at site 24 had DO concentrations of less than 1 mg/L, but near-bottom DO concentrations of 1 mg/L or less occurred 26 percent of the time at site 26, and 30 percent of the time at site 27 (fig. 32). Near-bottom DO concentrations of about 8 mg/L and greater occurred about the same amount of time throughout the reservoir, regardless of location.

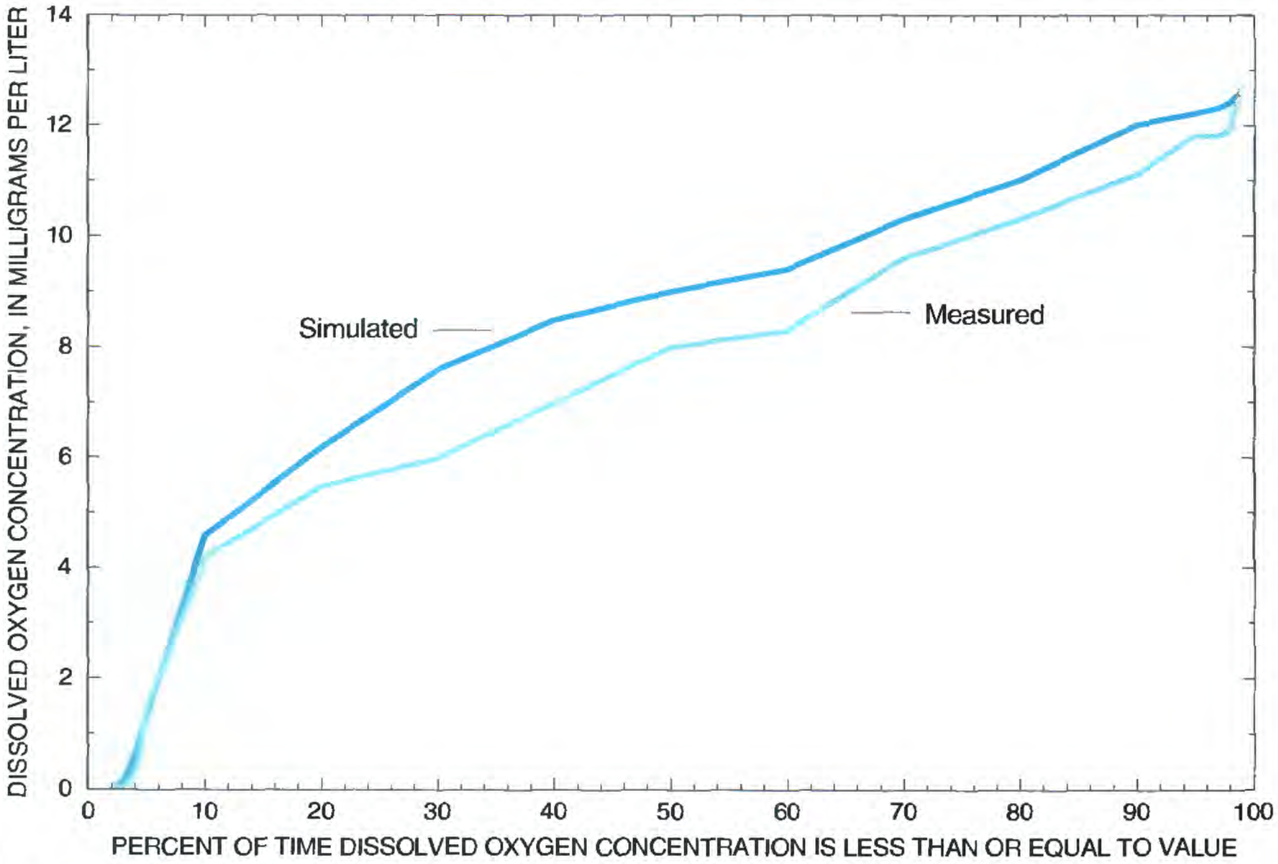

Figure 31. Measured and simulated frequencies of occurrence of dissolved oxygen concentrations at Rhodhiss Lake site 24, May 1, 1993, through March 25, 1994.

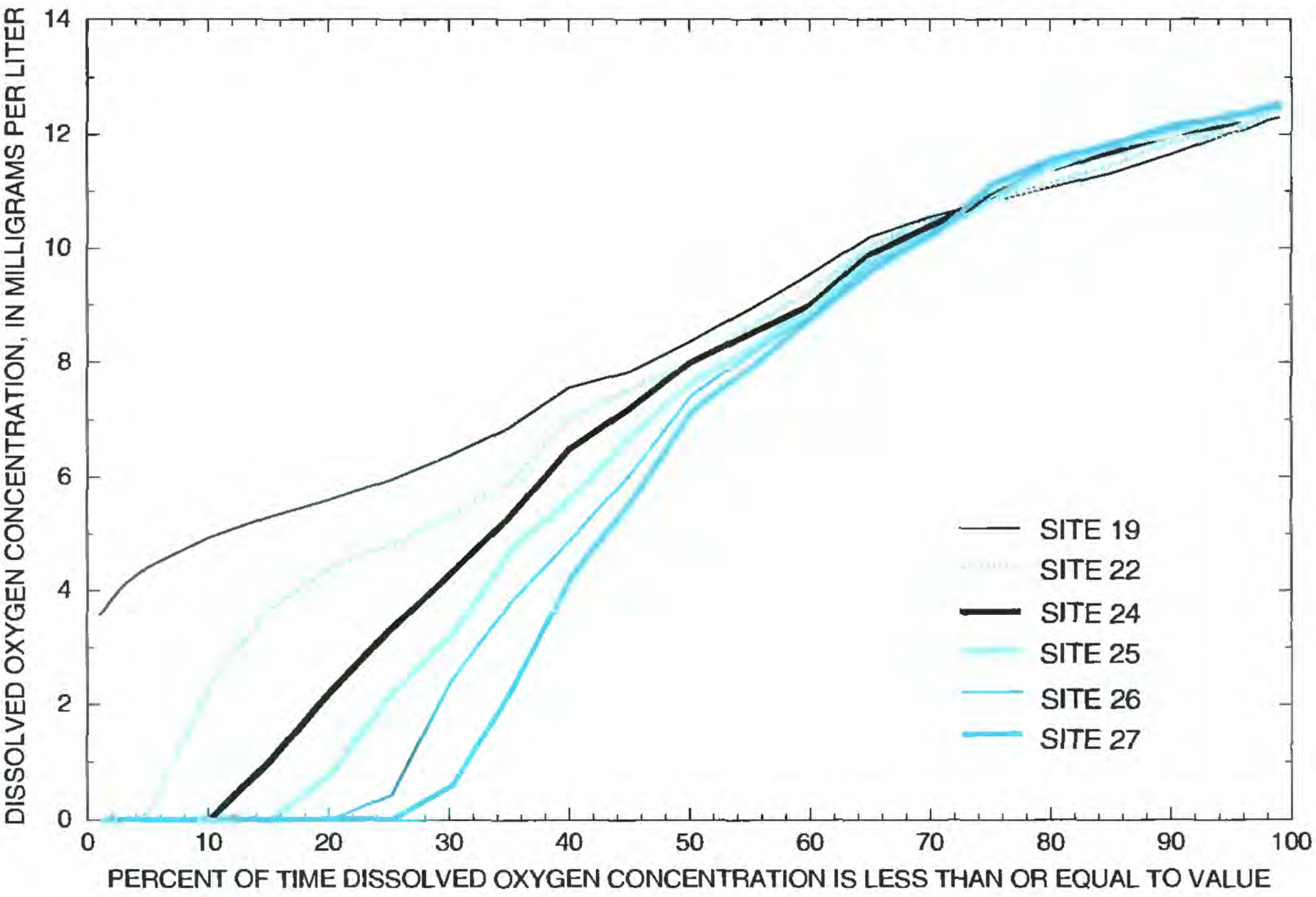

Figure 32. Simulated frequencies of occurrence of near-bottom dissolved oxygen concentrations at Rhodhiss Lake model segments 10, 14, 21, 24, 28, and 37.

Phosphorus and Algae

Algae has an (indirect) ambient water-quality standard, which is 40 µg/L for chlorophyll *a* (North Carolina Department of Environment Health, and Natural Resources, 1994a). Algal biomass also is an indication of the trophic state of a reservoir. In Rhodhiss Lake, the nutrient which appears to algal growth is phosphorus. Hence, additional information on simulated algae and PO_4 is presented.

Measured and simulated PO_4 concentrations at site 24 are in general agreement for much of the year (fig. 33). Timing of the occurrence of elevated PO_4 concentrations in bottom water is directly related to the simulated occurrence of low bottom-water DO concentrations. For example, the high PO_4 concentrations (>0.05 mg/L) from late June through late July are correlated with the occurrence of anoxic bottom water during that period (fig. 28).

Near-bottom PO_4 for December 1993 through February 1994 was significantly under-predicted (fig. 33). Elevated near-bottom PO_4 were measured during those months only at site 24; measured near-bottom PO_4 at site 27 remained low throughout the winter. However, measured TP concentrations at site 24 during December 1993 through February 1994 also were elevated (fig. 16), suggesting that the measured PO_4 concentrations at site 24 are not incorrect. If the measured PO_4 concentrations are indeed correct, then it is possible that there was an undocumented source of phosphorus to the reservoir—a source not included in the boundary conditions—such as nonpoint-source inputs directly to the reservoir. The proximity of site 24 to the Valdese WWTP discharge and the absence of high winter PO_4 values at site 27 suggest that a relation may exist between PO_4 at site 24 and the discharge. The median TP concentration in the Valdese discharge during April 1993 through March 1994 (based on monthly compliance monitoring data; North Carolina Division of Environmental Management, written commun., June 14, 1995) was 2.67 mg/L, and monthly concentrations during December 1993 through February 1994 were 2.81, 3.26, and 6.25 mg/L, respectively. These relatively high inputs, along with the absence of phytoplankton use of PO_4 during the winter months, indicate that wastewater plant discharges affected the PO_4 concentrations in Rhodhiss Lake.

Total phosphorus concentrations in Lower Creek were higher in January and February 1994 than during most of the rest of the study period (fig. 14). Three fairly large tributaries (Stafford Creek, McGalliard Creek, and Hoyle Creek) drain to Rhodhiss Lake just upstream from site 24 (fig. 2). Consequently, it is possible that phosphorus concentrations in these tributary streams also were high during the winter months, as in Lower Creek. And these high loadings were responsible for the high phosphorus concentrations at site 24.

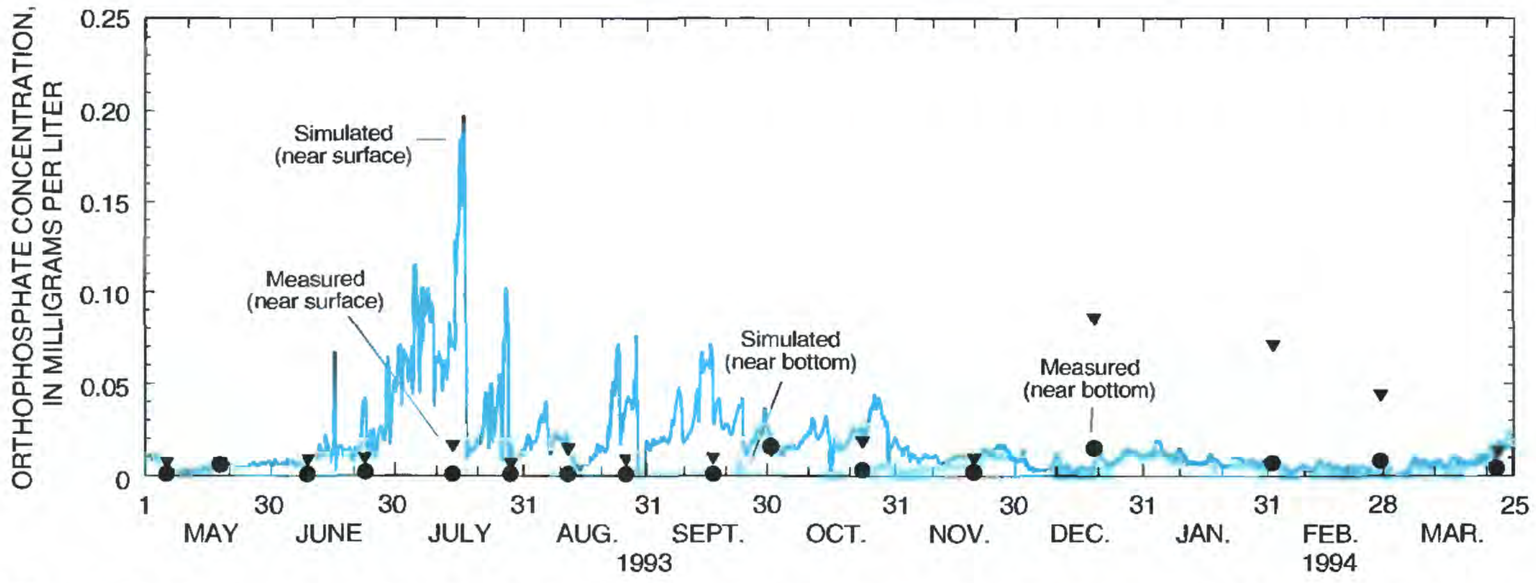

Figure 33. Measured and simulated phosphorus concentrations at Rhodhiss Lake site 24, May 1, 1993, through March 25, 1994.

Finally, it is possible that the calibrated model performs poorly in predicting winter PO_4 concentrations in Rhodhiss Lake. However, November 1993 and March 1994 measured and predicted PO_4 concentrations agreed closely. Under-prediction of winter PO_4 in Rhodhiss Lake also should not affect the simulation of the occurrence and magnitude of spring and summer algal blooms because the high winter PO_4 concentrations are flushed through the system by spring.

Simulated algal concentrations generally agree with measured values at site 24, with a few exceptions (fig. 34). Algae concentrations were over-predicted on July 14 and September 15, when PO_4 concentrations also were over-predicted (fig. 33). Although simulated algal concentrations at site 24 agree with measured values on June 9 and June 23, algal concentrations were under-predicted at site 27 for these dates. These differences likely relate to the patchy nature of phytoplankton in Rhodhiss Lake. In fact, on November 17, 1993, at site 24, USGS data indicate that the chlorophyll *a* concentration was 52 µg/L, but a sample collected by the North Carolina Division of Environmental Management (DEM) on the same day and within 100 m of the USGS sampling location had a chlorophyll *a* concentration of 13 µg/L. Although different sampling methodologies and analytical procedures were used by the USGS and DEM, these differences do not explain the large difference in measured chlorophyll *a*. The simulated algal concentration on November 17 more closely agrees with the DEM measurement than the USGS measurement. However, the elevated DO concentration at site 24 on November 17 (fig. 28) suggests the presence of strong photosynthetic activity.

The accurate simulation of algal concentrations is very difficult for several reasons. First, algae are not uniformly distributed vertically or horizontally in the reservoir, but often occur in patches. Consequently, obtaining a representative sample can be a problem, as indicated by the November 17, 1993, sample results. Second, phytoplankton are simulated by the Rhodhiss Lake model as a single assemblage, so distinctions among algal types which bloom under different ambient conditions cannot be made by model simulations. Third, simulated algal concentrations represent the accumulated results of simulated solids concentrations, light penetration, water temperature, transport, and nutrient concentrations. Errors in simulations of each of these parameters are reflected in the simulated algal concentrations. Finally, algal concentration (biomass) is simulated, but chlorophyll *a* is measured as an indicator of biomass. For this application, chlorophyll *a*, in micrograms per liter, is converted to algal concentration, in milligrams per liter, by multiplying chlorophyll *a* by the factor 0.067 (American Public Health Association and others, 1992; Cole and Buchak, 1995). No data were collected to verify this factor for Rhodhiss Lake. Other processes, such as self-shading and consumption of algae by zooplankton and fish, which may be important to the accurate simulation of algal concentrations, are not included in the model formulations.

Simulated PO_4 concentrations are strongly related to simulated bottom-water DO concentrations. Reasonable simulations of PO_4 concentrations in Rhodhiss Lake, therefore, depend on accurate simulations of DO. The release rate of PO_4 from bottom sediments used in the Rhodhiss Lake model during anoxic conditions is somewhat lower than values suggested by Cole and Buchak (1995). Even with the low release rate, PO_4 and algal concentrations are generally over-predicted (fig. 34). The effects of changes in influent PO_4 loads and in PO_4 bottom release rates on simulated results are characterized in a subsequent section.

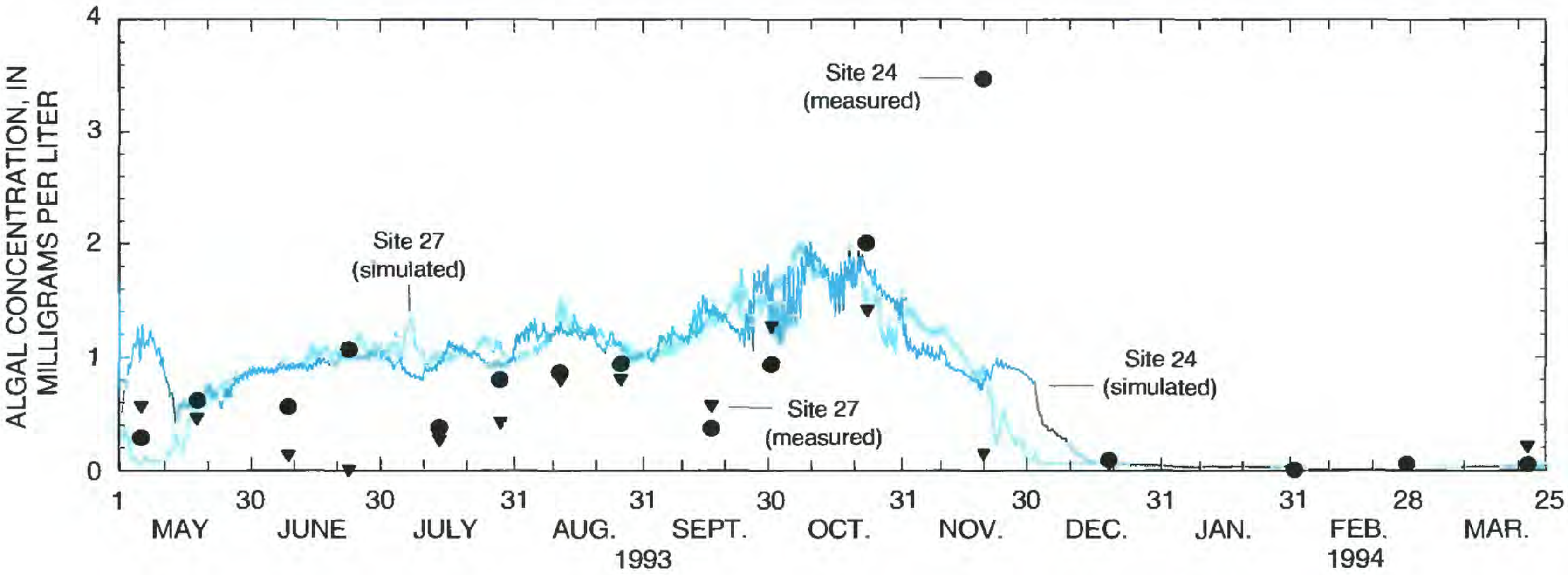

Figure 34. Measured and simulated algal concentrations at Rhodhiss Lake sites 24 and 27, May 1, 1993, through March 25, 1994.

Sensitivity Analysis

Sensitivity analysis is the determination of the effects of small changes in calibrated model parameters on model results. A complete sensitivity analysis for all of the model parameters in the Rhodhiss Lake model was not conducted. The Rhodhiss model includes more than 50 parameters (tables 8 and 9), and a complete sensitivity analysis would be a very lengthy process. However, more than 50 water-quality simulations were made after the hydrodynamic component of the model was calibrated. Results from these simulations form the basis for the sensitivity analysis presented in this section.

Model results were not sensitive to changes in hydraulic parameters (table 8). Small-scale physical processes are resolved by the relatively detailed Rhodhiss Lake model computational grid. Subgrid scale processes represented by the hydraulic parameters were relatively unimportant, so model results did not change significantly with changes in these parameters.

Vertical mixing appears to be over-predicted by the model. The vertical mixing coefficient, as formulated in the model, is a function of the vertical density gradient, the vertical gradient of horizontal velocity, wind shear, and several coefficients which are not allowed to vary. Consequently, simulated vertical mixing can only be controlled indirectly. Two factors that indirectly affect vertical mixing are wind-sheltering coefficient and the Chezy resistance coefficient. Neither factor greatly affected simulated vertical distributions of temperature and DO. Simulated temperature in the upper layers of the reservoir are somewhat affected by the coefficients governing light extinction and adsorption of solar radiation. These upper layer temperatures, in turn, affect the density distribution and thus, vertical mixing. Model results are relatively insensitive to the parameters controlling heat exchange at the reservoir bottom, partly because the area of the bottom-layer computational grids is small relative to the entire surface of the sediment-water interface.

Model parameters affecting chemical kinetics are summarized in table 9. Cole and Buchak (1995) suggested reasonable values, and in some cases a range of values, for each parameter. Initial Rhodhiss Lake model simulations were made using the values suggested by Cole and Buchak (1995).

Algal concentrations were most sensitive to the algal growth rate, but were fairly insensitive to the mortality rate, settling rate, and saturation light intensity. The simulated algal concentrations were relatively insensitive to the algal temperature-rate multipliers, but the timing of the initiation and ending of algal growth was affected by these multipliers.

DO concentrations appear to be controlled primarily by reaeration, photosynthetic oxygen production, and to a lesser extent, organic matter decay. The only parameter directly affecting reaeration is the wind-sheltering coefficient, but DO concentrations are insensitive to changes in this parameter. Photosynthetic oxygen production is most sensitive to the algal growth rate and the oxygen stoichiometric equivalent for algal growth. During some periods of high algal growth, simulated DO concentrations near the water surface were less than DO concentrations a few meters below the surface. This phenomenon was occasionally observed in the field (fig. 29A and 29B, June 9, 1993) and was likely the result of the combined effects of (1) desorption of DO in the surface layer under supersaturated conditions and (2) the downward transport of oxygen-producing algae from convective heating. Hence, the DO concentrations were often oversimulated at depths from 2 to 5 m below the surface (fig. 29A, July 28 and August 28, 1993). This also is further evidence that vertical mixing may be overpredicted by the model. Although a number of parameters control production of organic matter, including both dissolved and particulate material, the relation between organic matter and DO concentrations appears to be most sensitive to changes in the temperature-rate multipliers for organic matter decay. Sediment oxygen demand is represented in the Rhodhiss Lake model as a first-order process (decay of organic matter in the bottom layer), and a sediment oxygen demand rate was not explicitly included in the model.

According to model simulations, algal growth in the upper layers of the reservoir was limited by phosphorus. Consequently, phosphorus concentrations in the upper layers were generally quite low and most sensitive to the algal growth rate. Phosphorus is released from the bottom sediments when DO in the bottom layer is less than 0.1 mg/L. The phosphorus concentrations in the mid to lower layers of the reservoir and algal concentrations in the upper layers were sensitive to changes in the phosphorus release rate from the sediments. Changes in other parameters controlling phosphorus kinetics resulted in only small changes in simulated phosphorus and algal concentrations. Additional discussion on the effects of changes in PO_4 loadings on simulated DO and algal concentrations is included in the following section.

Model Application

The calibrated Rhodhiss Lake model was used to simulate the movement of a neutrally buoyant conservative material through the reservoir. The effects of increased air temperature and wind speed on DO concentrations were evaluated using the model. Simulations were made to evaluate the effects on DO concentrations and algae as a result of changes in PO_4 loadings from Catawba River inflows (nonpoint sources), from bottom sediments, and from a point source.

Simulation of Conservative Material Transport

Transport of material released from two locations at two different times was simulated. The material was released during August 16-17 (summer) and December 24-25 (winter). Inflow during the summer release was between 12 and 40 m^3/s and between 16 and 27 m^3/s during the winter. Outflows from the reservoir were less than 10 m^3/s during most of the time for both the summer and winter releases, with occasional higher releases. Release locations were at Huffman Bridge (site 20, fig. 2), the upstream boundary of the model, and in segment 17 (fig. 19; near the Valdese wastewater discharge). The temporal distribution of the release was triangular shaped, with a time base of 2 days and a peak concentration of 1,000 parts per thousand (ppt) or 1,000,000 mg/L. The inflow rate of the material was equal to the inflow rate of the water.

During the summer release, the Catawba River water temperature was colder than the near-surface waters in the reservoir. Consequently, the material released at Huffman Bridge sank fairly rapidly as the material moved into the reservoir. The concentration of the material near the water surface 5 km downstream from the release was less than 5 percent of the initial peak concentration. Most of the sinking occurred between 2 and 5 km downstream from the release (fig. 35A and B), which is about where the reservoir is transformed from a primarily riverine to a lacustrine system. The peak concentration attenuated quickly between 8 and 10 km downstream from the release (fig. 35C), and the near-bottom peak concentration arrived at the dam about 12 days after the release. However, the highest concentration at the dam was near mid-depth at about 14 percent of the initial peak concentration and arrived 13 days after the release. The mid-depth peak likely reflects both an interflow phenomenon and to a lesser degree, the effects of the reservoir withdrawal on flow patterns. The near-bottom concentration was 1 percent of the initial peak concentration 15 days after the release, but an additional 25 days were required for the mid-depth concentrations to fall below 1 percent of the initial peak.

Between 0 and 5 km downstream from the release, there was less attenuation of the peak concentration following the winter release than following the summer release (figs. 35A and B, 36A and B). However at Rhodhiss Dam, the peak concentration following the winter release (about 7 percent of the initial peak) was about half of the highest concentration at the dam after the summer release. The difference is the result of greater vertical mixing and, hence, dilution in the winter when the reservoir was less thermally stratified. Only about 17 days were required in the winter for the concentration at the dam to fall below 1 percent of the initial concentration.

Mixing and transport of a conservative material released from a point source in model segment 17 was simulated. The inflow rate of the tracer, which was equal to the inflow rate of the point source, was 0.21 m^3/s. This flow was on the order of 1 percent of the flow into the reservoir during the release periods. The temporal distribution of the material was the same as

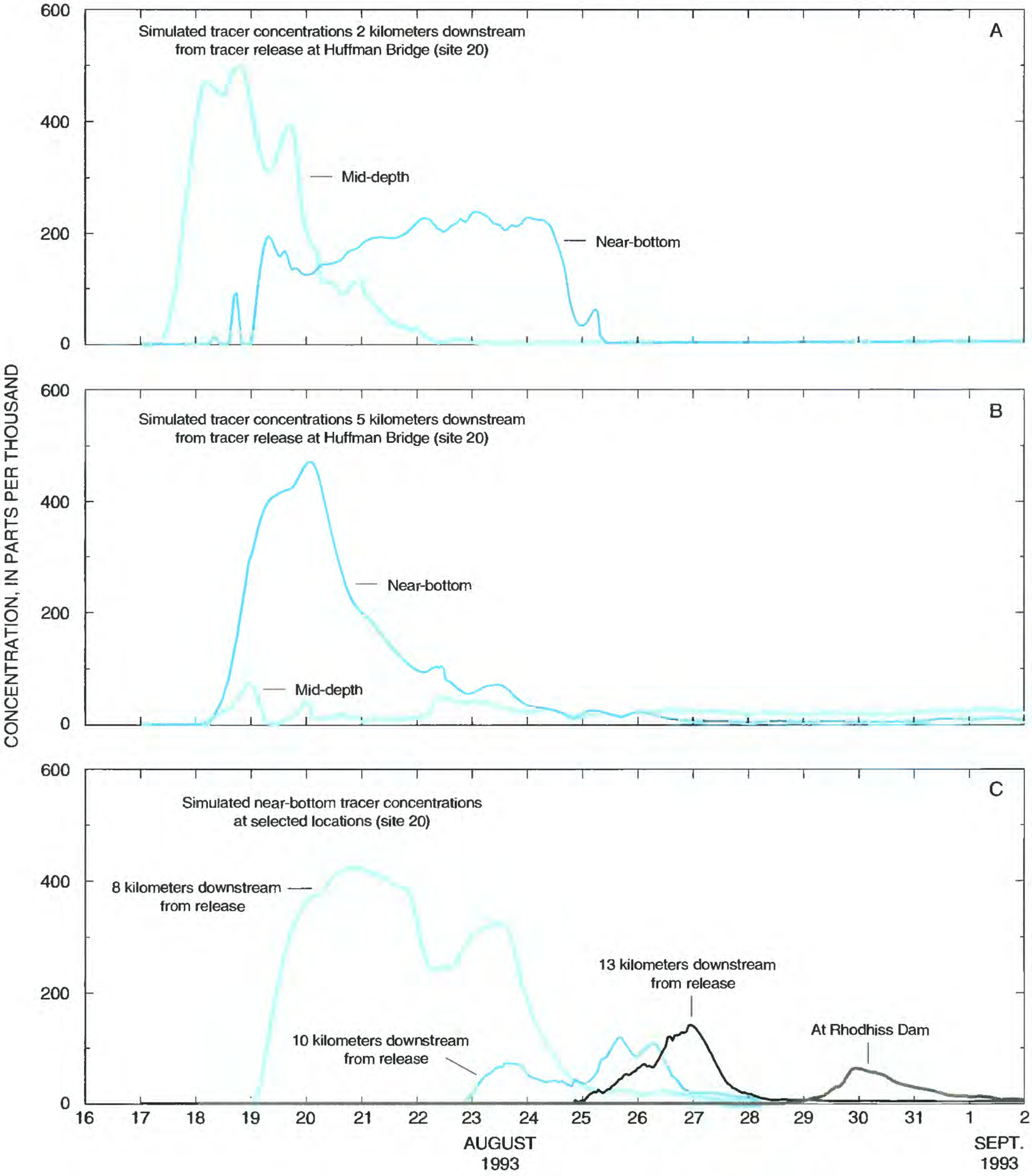

Figure 35. Simulated tracer concentrations at selected locations downstream from tracer releases in Rhodhiss Lake following a summer release at Huffman Bridge.

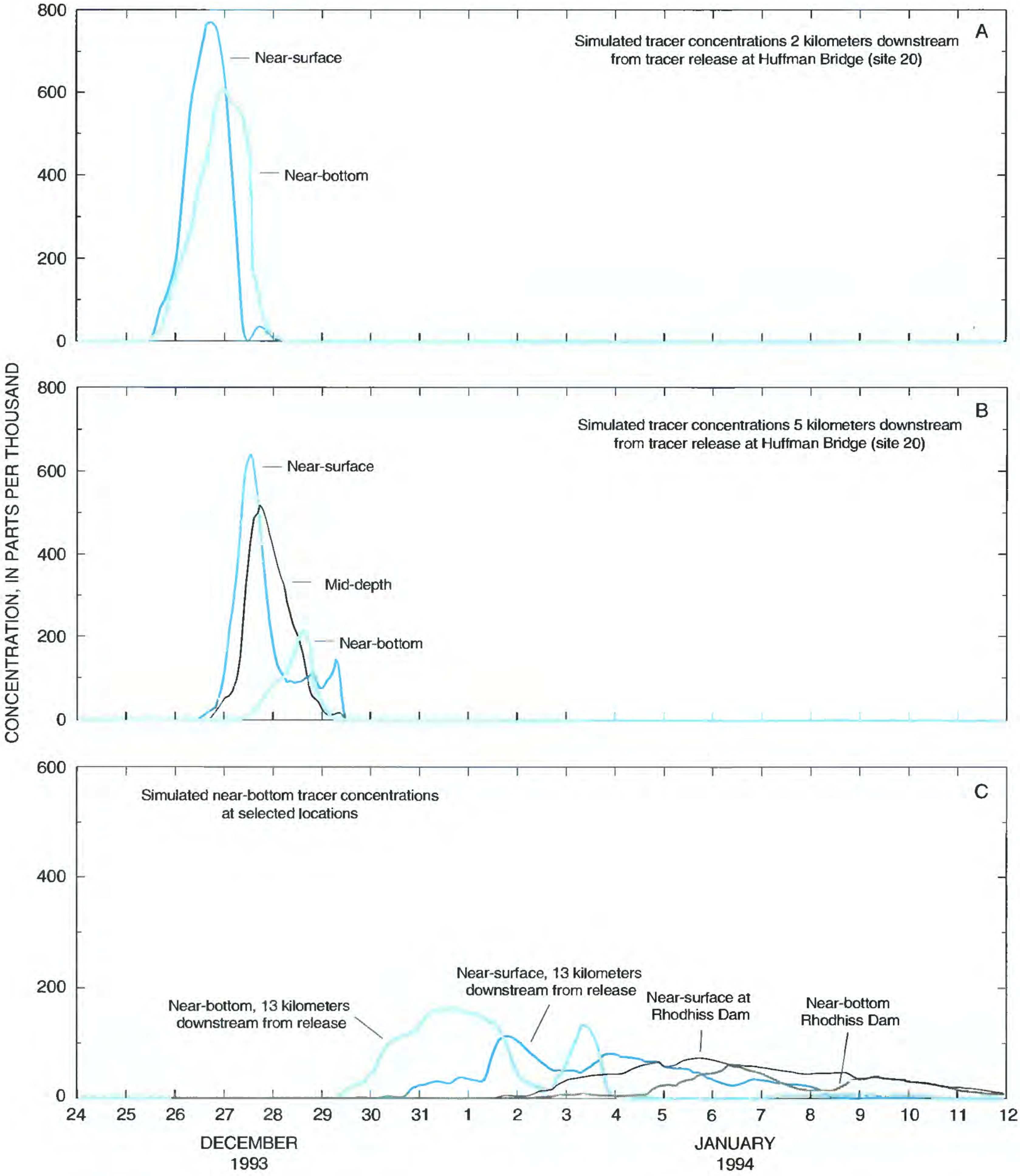

Figure 36. Simulated tracer concentrations at selected locations downstream from tracer release in Rhodhiss Lake following a winter release at Huffman Bridge.

for the simulated release at Huffman Bridge (site 20). The vertical position of the inflow was determined by matching the temperature of the inflow with the temperature in the reservoir so that the relative densities of the two flows (point source and reservoir) were considered in placing the inflow into the reservoir.

The Rhodhiss Lake model simulates the laterally averaged value of velocity, temperature, and material concentrations in each computational segment. This means that near a point-source release, the simulated peak concentration of a constituent will be less than the actual peak concentration because in the reservoir, the constituent is not laterally uniform near the release point. Rather, the lateral concentration distribution near the release point is asymmetric, with the peak concentration near the side of the reservoir at which the release point is located and with low or zero concentrations along the opposite shore. The rate of lateral mixing is proportional to the flow velocity, so lateral mixing will occur more rapidly in the riverine section of the reservoir than in the lacustrine section. These factors should be considered when interpreting simulated concentration distributions resulting from any point source released into the reservoir.

For the summer release of a conservative substance into segment 17, most of the substance remained in the surface layers, with little downward vertical mixing (fig. 37), primarily because the temperature of the discharge and surface waters in the reservoir were nearly the same. This implies that, under similar thermal conditions, nutrients introduced by a point source to the reservoir at this location in the summer might be delivered to the photic zone where the nutrients would be most available for uptake by phytoplankton.

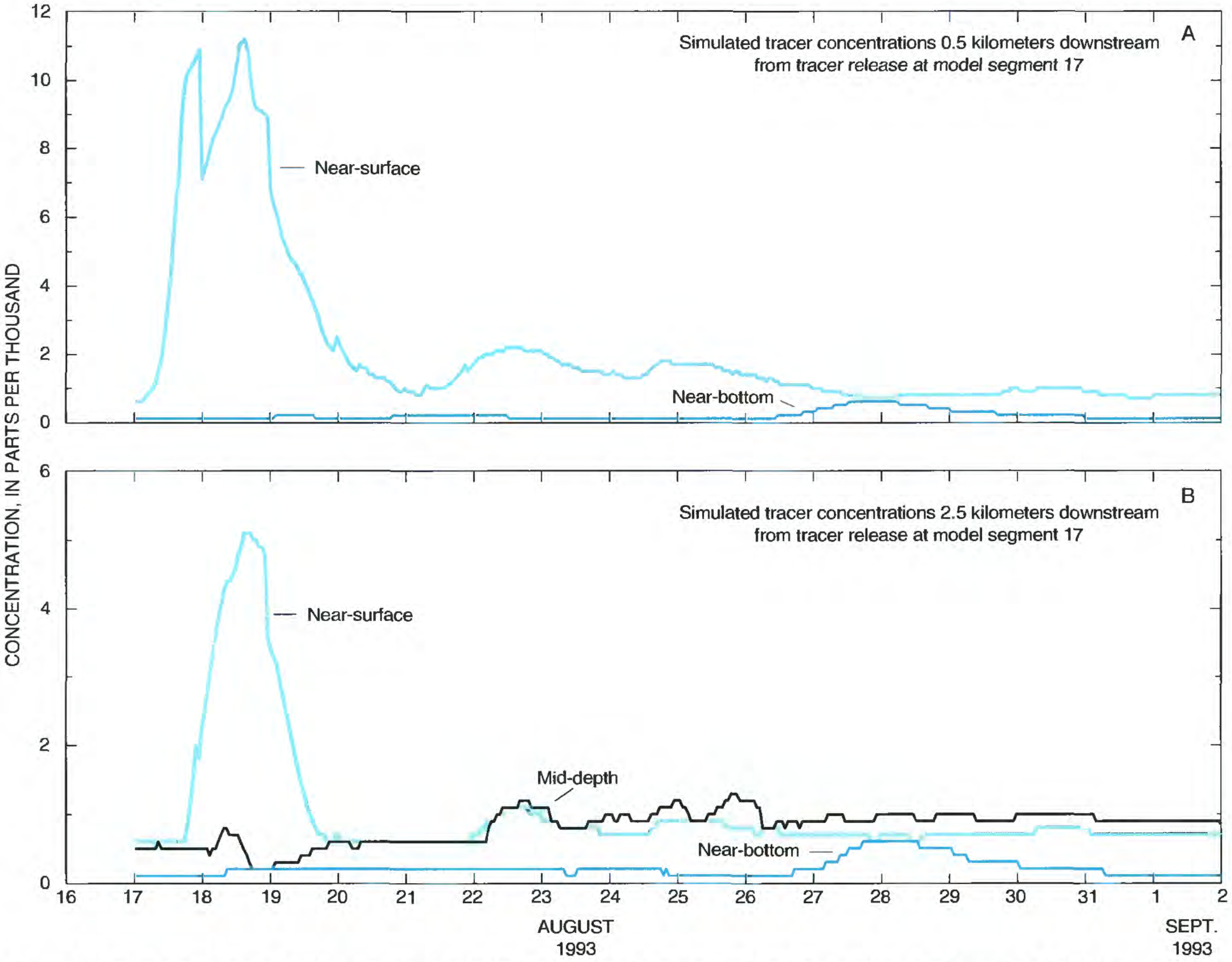

Figure 37. Simulated tracer concentrations at selected locations downstream from tracer release in Rhodhiss Lake following a summer release in model segment 17.

The primary difference between the simulated concentration distributions for the summer (fig. 37) and winter (fig. 38) releases into segment 17 was that greater vertical mixing occurred during the winter. Transport rates were much lower near the reservoir bottom than at the surface; the simulated peak concentration at 2.5 km downstream from the release occurred at the surface 2 days before the peak occurred near the bottom. Simulated peak concentrations at the dam were less than 0.1 percent of the initial concentration in the release.

These examples of simulation of the transport of a conservative material demonstrate (1) use of the Rhodhiss Lake model in evaluating the movement of a short-term or a long-term release into the reservoir; (2) the manner in which nonconservative materials move through the reservoir, without the confounding effects of chemical transformations, regeneration, and settling; (3) the difficulty in identifying a single residence time for the reservoir—"residence times" vary seasonally, as well as with depth; and (4) the effects of density stratification and vertical mixing on transport processes. Other scenarios involving different flow rates, density structures, release locations, release rates, and release durations can be simulated to further evaluate the movement of matter through Rhodhiss Lake.

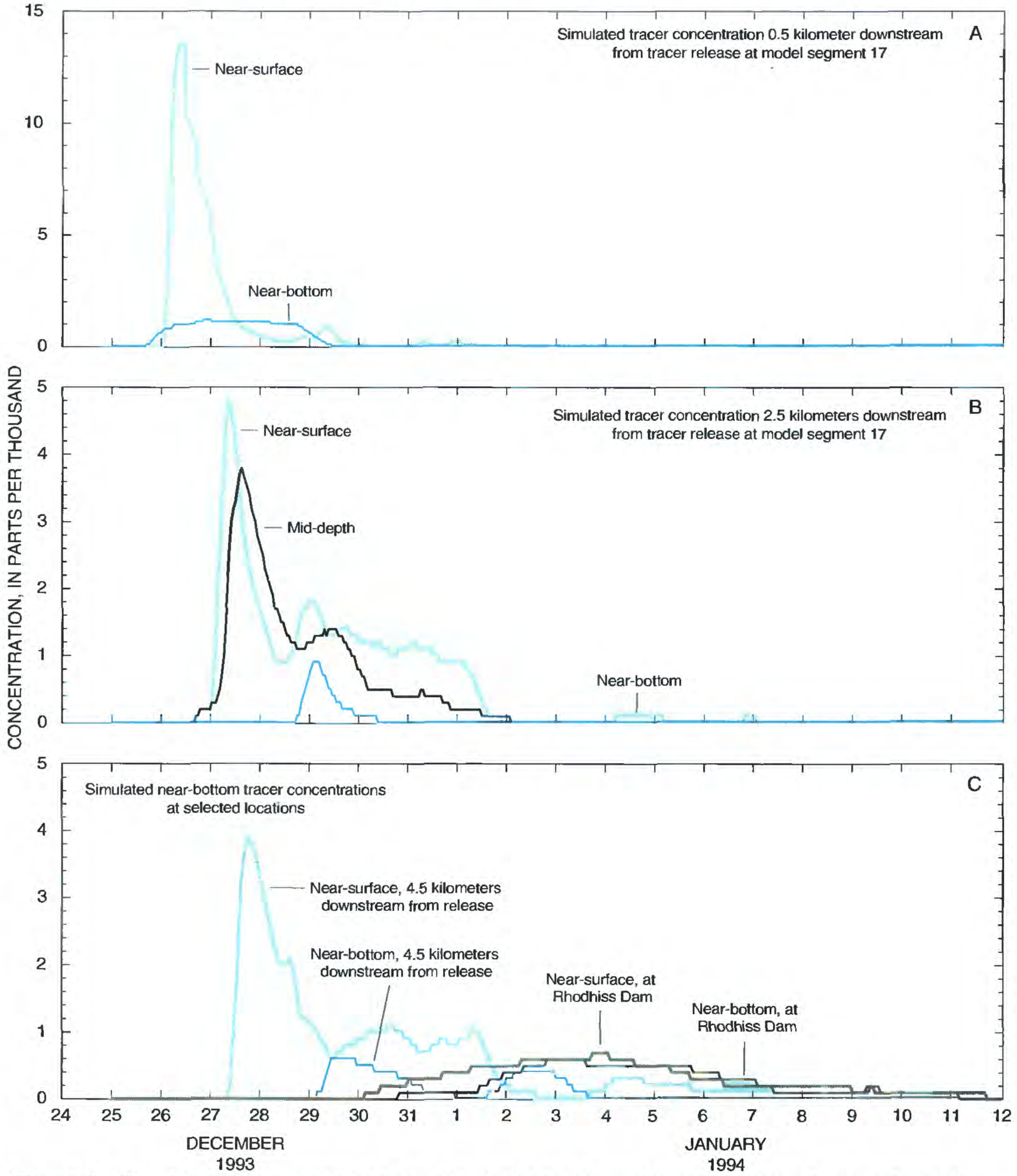

Figure 38. Simulated tracer concentrations at selected locations downstream from tracer release in Rhodhiss Lake following a winter release in model segment 17.

Simulation of Effects of Wind and Air Temperature

Changes in wind speed might be expected to have an effect on simulated vertical mixing rates and surface exchange of heat and DO. All wind speeds for the simulation period were increased by 30 percent. These changes had little effect on the vertical distribution of temperature or DO (fig. 39). Likewise, a reduction of wind speed by 30 percent had little effect on vertical mixing.

In order to evaluate the effects of warmer ambient temperatures on reservoir water quality, all recorded air temperatures were increased by 1 °C for the entire simulation period. Inflowing water temperatures were not adjusted, although changes in air temperature would likely result in an associated change in Catawba River water temperature. This means that the difference between the temperature of inflowing water and reservoir water will be greater than for the calibration condition. Because of this greater density difference, inflowing water will plunge to greater depths in the reservoir with subsequent confounding effects.

The warmer temperatures slightly increased the amount of time that simulated DO concentrations were less than 4 mg/L throughout the reservoir. For example, site 24 simulated near-bottom DO was below 4 mg/L about 4 more percent of the time with elevated air temperatures than with the base condition (fig. 40). Simulated algal concentrations also were only slightly affected by the change in air temperature. At model site 24, the maximum simulated algal concentration decreased from 2.02 mg/L (30 µg/L of chlorophyll *a*) to 1.85 mg/L (28 µg/L of chlorophyll *a*), and the median concentration decreased from 0.87 mg/L (13 µg/L of chlorophyll *a*) to 0.78 mg/L (11.6 µg/L of chlorophyll *a*) with increased air temperature. The decrease in DO may be associated with the lower algal concentrations as well as the lower DO saturation concentration associated with higher water temperatures.

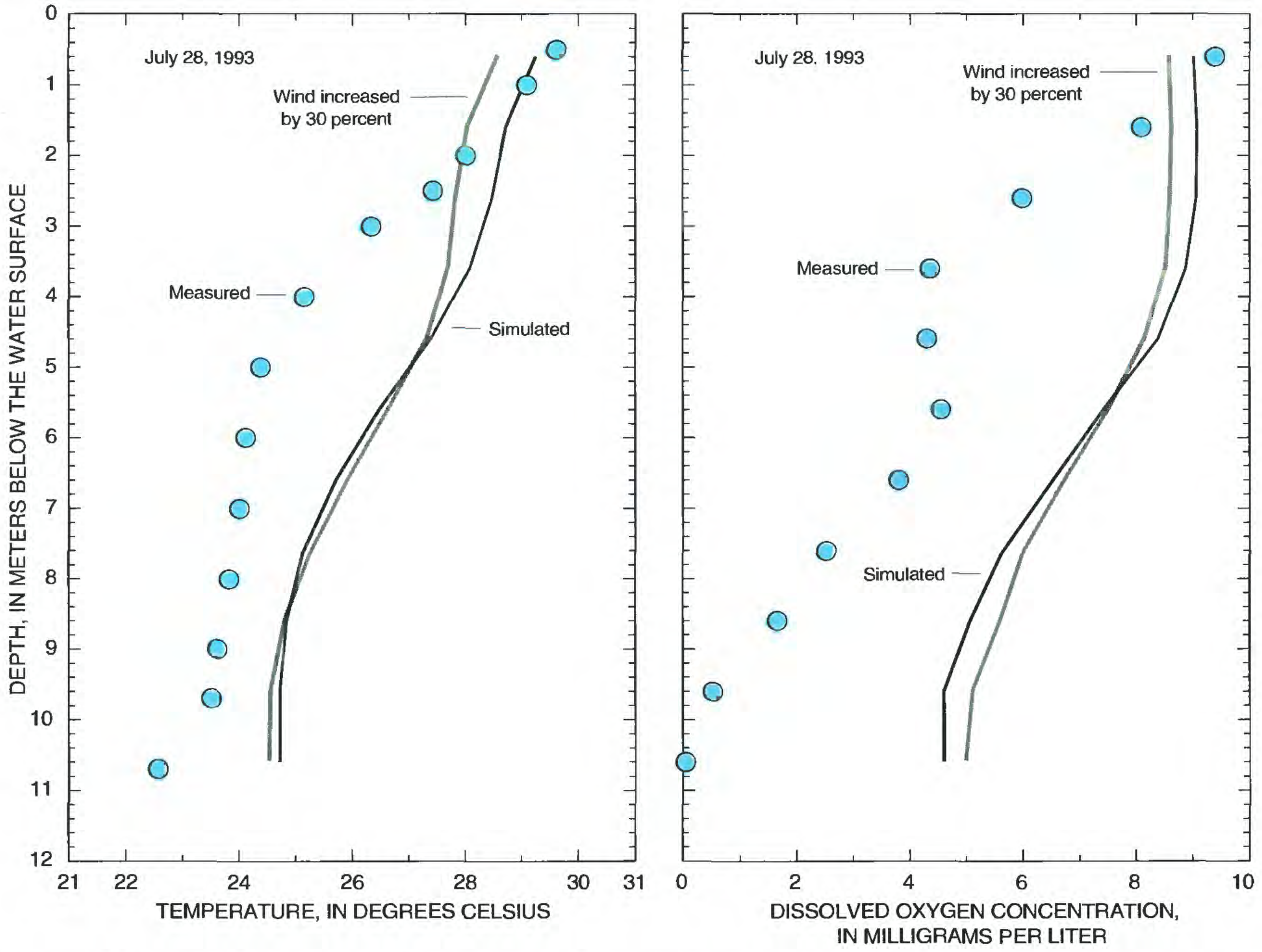

Figure 39. Simulated and measured vertical profiles of water temperature and dissolved oxygen concentration on July 28, 1993, at site 24, with simulations for measured wind, and wind speed increased by 30 percent.

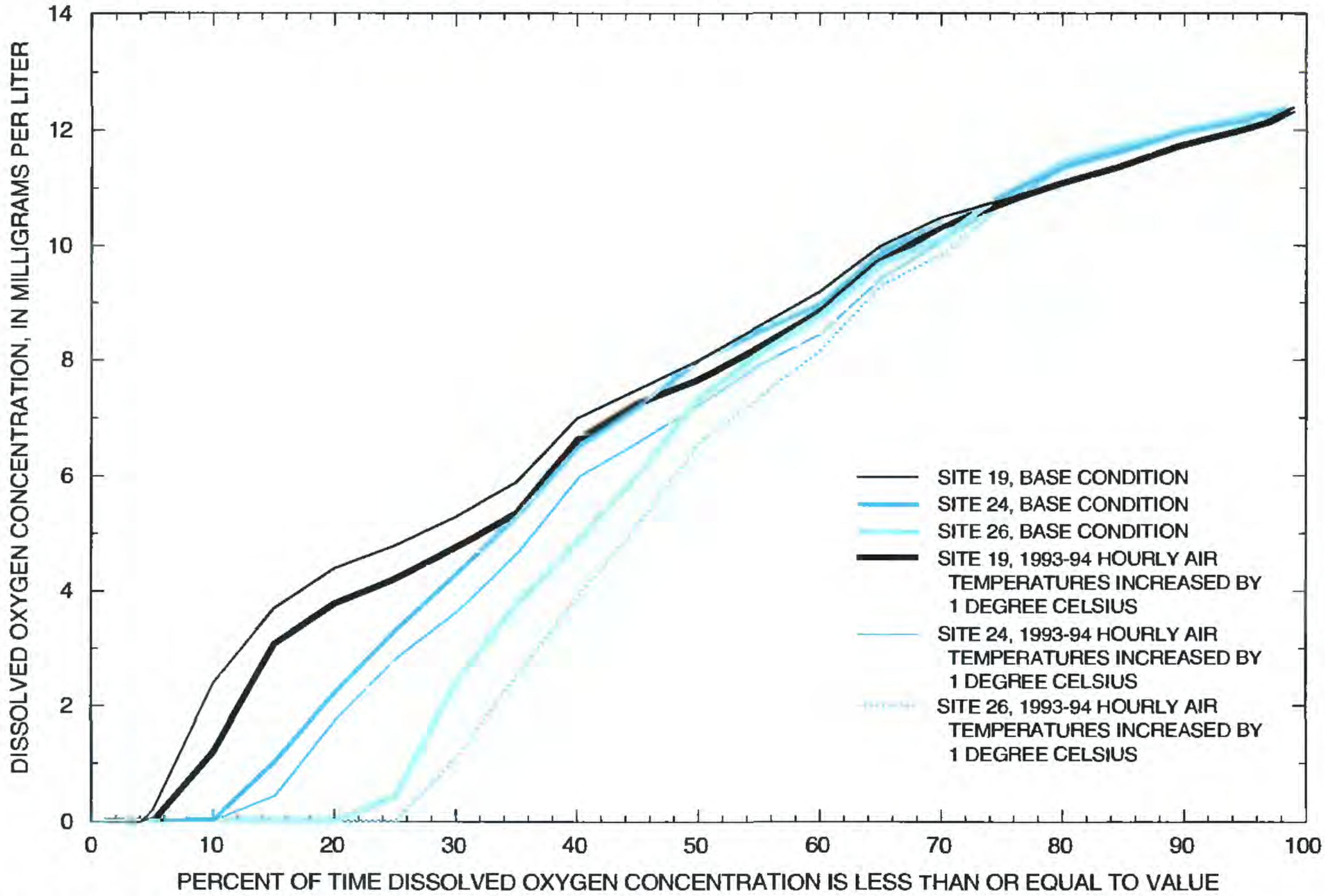

Figure 40. Simulated frequencies of occurrence of near-bottom dissolved oxygen concentrations at Rhodhiss Lake model segments 14, 21, and 28, May 1, 1993, through March 25, 1994, with simulations for measured air temperatures, and with measured air temperatures increased by 1 °C.

Simulation of Effects of Changes in Orthophosphate Loadings

Simulation results indicated that algal growth in the upper 2 to 4 m of Rhodhiss Lake is limited by PO_4 availability. Consequently, simulated algal concentrations should be affected by PO_4 concentrations in the reservoir. PO_4 inflow concentrations at the upstream boundary were reduced by 30 percent to simulate the possible effects of future management actions, including possible phosphate reductions, on algal concentrations. Reductions in the inflow concentrations resulted in an approximately 20-percent decrease in the peak algal concentration (fig. 41). Most of the reduction in simulated algal biomass occurred in the late summer and early fall.

The simulated release rate of PO_4 from bottom waters, which is proportional to sediment oxygen demand, was reduced by 50 percent, and water-quality conditions were simulated. The reduction in the bottom-water PO_4 release rate resulted in only a fairly small reduction in simulated algal concentrations. The greatest effects were in the spring when contributions of PO_4 from bottom waters are small.

As previously indicated, PO_4 concentrations in the point-source discharge were assumed to be zero for the model calibration simulations. This was because compliance monitoring data from the facility did not include information on PO_4 concentrations, although monthly TP concentrations were reported. Reported monthly TP concentrations from March 1993 through March 1994 varied from 0.67 to 7.91 mg/L.

The effects of changes in the quantity, quality, and location of the mid-reservoir point-source discharge were simulated. The discharge point was assumed to be located at an elevation of 294.0 m, or a depth of about 9 m. For the initial simulation, PO_4 concentrations were assumed to be equal to 50 percent of the TP concentrations and a reported flow rate of 0.21 m^3/s. Simulated near-surface algal concentrations in model segment 21 (site 24) differed by less than 5 percent from concentrations simulated assuming no PO_4 in the point-source discharge (figs. 34 and 42). Additional simulations were conducted with (1) PO_4 concentrations equal to reported TP concentrations and a flow of 0.21 m^3/s, and (2) PO_4 concentrations equal to reported TP concentrations and a flow of 0.5 m^3/s. Changes in the PO_4 concentrations and flow rate had very little effect on simulated algal concentrations. However, DO concentrations were affected somewhat. For the most extreme scenario (PO_4 equal to TP

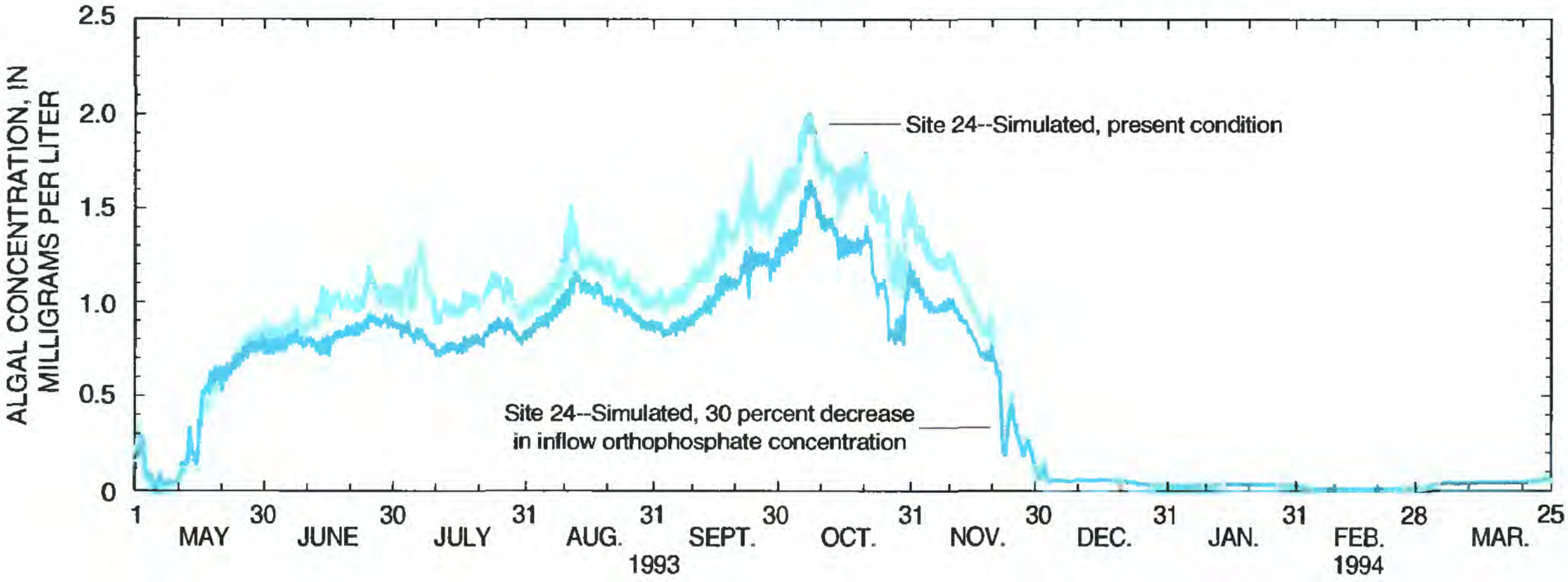

Figure 41. Simulated algal concentrations at site 24, May 1, 1993, through March 25, 1994, using estimated orthophosphate inflows and estimated orthophosphate inflows reduced by 30 percent.

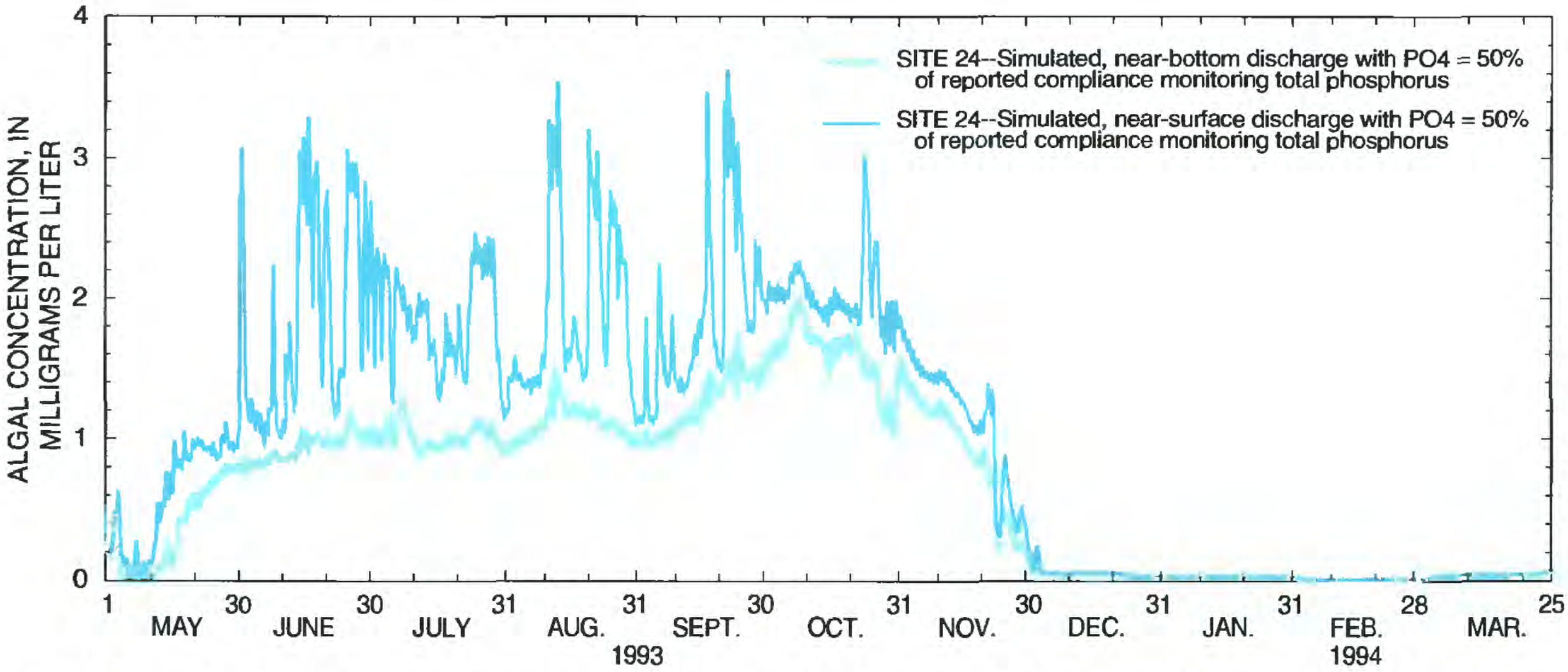

Figure 42. Simulated algal concentrations for near-bottom (present condition) and near-surface point-source discharge at site 24, May 1, 1993, through March 25, 1994.

concentrations, and flow = 0.5 m^3/s) simulated near-bottom DO concentrations were lower than those simulated, assuming no PO_4 in the point-source discharge (base condition). For example, at site 26 DO was less than 4.9 mg/L about 40 percent of the time; for the most extreme point-source dischage condition DO was less than about 4.1 mg/L about 40 percent of the time.

A final simulation was performed by assuming that the point-source discharge was located about 1 m below the water surface, PO_4 concentrations were equal to 50 percent of reported TP concentrations, and the point-source discharge was 0.21 m^3/s. The delivery of the PO_4 directly to the photic zone had a dramatic effect on simulated algal concentrations. Simulated near-surface algal concentrations at site 24 were as much as three times greater than those simulated for the base condition. Because the discharge is located in the hypolimnion and the reservoir is stratified during much of the growing season, much of the PO_4 from the point-source discharge remains in hyolimnion and does not directly affect algal growth. Specific conductance measurements made during the summer near the location of the point-source, indicate that the waste-water plume remains several meters below the water surface (fig. 8). However, the long-term effects of the discharge on reservoir water quality are not known. As previously shown, algal growth in Lake Rhodhiss is stimulated by the release of PO_4 from bottom sediments.

CONCLUSIONS

From January 1993 through March 1994, ambient water-quality conditions in Rhodhiss Lake varied spatially and seasonally. Distributions of temperature revealed dynamic patterns of water circulation and material transport in the reservoir. Generally, the shallow upstream end of Rhodhiss Lake was unstratified and well oxygenated. This riverine zone was characterized by high turbidity and total suspended solids, and elevated concentrations of nitrate, orthophosphate, and total phosphorus. Concentrations of total suspended solids, nitrate, orthophosphate, and total phosphorus decreased in a downstream direction from the headwaters to the dam. An increase in specific conductance was frequently observed at mid-reservoir, usually at a depth of 4 to 6 m, downstream from a municipal WWTP outfall.

From about mid-reservoir to the dam, Rhodhiss Lake thermally stratified during the summer of 1993. During the summer, headwater inflows generally were cooler than surface waters downstream and, therefore, tended to sink and move as an interflow toward the dam. Fall and winter temperature distributions indicated multiple, alternating periods of stratification and mixing from mid-reservoir to the dam. Following the onset of thermal stratification in May 1993, dissolved oxygen in this reach was quickly depleted from bottom waters, and low concentrations persisted in the hypolimnion throughout the summer. Inorganic nutrients—nitrate, ammonia, and orthophosphate—were depleted from the epilimnion during the summer, probably by algal uptake. At the same time, ammonia concentrations increased in the hypolimnion. Concentrations of orthophosphate and total phosphorus also tended to be higher in bottom waters than in surface waters during the summer.

Based on nutrient concentrations, Rhodhiss Lake is classified eutrophic. However, nuisance levels of phytoplankton were rarely observed, possibly because short residence time and mixing patterns suppressed algal growth. Limited light as a result of high abiogenic turbidity also may have been a factor at the Huffman Bridge site. From May through September 1993, mean chlorophyll *a* concentrations were 1.4 μg/L at Huffman Bridge, 10.2 μg/L at mid-reservoir, and 7.9 μg/L near the dam. A maximum concentration of 52 μg/L was observed at mid-reservoir on November 17, 1993, and was the only value that exceeded the State water-quality standard for chlorophyll *a* of 40 μg/L.

Lower Creek had high specific conductance, high concentrations of total suspended solids, and was nutrient enriched. Fecal coliform concentrations exceeded 200 cols/100 ml in 76 percent of the samples. The two highest values occurred when streamflow was elevated, but values in excess of 200 cols/100 ml were observed across a wide range of flow conditions. Concentrations of fecal coliform bacteria greater than 200 cols/100 ml also were observed in Rhodhiss Lake—37 percent of the time in the headwaters and 16 percent of the time at mid-reservoir and in the forebay. Maximum concentrations at reservoir sites occurred in conjunction with heavy rainfall in March 1993.

Loadings of total suspended solids, nitrogen, and phosphorus were calculated for Lower Creek site 53, Rhodhiss Lake at Huffman Bridge (site 20), and for selected point-source discharges. Results indicated that almost all of the suspended solids and the majority of the nitrogen and phosphorus entering the headwaters of the reservoir originated from nonpoint sources. Approximately 26 percent of the suspended solids load, 21 percent of the total phosphorus load, and 6 percent of the total nitrogen load to the upper end of Rhodhiss Lake occurred during one storm event during March 23-25, 1993.

Nonetheless, point sources contributed significant amounts of nutrients. Seven point sources accounted for more than 99 percent of the permitted wastewater flow in the Rhodhiss Lake watershed downstream from Lake James. Five of these point sources were located upstream from Huffman Bridge and contributed 27 percent of the total nitrogen load and 22 percent of the total phosphorus load. One of the remaining point sources, a municipal WWTP, added approximately 80,900 kg of nitrogen and 30,500 kg of phosphorus to the mid-section of Rhodhiss Lake during the study period.

A hydrodynamic and water-quality computer model was used to simulate flow, transport, and water-quality conditions in Rhodhiss Lake for 1993-94 conditions, and for other selected hypothetical conditions. The model domain extended from Huffman Bridge to Rhodhiss Dam, or a distance of 18.5 km, and included five embayments or coves. There were 37 computational segments along the mainstem of the reservoir, and each cove was represented by three segments. All segments are 500 m long and 1 m thick. Segment widths were chosen to represent the longitudinal and vertical width variations in the reservoir and to properly represent the reservoir volume, which was estimated to be about 59 million m^3.

Model simulations were made using data from April 1, 1993, through March 25, 1994. Model data included (1) a record of hourly flows at Huffman

Bridge and at Rhodhiss Dam, (2) estimated daily mean inflows for the five embayments, (3) hourly water temperature at Huffman Bridge, (4) daily mean water temperature for the five embayments, (5) hourly meteorological conditions (air temperature, dewpoint temperature, cloud cover, and wind speed and direction), (6) water-supply withdrawal rates, (7) wastewater discharge rates, and (8) concentrations of selected constituents in the inflows. The model includes five parameters governing hydrodynamics and heat transport, and 57 chemical kinetic rate coefficients.

There was good agreement between measured and simulated water levels at Rhodhiss Dam. The root mean square difference between measured and simulated water levels was 0.085 m, the maximum positive (water level under-predicted) difference was 0.15 m, and the maximum negative difference was 0.38 m. Eighty percent of the differences between measured and simulated water levels were between 0.02 and 0.12 m. The total range in measured water level during the simulation period was 1.32 m.

At the mid-reservoir measurement site, water temperature during April 1993 through March 1994 ranged from about 4 to about 30 °C. The difference between each of the measured and corresponding simulated water temperatures was computed for the mid-reservoir site. The mean difference was -0.24 °C; 80 percent of the differences were between 1.26 and -1.80 °C; and 95 percent of the differences were between 1.7 and -2.4 °C. All of the results from the water temperature simulations suggest that (1) the calibrated model provides a reasonable simulation of water temperature in Rhodhiss Lake, with most of the simulated values within about 2 °C of the actual value; and (2) near-bottom water temperature, particularly in the deeper parts of the reservoir, is typically over-predicted from 1 to 3 °C during the warmer months. The simulated near-surface water temperatures generally agree more closely with corresponding measured values than simulated near-bottom water temperatures. This, along with the simulated vertical profiles of water temperature, suggest that the model over-predicts vertical mixing, resulting in excessive transport of warmer surface waters to the cooler, deeper waters of the reservoir.

Eleven water-chemistry constituents were included in the Rhodhiss Lake model. Analysis of results focused on DO, algae, and PO_4 because ambient water-quality standards exit for DO and algae, and also because of the importance of PO_4 in stimulating phytoplankton growth. The difference between each measured DO value at the mid-reservoir measurement site and the simulated DO at the corresponding time and depth was calculated. These differences were compared to the corresponding measured DO concentrations, the measurement date, and the depth of measurement. Measured and simulated DO closely agree when the ambient DO concentrations are greater than about 8 mg/L. When the measured DO is between about 5 and 8 mg/L, the simulated DO is typically greater than the corresponding value. Differences between measured and simulated DO for measured concentrations less than about 5 mg/L vary, but differences are likely due to poor simulation of the timing of events. Most of the over-predictions at the mid-reservoir site were at mid-depth, or depths between about 2 and 7 m below the water surface. These are the depths where the DO values of between 5 and 8 mg/L, which were over-simulated, most often occur.

The cumulative frequency of occurrence of measured DO concentrations at the mid-reservoir site was determined and a similar analysis was performed using the simulated DO values which correspond to times at which DO measurements were made. The frequency of occurrence of DO concentrations less than 5 mg/L was almost the same for measured and simulated DO. For DO concentrations between about 5 and 12 mg/L, simulated DO values were greater than measured DO values. For example, about 50 percent of the time simulated DO values were less than or equal to about 9.5 mg/L; but measured DO values were less than or equal to about 8.5 mg/L 50 percent of the time. Generally, regulators and resource managers are most interested in the occurrence of DO concentrations less than 5 mg/L as opposed to higher values.

The calibrated model provides a reasonable simulation of DO concentration in Rhodhiss Lake. Near-surface and near-bottom DO appears to be predicted better than DO concentrations at mid-depth, where DO typically is over-predicted. There is good agreement between simulated and measured frequency of occurrence of DO concentrations less than 5 mg/L in the reservoir; simulation of the exact timing of the low DO events appears to be within about 5 days of the actual occurrence. The simulated DO profiles do not always exhibit the complex variations of DO with depth which are seen in the measurements.

The calibrated Rhodhiss Lake model was applied using 1993-94 boundary data to simulate the movement of a neutrally buoyant, conservative tracer through the reservoir. Results from these simulations demonstrate (1) the use of the Rhodhiss Lake model in evaluating the movement of a short-term or long-term release into the reservoir; (2) the manner in which nonconservative materials move through the reservoir, without the confounding effects of chemical transformations, regeneration, and settling; (3) the difficulty in identifying a single residence time for the reservoir—"residence times" vary seasonally, as well as with depth; and (4) the effects of density stratification on transport processes.

The effects of increased air temperature and wind speed on DO concentrations were evaluated using the model. The warmer temperatures slightly increased the amount of time that simulated DO concentrations were below 4 mg/L throughout the reservoir, but increased wind speed had little effect on DO concentrations (probably because of interflow). Simulations were made of the effects on DO concentrations and algae as a result of changes in PO_4 loadings from Catawba River inflows (nonpoint sources) and from bottom sediments. A 30-percent reduction in inflow concentrations resulted in about a 20-percent decrease in the maximum algal concentration. A 50-percent reduction in the PO_4 release rate from bottom sediments resulted in only a small reduction in algal concentrations, primarily in the spring.

The effects of changes in point-source discharge concentrations of PO_4 were simulated. Algal concentrations were affected only slightly; but when the discharge rate was doubled and the PO_4 concentration was equal to the measured total phosphorus concentration in the discharge, DO concentrations in the reservoir were reduced. When the point-source discharge was moved from about 9 m below the water surface to 1 m below the water surface, simulated algal concentrations were 2 to 3 times greater than for the deep discharge. This was a result of the increased deliveries of PO_4 to the photic zone.

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