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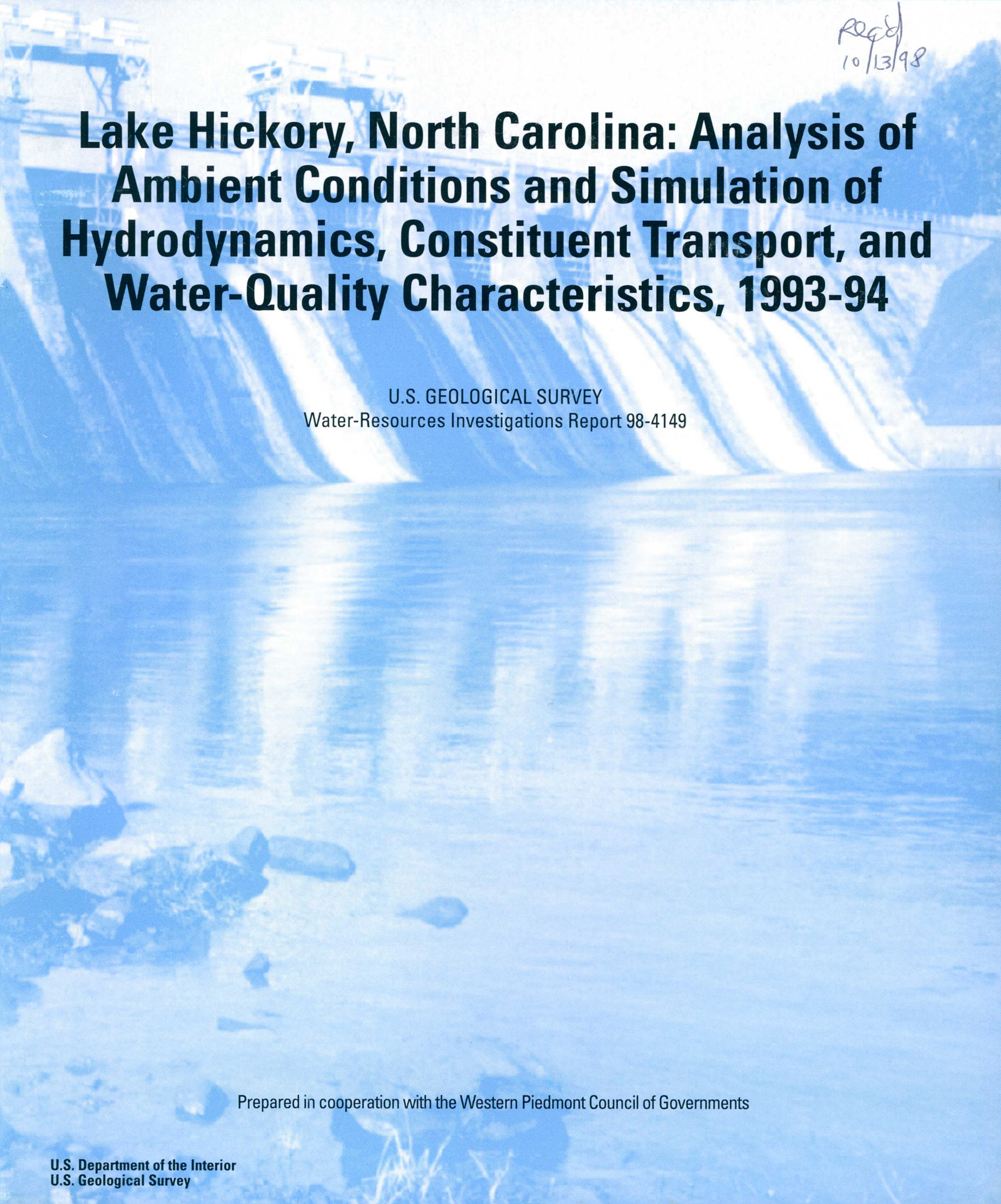

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Lake Hickory, North Carolina: Analysis of Ambient Conditions and Simulation of Hydrodynamics, Constituent Transport, and Water-Quality Characteristics, 1993-94

U.S. GEOLOGICAL SURVEY
Water-Resources Investigations Report 98-4149

Prepared in cooperation with the Western Piedmont Council of Governments

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

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By Jerad D. Bales and Mary J. Giorgino

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U.S. DEPARTMENT OF THE INTERIOR
MARCH 3, 1849

Raleigh, North Carolina
1998

U.S. DEPARTMENT OF THE INTERIOR
BRUCE BABBITT, Secretary

U.S. GEOLOGICAL SURVEY
Thomas J. Casadevall, Acting Director

For additional information write to:

District Chief
U.S. Geological Survey
3916 Sunset Ridge Road
Raleigh, NC 27607

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CONVERSION FACTORS, TEMPERATURE, VERTICAL DATUM, AND DEFINITIONS

Multiply	By	To obtain
	Length	
centimeter (cm)	0.394	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

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

$$°F = (1.8 \times °C) + 32$$

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 (µS/cm) at 25 degrees Celsius.

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

Fecal coliform concentrations are given in colonies per 100 milliliters (cols./100 mL).

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

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

ABSTRACT

From January 1993 through March 1994, circulation patterns and water-quality characteristics in Lake Hickory varied seasonally and were strongly influenced by inflows from Rhodhiss Dam. The upper, riverine portion of Lake Hickory was unstratified during much of the study period. Downstream from the headwaters to Oxford Dam, Lake Hickory thermally stratified during the summer of 1993. During stratification, releases from Rhodhiss Dam plunged beneath the warmer surface waters of Lake Hickory and moved through the reservoir as interflow. During fall and winter, Lake Hickory was characterized by alternating periods of mixing and weak stratification.

Water-quality conditions in the headwaters of Lake Hickory were largely driven by conditions in water being released from Rhodhiss Dam. In general, water clarity increased, and concentrations of suspended solids, phosphorus, and summertime chlorophyll *a* decreased in a downstream direction from the headwaters of Lake Hickory to Oxford Dam. Two chlorophyll *a* samples from the upper portion of Lake Hickory exceeded the North Carolina water-quality standard of 40 micrograms per liter during the investigation. Downstream from the headwaters, dissolved oxygen was rapidly depleted from Lake Hickory bottom waters beginning in May 1993, and anoxic conditions persisted in the hypolimnion throughout the summer. During summer stratification, concentrations of nitrite plus nitrate, ammonia, and orthophosphate were low in the epilimnion, but concentrations of ammonia near the bottom of the reservoir increased as the hypolimnion became anoxic.

Concentrations of fecal coliform bacteria exceeded 200 colonies per 100 milliliters in only one of 60 samples collected from Lake Hickory. In contrast, concentrations of fecal coliform bacteria exceeded 200 colonies per 100 milliliters in 40 percent of samples collected from the Upper Little River, and in 60 percent of samples collected from the Middle Little River, two tributaries to Lake Hickory.

Load estimates for the period April 1993 through March 1994 indicated that releases from Rhodhiss Dam accounted for most of the suspended solids, nitrogen, and phosphorus entering the headwaters of Lake Hickory. Loads of nitrogen and phosphorus from point-source discharges were potentially important, but loads of suspended solids from these discharges were insignificant relative to other sources.

The CE-QUAL-W2 model was applied to Lake Hickory from the U.S. Highway 321 bridge to Oxford Dam—a distance of 22 kilometers—and was calibrated by using data collected from April 1993 through March 1994. During the simulation period, measured water levels varied a total of 1.14 meters, and water temperatures ranged from 4 to 31 degrees Celsius. The calibrated model provided good agreement between measured and simulated water levels at Oxford Dam. Likewise,

simulated water temperatures were generally within 1 degree Celsius of measured values; however, water temperatures were oversimulated for the fall of 1993. Simulated dissolved oxygen concentrations generally agreed with measurements; however, the model tended to oversimulate dissolved oxygen concentrations during the late summer and early fall. There was good agreement between simulated and measured frequency of occurrence of dissolved oxygen concentrations less than 4 milligrams per liter.

Simulations of tracer dye releases demonstrated the effects of stratification on dilution and rate of transport in Lake Hickory. Simulations were made of the effects of changes in nutrient loads from inflows and from bottom sediments. A simulated 30-percent reduction in inflow concentrations of orthophosphate, ammonia, and nitrate at the U.S. Highway 321 bridge delayed the initial springtime pulse of algal growth by about 2 weeks, but had little effect on dissolved oxygen concentrations. Likewise, a reduction in the release rate of orthophosphate and ammonia from bottom sediments had very little effect on simulated algae and dissolved oxygen concentrations. To simulate the effects of shoreline development on Lake Hickory water quality, concentrations of orthophosphate, ammonia, and nitrate in Snow Creek were increased to levels representative of streams draining developed areas in Charlotte, N.C. Maximum simulated algae concentrations increased about 100 percent relative to current conditions, indicating that Lake Hickory may be sensitive to increased nutrient loads into the middle reaches of the reservoir.

INTRODUCTION

Lake Hickory (fig. 1) was constructed in 1925 to supply hydroelectric power for a growing population and textile industry in the upper Catawba River Basin. 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 circulation and constituent transport in the reservoir is lacking. This information is needed to more effectively manage the reservoir and its watershed and to predict water-quality responses to changes in constituent loading or hydrologic regime. Such changes may result from State watershed-protection regulations, proposed wastewater-treatment plant expansions in the Lake

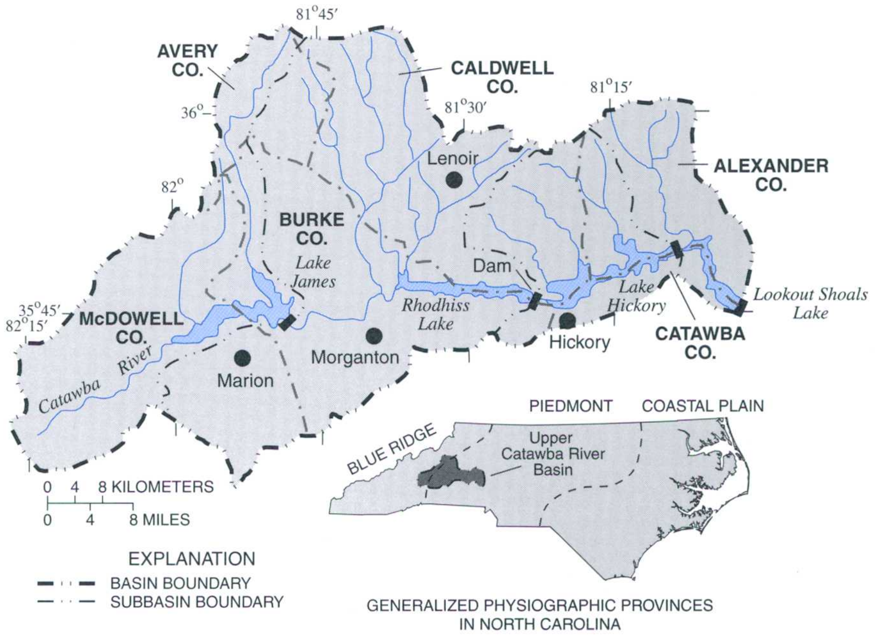


Figure 1. Location of Lake Hickory in the upper Catawba River Basin of North Carolina.

Hickory watershed, and a new basinwide approach used by State regulators to permit point-source discharges. Loads of nonpoint-source pollutants could change as a result of implementing forestry and agricultural best-management practices, the continued administration of a statewide sedimentation control program, and increasing urban development.

In January 1993, an investigation was undertaken to characterize circulation and water-quality conditions in Lake Hickory and to simulate the effects of changes in external loads to the reservoir. 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, N.C., to monitor water levels, streamflow, and water quality 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 (Jaynes, 1994; Giorgino and Bales, 1997). The Western Piedmont Council 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.

Purpose and Scope

Objectives of this investigation were to (1) characterize ambient hydrologic and water-quality conditions in Lake Hickory and two of its tributaries—the Upper Little River and the Middle Little River; (2) calibrate the CE-QUAL-W2 model for Lake Hickory; and (3) run several model simulations. This report describes ambient conditions and presents the results of model calibration and simulations. Conditions and model results for Lake Hickory are compared to results obtained for Rhodhiss Lake.

Results are summarized for the period January 1993 through March 1994 for selected water-quality characteristics including temperature, DO, pH, specific conductance, suspended solids, nitrogen and phosphorus species, chlorophyll *a*, and fecal coliform bacteria. Temporal and spatial patterns of water quality in Lake Hickory are described and, when applicable, conditions are compared to prevailing water-quality standards. Loads of total suspended solids, nitrogen, and phosphorus to the Upper Little River and the Middle Little River are presented, as are estimated loads from selected point-source discharges in the Lake Hickory watershed.

Hydraulic circulation and water-quality characteristics in Lake Hickory were simulated by using the U.S. Army Corps of Engineers CE-QUAL-W2 model. This laterally averaged, two-dimensional model was calibrated by using data collected from April 1993 through March 1994. The calibrated model was used to simulate the effects of changes in nutrient loads on reservoir water quality.

Description of Study Area

The upper Catawba River Basin extends from the Blue Ridge Province into the Piedmont Province of North Carolina (fig. 1). The basin 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).

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 (fig. 1).

Lake Hickory is a narrow, riverine reservoir (fig. 2) with a surface area of 14 square kilometers (km^2) and a maximum depth of 26 meters (m). The predominant inflow to Lake Hickory is water released through a sub-surface withdrawal from Rhodhiss Dam, which is regulated by Duke Power Company. Other major tributaries include Gunpowder Creek, the Upper Little River, and the Middle Little River, all of which drain areas north of the reservoir (fig. 2). Numerous smaller tributaries drain areas to the south (fig. 2). The Lake Hickory watershed is rolling to hilly, with forested areas, agricultural pastures and cropland, and urban, residential, and industrial areas concentrated around the city of Hickory (fig. 1). The shoreline of Lake Hickory is popular for residential development.

Lake Hickory supplies drinking water to Hickory and Longview, and is used by Duke Power Company to generate hydroelectric power. The reservoir receives treated effluent from several municipalities, industries, schools, and other facilities, along with runoff from urban and agricultural areas. Water-quality problems such as algal blooms and elevated concentrations of

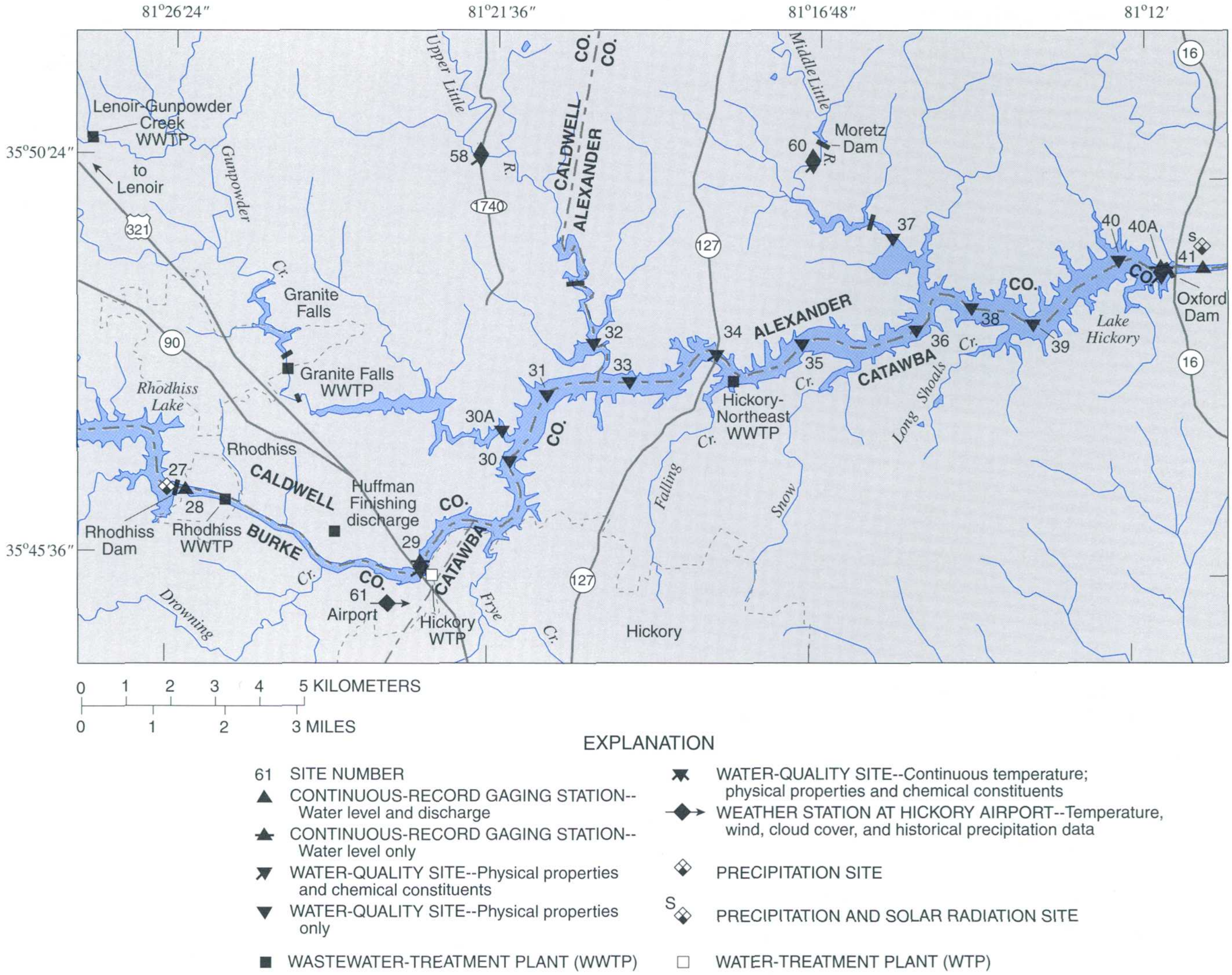


Figure 2. Locations of data-collection sites and selected point-source discharges in the Lake Hickory study area, 1993–94.

chlorophyll *a* and phosphorus have been observed in Lake Hickory (North Carolina Department of Environment, Health, and Natural Resources, 1994a).

Previous Studies

Although water-quality conditions in Lake Hickory 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. Numerous problems were observed in streams draining urban areas of the Lake Hickory watershed during this survey, including Drowning, Gunpowder, Frye, and Falling Creeks (fig. 2; North Carolina Department of Water Resources, 1961). Domestic sewage from municipal, industrial, and private outfalls depleted dissolved oxygen (DO) in the streams and increased concentrations of fecal coliform bacteria to millions of colonies per 100 milliliters (mL) of water. Industrial discharges contributed additional oxygen-demanding wastes, bacteria, and toxic constituents, such as copper and phenols, to streams near the city of Hickory and the town of Rhodhiss (figs. 1 and 2). Wastewater discharges into Gunpowder Creek caused heavy algal growth in that creek's impoundments, occasionally resulting in supersaturated dissolved oxygen concentrations (fig. 2). Similar water-quality problems were observed in other urban-industrial areas of the Catawba River Basin during the late 1950's before many technological advances in wastewater treatment and water-quality protection measures were implemented (North Carolina Department of Water Resources, 1961).

Wilder and Slack (1971a, b) reported that concentrations of nitrate and chloride in Middle Little River and Drowning Creek were similar to natural levels, but concentrations of nitrate in Gunpowder Creek were slightly elevated. The North Carolina Department of Environment, Health, and Natural Resources (1994a, b) reported that Upper Little River and Middle Little River fully supported their designated uses and received "Good" water-quality ratings based on analysis of benthic macroinvertebrates. A "Poor" rating was given to a small tributary receiving toxic effluent from a textile finishing plant. This tributary drains into the mainstem of Lake Hickory upstream from the U.S. Highway 321 bridge.

Water-quality investigations of unpolluted streams near Lake Hickory 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 Lake Hickory drainage area, 16 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, and trace elements (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 Lake Hickory primarily focused on classifying the trophic status of the reservoir. Since the early 1970's, investigators have described Lake Hickory 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, 1994a) 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 and dominance of blue-green algae consistently occur in Lake Hickory.

Acknowledgments

Sandra Gillaspie and Diane Wilburn of the North Carolina Division of Water Quality supplied compliance-monitoring data used to estimate loads 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 Lake Hickory which aided model calibration. Tom Cole of the U.S. Army Corps of Engineers provided valuable guidance on the CE-QUAL-W2 model.

METHODS OF DATA COLLECTION

Data collection began in January 1993 and continued through March 1994. Hydrologic measurements were made at the reservoir headwaters and forebay and in the Upper Little River and the Middle Little River (fig. 2; table 1). Physical water-quality properties were measured at 14 sites in Lake Hickory and at the two tributary sites. Water samples were collected for chemical characterization from three sites in the reservoir and from the Upper Little River and Middle Little River sites. Field measurements 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 reported by Jaynes (1994).

Continuous-stage gages recorded water level every 15 minutes at Lake Hickory site 29, Upper Little River site 58, and Middle Little River site 60 (fig. 2). At sites 58 and 60, streamflow (or discharge) was computed by using stage-discharge relations developed during the investigation. Hourly inflow to Lake Hickory from Rhodhiss Dam was calculated from water-release records provided by Duke Power Company (site 28, fig. 2). Duke Power Company also provided hourly records of water levels in the forebay

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

[Physical water-quality properties and chemical constituents are listed in text. —, data not collected for the Lake Hickory investigation; x, data collected]

Site number (fig. 2)	Site location and USGS identification number	Type of hydrologic measurement		Type of water-quality data		Type of meteorologic data
		Stage	Discharge	Physical	Chemical	
27	Rhodhiss Lake at Rhodhiss Dam, 0214148975	—	—	—	—	Precipitation[a]
28	Lake Hickory below Rhodhiss Dam, 0214149005	—	x[a]	—	—	
29	Lake Hickory at U.S. Highway 321, 0214157625	x	—	x	x	
30	Lake Hickory above Gunpowder Creek, 0214179775	—	—	x	—	
30A	Gunpowder Creek at Secondary Road 1751, 0214179800	—	—	x	—	
31	Lake Hickory above Upper Little River, 0214179810	—	—	x	—	
32	Lake Hickory, Upper Little River arm 0.6 miles above mouth, 0214183410	—	—	x	—	
33	Lake Hickory below Upper Little River, 0214183550	—	—	x	—	
34	Lake Hickory at N.C. Highway 127, 0214184000	—	—	x	x	
35	Lake Hickory, 0214187210	—	—	x	—	
36	Lake Hickory, 0214187285	—	—	x	—	
37	Lake Hickory, Middle Little River arm below Secondary Road 1137, 0214192595	—	—	x	—	
38	Lake Hickory, 0214193528	—	—	x	—	
39	Lake Hickory, 0214193585	—	—	x	—	
40	Lake Hickory, 0214193597	—	—	x	—	
40A	Lake Hickory above Oxford Dam, 0214196095	x[a]	—	x	x	Precipitation[a], air temperature, solar radiation
41	Catawba River below Oxford Dam, 0214196105	—	x[a]	—	—	
58	Upper Little River at Secondary Road 1740 at Petra Mills, 0214183365	x	x	x	x	
60	Middle Little River at Moretz Dam near Bethlehem, 0214192500	x	x	x	x	
61	Hickory Airport, FAA weather station, 354426081233201	—	—	—	—	Air temperature, wind speed and direction, cloud cover, historical precipitation

[a]Measured by Duke Power Company.

of Lake Hickory (site 40A, fig. 2) and releases from Oxford Dam (site 41, fig. 2).

Temperature, DO, specific conductance, and pH were measured in the field with a multiparameter water-quality meter. At the two tributary sites, measurements were made 0.1 m beneath the water surface. At the 14 reservoir sites, measurements were made at 1- or 2-m vertical intervals throughout the water column. Water transparency was measured at reservoir sites with a Secchi disk. In addition, automated monitors recorded water temperature every 15 minutes at headwater site 29 at three depths—near the surface, at mid-depth, and near the bottom.

To obtain representative samples at Upper Little River site 58 and Middle Little River site 60, water was collected from several locations across the stream by using a depth-integrating sampler (Edwards and Glysson, 1988). These 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), carbonaceous biochemical oxygen demand (BOD), hardness, alkalinity, iron, calcium, and magnesium. In addition, grab (point) samples for fecal coliform bacteria were collected by hand-dipping or by using a weighted bottle holder at mid-channel.

The three water-chemistry sites in Lake Hickory represented headwater, mid-reservoir, and forebay areas of the reservoir (sites 29, 34, and 40A, 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 total organic carbon analysis were collected at the same depths by 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 29, and thermocline samples were collected only five times at site 34 and six times at site 40A. Thermocline data were reported by Jaynes (1994) and were used as input to the water-quality model for Lake Hickory; however, these data 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.

Samples for fecal coliform bacteria were analyzed by the City of Hickory Water Filtration Plant by 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, N.C., by 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, Colo. 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). The laboratory reporting level for total organic nitrogen plus ammonia was 0.2 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).

Although not discussed at length in this report, meteorological data needed to calibrate the CE-QUAL-W2 model also were collected. Daily totals of precipitation at Rhodhiss and Oxford Dams were provided by Duke Power Company (sites 27 and 40A, fig. 2). The USGS measured air temperature and solar radiation at Oxford Dam (site 40A, fig. 2). Additional meteorological data including air temperature, dewpoint temperature, wind speed and direction, cloud cover, and historical precipitation were obtained from a Federal Aviation Administration weather station located at the Hickory Airport (site 61, fig. 2).

AMBIENT CONDITIONS

Hydrologic and water-quality conditions are summarized in this section. Hydrologic data include precipitation records, records of inflow from Rhodhiss Dam and outflow from Oxford Dam, and measurements of water level and streamflow from Upper Little River and Middle Little River. Summarized water-quality conditions include measurements of temperature, DO, pH, and specific conductance, and concentrations of selected chemical and biological constituents. Loads of suspended solids and nutrient species in the headwaters of Lake Hickory, the Upper Little River, the Middle Little River, and from selected point-source discharges 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 this investigation, monthly precipitation ranged from 1.42 cm during June 1993 to 21.39 cm during March 1993 (fig. 3). In general, 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.

Flow in the headwaters of Lake Hickory was dominated by upstream releases from Rhodhiss Dam

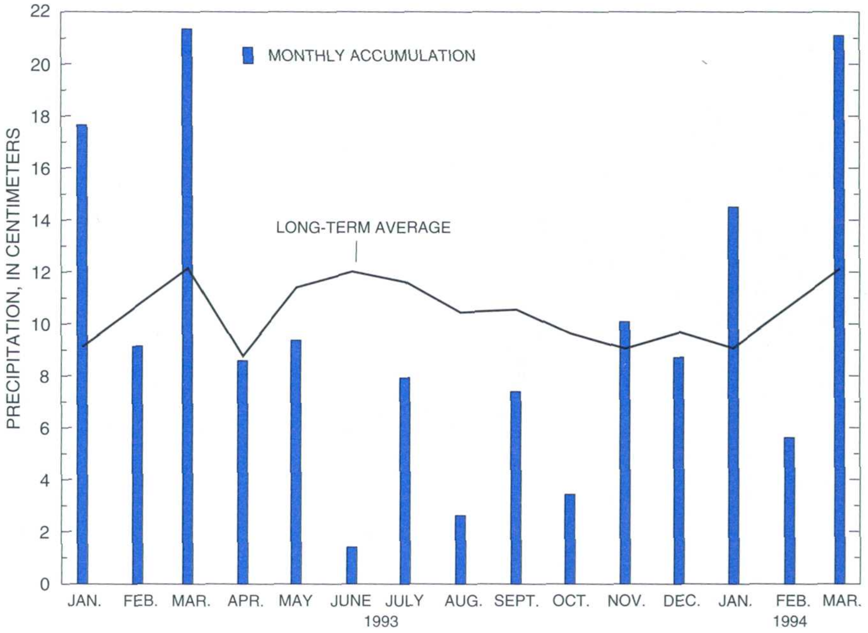


Figure 3. Relation of monthly precipitation from January 1993 through March 1994 with long-term average precipitation near Hickory, N.C.

(site 28, fig. 2). Daily mean discharges from Rhodhiss Dam ranged from 1.7 to 209 cubic meters per second (m^3/s) with a median of 59 m^3/s. Peak releases occurred during winter and spring and were especially high after rainfalls in March 1993 and March 1994 (figs. 3 and 4A). In general, releases were lower during the summer and fall of 1993. Patterns of release were affected not only by precipitation, but also by operations at the Rhodhiss hydropower plant. The lowest discharges occurred on weekends when power demands were low.

Hydrologic data indicate that Lake Hickory generally is characterized by short hydraulic retention times. Bathymetric measurements made by the USGS in June 1994 indicate that the current full-pool volume of Lake Hickory, 128 million cubic meters (m^3), is less than the 157 million m^3 previously reported by the U.S. Environmental Protection Agency (1975). Theoretical hydraulic retention times for Lake Hickory were estimated by dividing reservoir volume by daily outflows from Oxford Dam. From January 1993 through March 1994, retention times varied from 3.8 to 65 days and averaged 19 days.

Flow at Upper Little River site 58 and Middle Little River site 60 followed precipitation patterns, peaking after heavy rains in March 1993 and March 1994 (figs. 3, 4B and C). Flows were low in both tributaries during the summer and fall of 1993 (fig. 4). Daily mean discharge ranged from 0.48 to 28 m^3/s at Upper Little River site 58, and from 0.76 to 24 m^3/s at Middle Little River site 60.

Water-Quality Conditions

Temperature, DO, pH, and specific conductance were measured at 16 sites in the Lake Hickory study area, including 11 sites along the reservoir mainstem, three in reservoir coves, one in the Upper Little River, and one in the Middle Little River. Discussion of these four properties focuses on characteristics at the 11 mainstem sites (sites 29, 30, 31, 33, 34, 35, 36, 38, 39, 40, and 40A, fig. 2) and to a lesser extent, at the two tributary sites (sites 58 and 60, fig. 2). Data from the 11 mainstem sites are presented graphically as depth-distance diagrams. 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 delineate regions with similar values. Results for four sampling dates illustrate seasonal patterns in temperature, DO, pH, and specific conductance.

Water samples were collected and analyzed for several chemical and biological constituents at five sites in the Lake Hickory study area—in the headwaters, mid-reservoir, and forebay of Lake

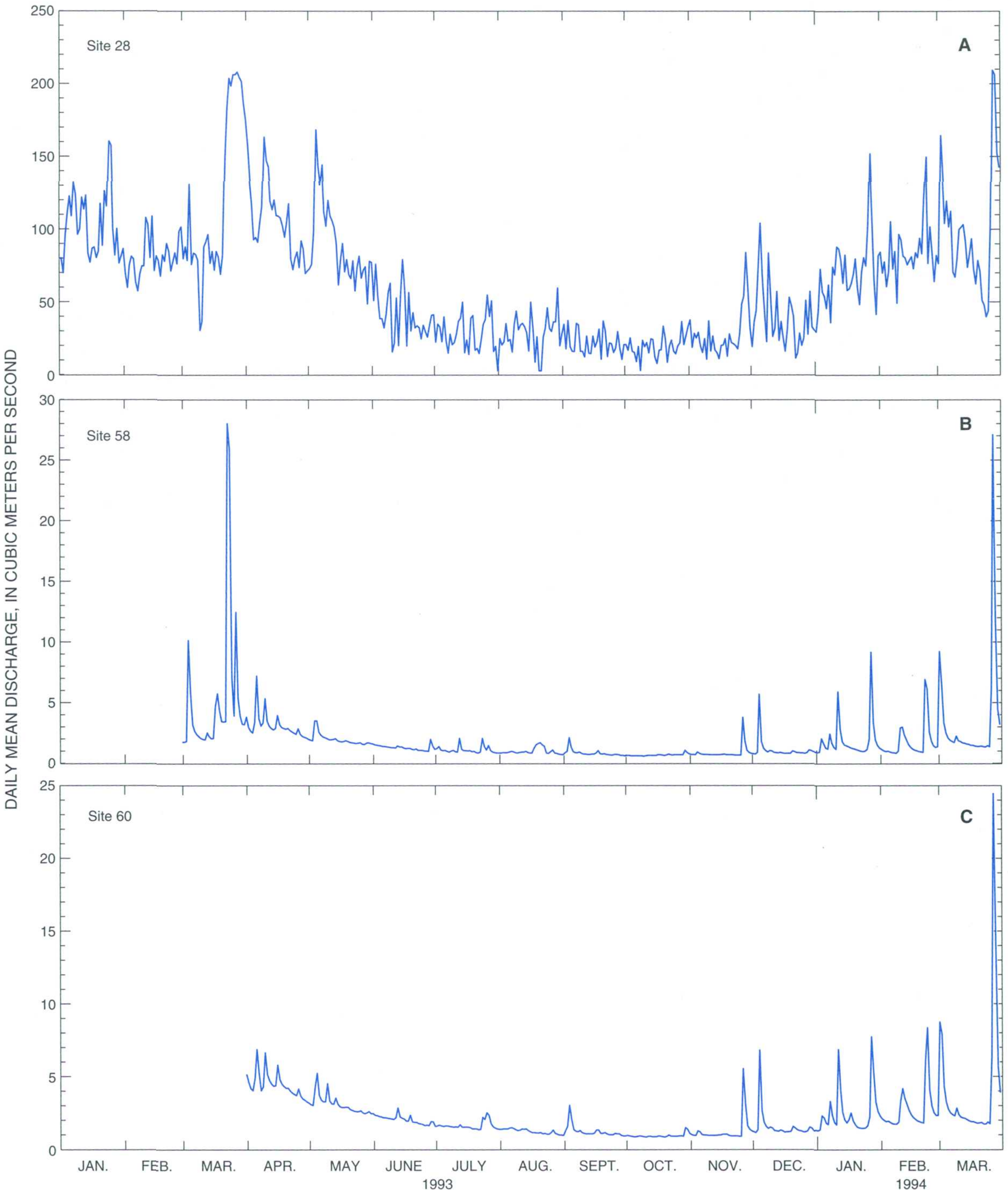


Figure 4. Daily mean discharge at (A) Lake Hickory site 28, (B) Upper Little River site 58, and (C) Middle Little River site 60, January 1993 through March 1994.

Hickory (sites 29, 34, and 40A, respectively, fig. 2), and in the Upper Little River and the Middle Little River (sites 58 and 60, respectively, fig. 2). In this section, results are summarized for suspended solids, nitrogen and phosphorus species, chlorophyll *a*, and fecal coliform bacteria. Analytical results for all constituents were reported by Jaynes (1994).

Temperature

Distributions of temperature varied both seasonally and spatially and indicated complex water-circulation patterns in Lake Hickory. In the riverine headwaters of Lake Hickory (site 29), daily mean temperatures at the surface and the bottom rarely differed by more than 1 degree Celsius (°C) except during June and July 1993 and exceeded 2 °C only four times during the study (Jaynes, 1994). Temperature distributions downstream from site 29 indicated that this part of Lake Hickory began to stratify in May 1993 and remained stratified through the summer (fig. 5). Stratification gradually weakened during the fall as surface temperatures cooled. Partial mixing occurred in late September 1993 when sites 29 to 36 were completely isothermal and sites 38 to 40A were isothermal to a depth of 20 m. Temperature measurements indicated that the reservoir underwent complete mixing by mid-December 1993 (fig. 5), and well mixed conditions were observed during monthly visits through February 1994. Lake Hickory began to restratify in March 1994.

Inflow to the headwaters of Lake Hickory primarily consisted of releases from Rhodhiss Dam. These releases were made through a sub-surface outlet that extends vertically from 1.5 to 12.5 m below the full-pool elevation of Rhodhiss Lake (Jon Knight, Duke Power Company, written commun., September 1994). During summer stratification, these sub-surface releases from Rhodhiss Dam were cooler and less oxygenated than surface waters in Lake Hickory. Temperature distributions indicate that this cooler inflow tended to sink beneath the warmer surface waters of Lake Hickory, and then continued downstream as interflow (see diagram for July 27, 1993, fig. 5). The interflow phenomenon was reinforced by sub-surface releases through Oxford Dam, where the withdrawal structure extends from 3.6 to 15.5 m below the full-pool elevation of Lake

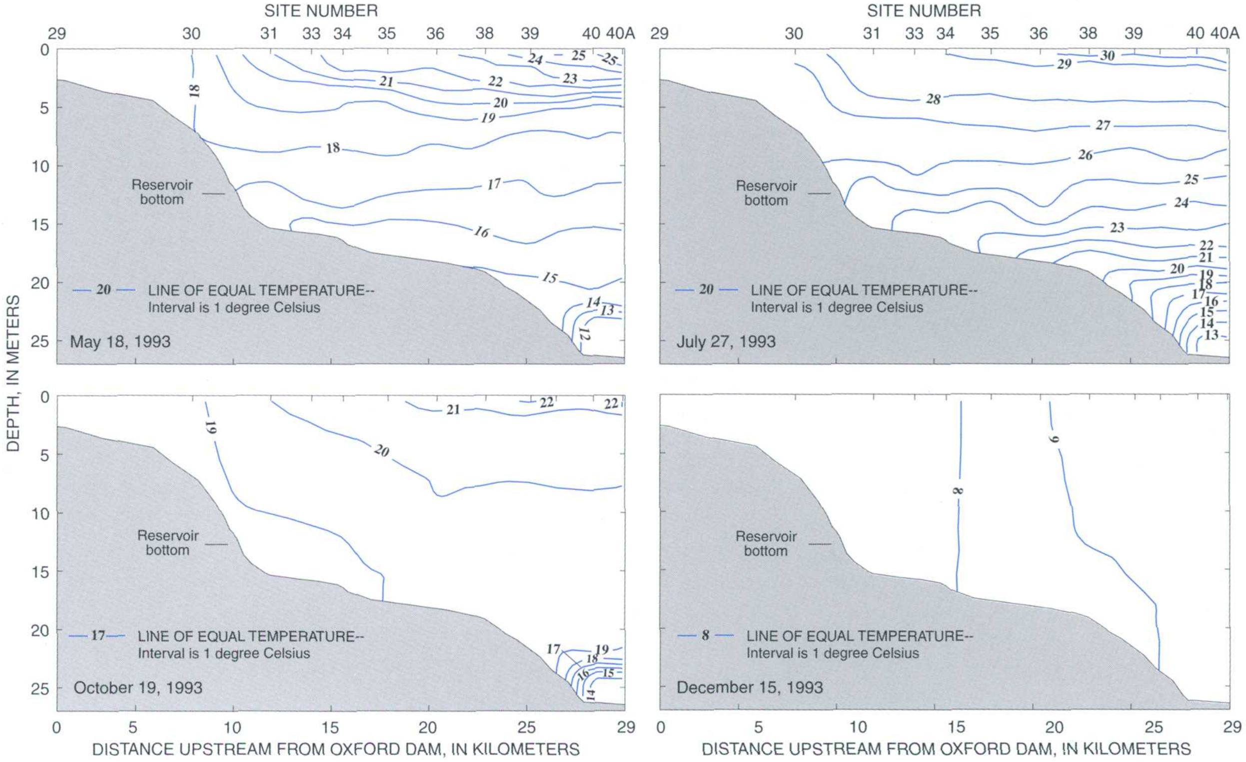


Figure 5. Water temperature in Lake Hickory for selected days during the study period (vertical scale exaggerated). Site locations shown in figure 2.

Hickory (Jon Knight, written commun., September 1994). As a result of these factors, summertime inflows from Rhodhiss Lake may have moved relatively quickly through Lake Hickory with minimal vertical mixing. This circulation pattern insulated the hypolimnion from the epilimnion and had important implications for the distribution of other water-quality constituents in Lake Hickory. A similar interflow pattern was observed in Rhodhiss Lake during the summer of 1993 (Giorgino and Bales, 1997).

Dissolved Oxygen

Dissolved oxygen concentrations in Lake Hickory were strongly influenced by DO concentrations in water released from Rhodhiss Lake and by temperature distributions in Lake Hickory. Throughout the study, concentrations of DO in the headwaters of Lake Hickory (site 29, fig. 2) were similar to concurrent concentrations in the forebay of Rhodhiss Lake at the depth of the withdrawal structure (Giorgino and Bales, 1997).

Dissolved oxygen in Lake Hickory was nearly saturated and was well mixed in the water column from January through April 1993. With the onset of persistent stratification in May 1993 (fig. 6), the effects of the interflow phenomenon described in the previous section became apparent. During summer stratification, DO concentrations at the surface were near or above saturation and abruptly decreased 3 to 4 m below the surface. From July through mid-September 1993, DO concentrations within the interflow generally were less than 5 mg/L and were comparable to sub-surface concentrations in the forebay of Rhodhiss Lake. At depths below the interflow, demands from hypolimnetic waters and bed sediments depleted oxygen from the hypolimnion. A large portion of Lake Hickory had DO concentrations less than 1.0 mg/L by July 1993 (fig. 6), and anoxic conditions persisted in the hypolimnion through mid-September 1993. Summertime anoxia was associated with increased specific conductance, increased concentrations of ammonia and orthophosphate, and decreased concentrations of nitrite plus nitrate in the hypolimnion, as described later in this report.

On September 28, 1993, most of Lake Hickory had DO concentrations less than 5.0 mg/L (fig. 6). During this partial mixing event, temperatures were

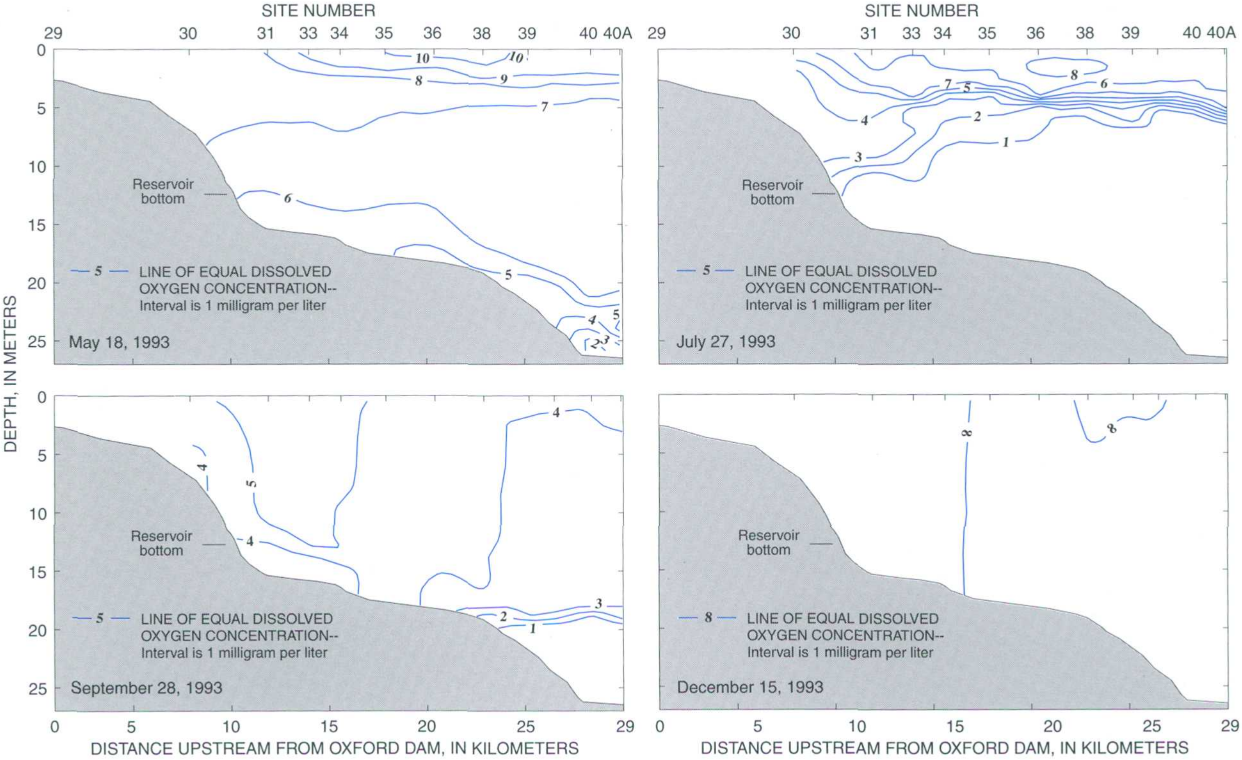


Figure 6. Dissolved oxygen concentrations in Lake Hickory for selected days during the study period (vertical scale exaggerated). Site locations shown in figure 2.

nearly isothermal down to about 20 m. The low DO concentrations were caused by oxygen-depleted hypolimnetic water mixing into the upper water column. From December 1993 through March 1994, dissolved oxygen returned to nearly saturated and well mixed conditions.

North Carolina water-quality standards require that surface 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. DO concentrations in the hypolimnion of Lake Hickory typically were less than 5.0 mg/L during summer stratification (fig. 6); however, low concentrations in reservoir bottom waters are not considered violations of the State DO standard if caused by natural conditions (North Carolina Department of Environment, Health, and Natural Resources, 1997). No violations of the State DO standard were observed at Upper Little River or Middle Little River, two tributaries to Lake Hickory. At these two sites (58 and 60, respectively), DO concentrations always were more than 90-percent saturated.

pH

During the study, pH values in Lake Hickory ranged from 6.1 to 8.8. During winter and spring of both 1993 and 1994, pH was fairly uniform throughout the reservoir. However, beginning in mid-May and continuing through mid-July 1993, pH was elevated and strongly stratified near the surface of Lake Hickory from site 33 downstream to the dam (fig. 7). High (alkaline) pH coincided with supersaturated DO concentrations, 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 conditions (Wetzel, 1983).

At Upper Little River site 58, pH ranged from 6.4 to 7.6 standard units with a median of 7.1. The pH at Middle River site 60 ranged from 6.7 to 7.5 standard units, and the median was 6.9. All pH measurements were within the range of 6.0 to 9.0 that the North Carolina Department of Environment, Health, and Natural Resources (1997) recommends to protect aquatic life and for general water use.

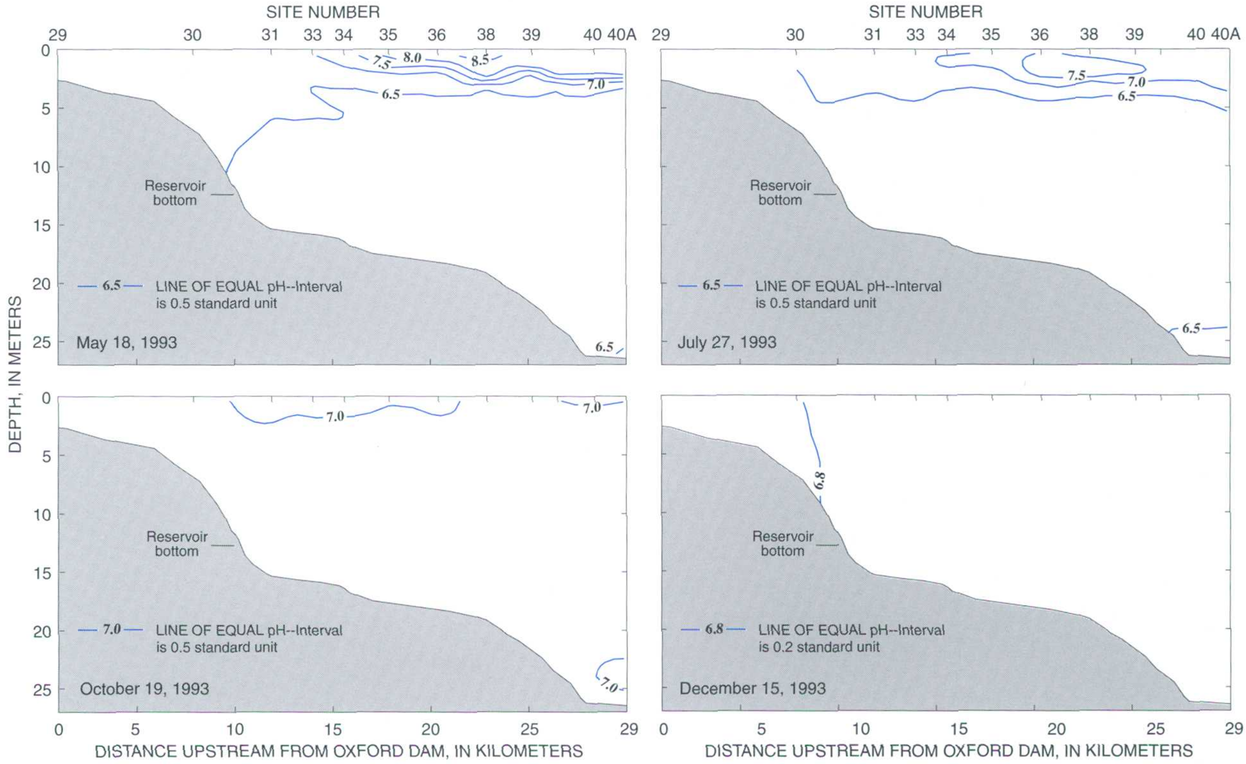


Figure 7. pH in Lake Hickory for selected days during the study period (vertical scale exaggerated). Site locations shown in figure 2.

Specific Conductance

Specific conductance at Upper Little River site 58 ranged from 23 to 31 microsiemens per centimeter (µS/cm) at 25 °C with a median of 28 µS/cm, and concentrations at Middle Little River site 60 ranged from 22 to 33 µS/cm with a median of 29 µS/cm. These levels were comparable to those observed in a minimally disturbed, forested watershed near the Lake Hickory study area (High Shoals Creek near Dysartsville, McDowell County) (Caldwell, 1992).

Specific conductance in Lake Hickory generally was low in the winter and spring of 1993, then gradually increased through summer to peak in the late fall (fig. 8). Concentrations in the headwaters were driven by concentrations in releases from Rhodhiss Lake, which also varied seasonally as described by Giorgino and Bales (1997). Specific conductance in the mainstem of Lake Hickory ranged from 37 to 139 µS/cm with a median of 57 µS/cm. The maximum concentration was measured near the reservoir bottom at site 40A on October 19, 1993.

Distributions of specific conductance also may have been influenced by prevailing water-circulation patterns and by the development of hypolimnetic anoxia in Lake Hickory. As previously noted, summertime inflow from Rhodhiss Dam plunged below the surface of Lake Hickory and moved through the reservoir as interflow. During stratified conditions, specific conductance in Lake Hickory generally was lowest above the interflow and highest in the hypolimnion, especially near the reservoir bottom (fig. 8). Under anoxic conditions, such as in the hypolimnion of Lake Hickory, the sediment-water interface becomes chemically reduced, additional ionic constituents may be released from the sediment into overlying waters (Mortimer, 1941, 1942), and specific conductance may increase (Wetzel, 1983).

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 decrease water clarity and may increase the cost of drinking-water treatment. In this investigation, total suspended solids (TSS) and volatile suspended solids (VSS) were measured.

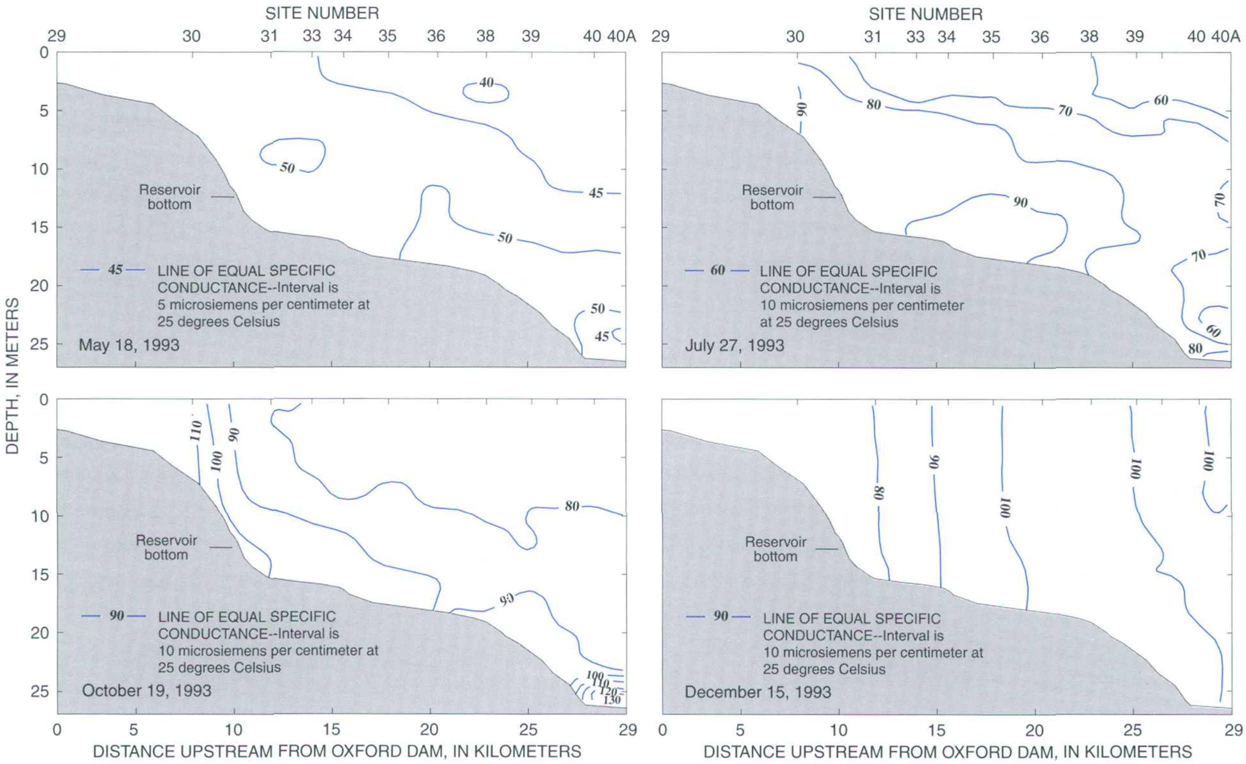


Figure 8. Specific conductance in Lake Hickory for selected days during the study period (vertical scale exaggerated). Site locations shown in figure 2.

The highest concentrations of TSS were observed at Upper Little River site 58 and Middle Little River site 60 (fig. 9; table 2). Maximum concentrations of 142 and 143 mg/L were measured at sites 58 and 60, respectively, during high flow in March 1993. As expected, concentrations of solids were directly related to streamflow at the two tributary sites throughout the investigation because more solids are carried in suspension at high water velocity than at low velocity. Median concentrations of TSS were 8 mg/L at site 58 and 14 mg/L at site 60 (table 2).

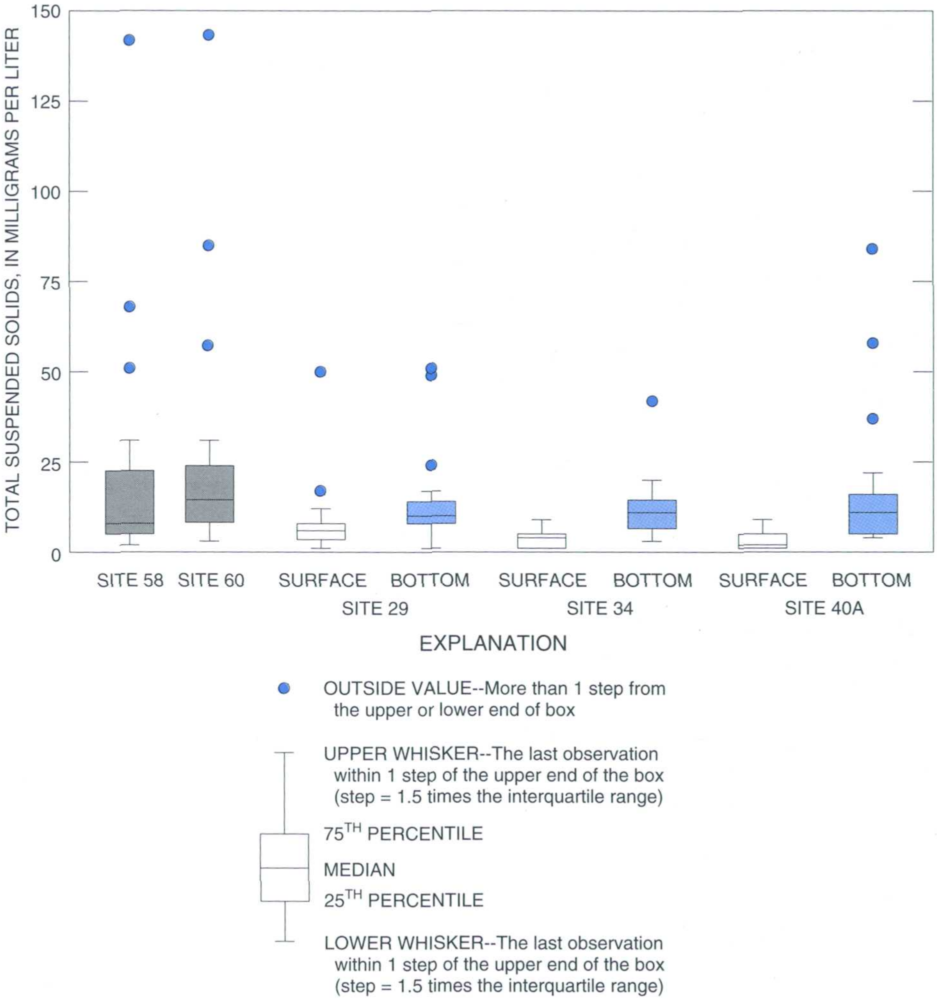


Figure 9. Total suspended solids concentrations at Upper Little River site 58, Middle Little River site 60, and Lake Hickory sites 29, 34, and 40A, January 1993 through March 1994.

Table 2. Statistical summary of selected water-quality constituents at Upper Little River site 58 and Middle Little River site 60, January 1993 through March 1994

[Max, maximum; Min, minimum; Mean, arithmetic mean; Med, median; mg/L, milligrams per liter; <, less than; *, value is estimated by using a log-probability regression to predict the values of data below the detection limit; cols./100 mL, colonies per 100 milliliters; †, geometric mean]

Constituent	Site 58				Site 60			
	Max	Min	Mean	Med	Max	Min	Mean	Med
Solids, total suspended (mg/L)	142	2	21	8	143	3	26	14
Solids, volatile suspended (mg/L)	22	<1	5*	3	23	<1	6*	5
Nitrogen, nitrite plus nitrate dissolved (mg/L as N)	.341	.053	.248	.254	.549	.084	.264	.254
Nitrogen, ammonia dissolved (mg/L as N)	.057	<.002	.018*	.013	.069	.005	.037	.036
Nitrogen, organic total (mg/L as N)	.30	.14	.20	.19	.97	.13	.26	.18
Phosphorus, total (mg/L as P)	.107	.006	.028	.016	.139	.009	.038	.027
Phosphorus, orthophosphate dissolved (mg/L as P)	.007	<.001	.002*	.001	.014	<.001	.002*	<.001
Fecal coliform bacteria (cols./100 mL)	800	27	145†	185	980	20	223†	240

Among the three sites in Lake Hickory, median concentrations of TSS in surface samples decreased from 6 mg/L at site 29 to 4 mg/L at site 34 and 2 mg/L at site 40A (fig. 9; table 3). Water clarity increased in a downstream direction, with median Secchi disk transparency values of 0.75 m at site 29, 1.1 m at site 34, and 1.3 m at site 40A (table 3). At each reservoir site, bottom samples had higher TSS concentrations than surface samples (fig. 9). Median concentrations of TSS in bottom samples were 10 mg/L at site 29 and 11 mg/L at sites 34 and 40A (fig. 9; table 3).

Volatile suspended solids represent the organic component of total suspended solids. Median VSS concentrations were 3 mg/L at Upper Little River site 58 and 5 mg/L at Middle Little River site 60 (table 2). Median concentrations of VSS were less than or equal to 2 mg/L in Lake Hickory surface samples and 4 mg/L in bottom samples (table 3).

Higher concentrations of TSS were observed in Rhodhiss Lake, especially in the upper portion of that reservoir; however, VSS concentrations were similar in Rhodhiss Lake and Lake Hickory (Giorgino and Bales, 1997). Results suggest that the headwaters of Lake Hickory had lower inputs of inorganic solids than the headwaters of Rhodhiss Lake, probably because some solids were retained in Rhodhiss Lake.

Nitrogen

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

Table 3. Statistical summary of selected water-quality properties and constituents at Lake Hickory sites 29, 34, and 40A, January 1993 through March 1994

[Max, maximum; Min, minimum; Mean, arithmetic mean; Med, median; mg/L, milligrams per liter; <, less than; *, value is estimated by using a log-probability regression to predict the values of data below the detection limit; µg/L, micrograms per liter; cols./100 mL, colonies per 100 milliliters; †, geometric mean]

Constituent or property	Site 29				Site 34				Site 40A			
	Max	Min	Mean	Med	Max	Min	Mean	Med	Max	Min	Mean	Med
					Near surface							
Transparency, Secchi disk (meters)	0.90	0.15	0.69	0.75	1.60	0.60	1.12	1.10	2.30	0.35	1.27	1.30
Solids, total suspended (mg/L)	50	1	8	6	9	<1	4*	4	9	<1	3*	2
Solids, volatile suspended (mg/L)	8	<1	3*	2	8	<1	2*	1	6	<1	2*	1
Nitrogen, nitrite plus nitrate dissolved (mg/L as N)	.315	.058	.195	.192	.367	.006	.166	.166	.351	<.005	.183*	.227
Nitrogen, ammonia dissolved (mg/L as N)	.142	.008	.074	.074	.757	.004	.089	.053	.149	.002	.052	.050
Nitrogen, organic total (mg/L as N)	.89	.11	.28	.16	.49	.08	.20	.18	.58	.05	.22	.19
Phosphorus, total (mg/L as P)	.126	.020	.052	.047	.073	.019	.033	.031	.050	.012	.025	.019
Phosphorus, orthophosphate dissolved (mg/L as P)	.020	<.001	.010*	.010	.038	<.001	.009*	.002	.025	<.001	.006*	.001
Chlorophyll *a* (µg/L)	110	.300	13	1.6	26	.4	7.4	5.0	12	.4	4.7	5.4
Fecal coliform bacteria (cols./100 mL)	200	4	27†	23	800	<1	9†	9	98	<1	9†	10
					Near bottom							
Solids, total suspended (mg/L)	51	1	14	10	42	3	12	11	84	4	17	11
Solids, volatile suspended (mg/L)	9	<1	4*	3	9	1	4	4	14	1	4	3
Nitrogen, nitrite plus nitrate dissolved (mg/L as N)	.317	.134	.217	.202	.425	.085	.235	.228	.360	.005	.229	.256
Nitrogen, ammonia dissolved (mg/L as N)	.193	.054	.111	.095	.632	.015	.164	.109	.828	.066	.216	.135
Nitrogen, organic total (mg/L as N)	1.15	.10	.25	.18	.41	.07	.18	.16	1.22	.00	.21	.16
Phosphorus, total (mg/L as P)	.127	.032	.059	.050	.071	.023	.048	.046	.217	.015	.071	.054
Phosphorus, orthophosphate dissolved (mg/L as P)	.020	.001	.012	.012	.040	.001	.014	.014	.065	.001	.016	.010

At Upper Little River site 58 and Middle Little River site 60, NO_2+NO_3 and TON were present in much higher concentrations than NH_4 (fig. 10; table 2). In well oxygenated waters such as these two tributaries, NH_4 is rapidly assimilated by aquatic plants or converted to nitrate by bacteria; hence, concentrations are generally low (Wetzel, 1983). None of the three nitrogen species followed clear seasonal patterns at these tributary sites (fig. 10). Concentrations of NO_2+NO_3 observed in Upper Little River and Middle Little River were within ranges previously observed for unpolluted streams in the same geochemical zone (Simmons and Heath, 1982; Caldwell, 1992). Ranges of NH_4 and TON concentrations at the two tributary sites were larger than those reported by Simmons and Heath (1982) for unpolluted streams, but comparable to concentrations reported by Caldwell (1992) for streams in minimally disturbed watersheds.

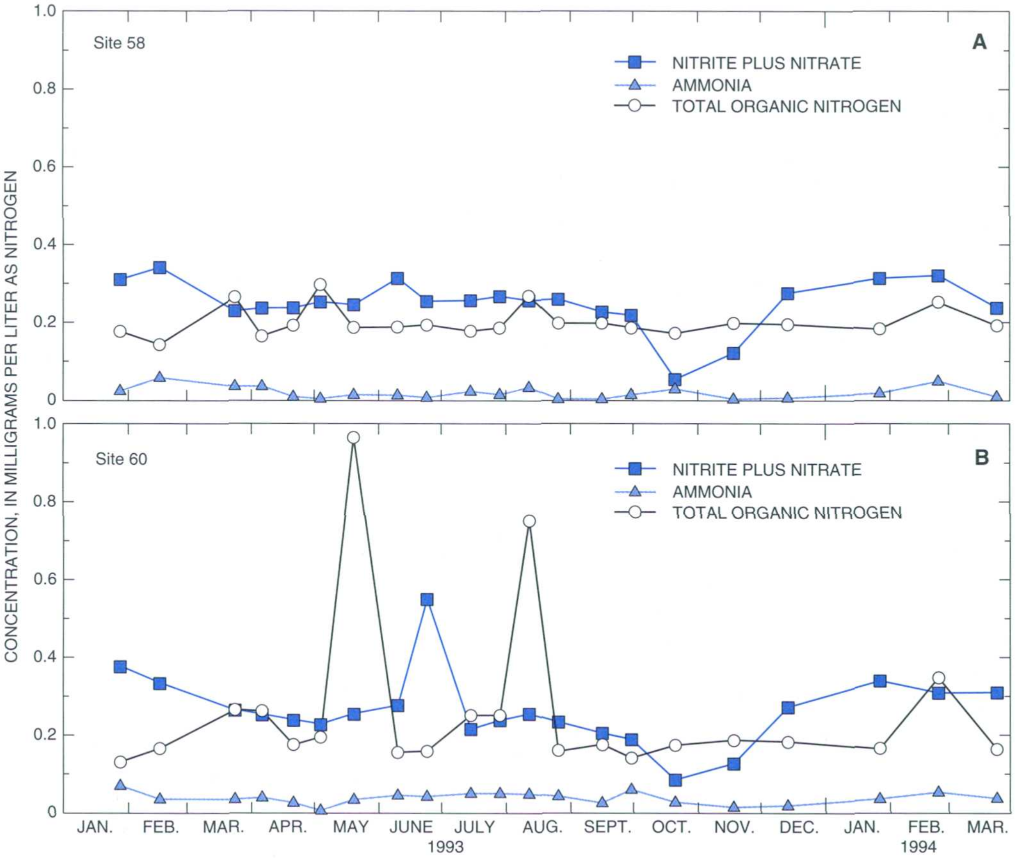


Figure 10. Concentrations of nitrite plus nitrate, ammonia, and total organic nitrogen at (A) Upper Little River site 58 and (B) Middle Little River site 60, January 1993 through March 1994.

In Lake Hickory, NO_2+NO_3 was the predominant nitrogen species during cool months when the reservoir was unstratified or weakly stratified (fig. 11). With the onset of stratification in summer 1993, surface NO_2+NO_3 concentrations declined to less than 0.2 mg/L at site 29, and less than 0.1 mg/L at sites 34 and 40A. Surface concentrations of NH_4 also remained low during the growing season (fig. 11). A simultaneous increase in TON suggests that phytoplanktonic assimilation was responsible for depleting both NO_2+NO_3 and NH_4 from the epilimnion of Lake Hickory (fig. 11). TON constituted the majority of the total nitrogen present in surface samples at all sites during summer stratification. In October 1993, an unusually high concentration of NH_4 (0.757 mg/L) was observed near the surface at site 34

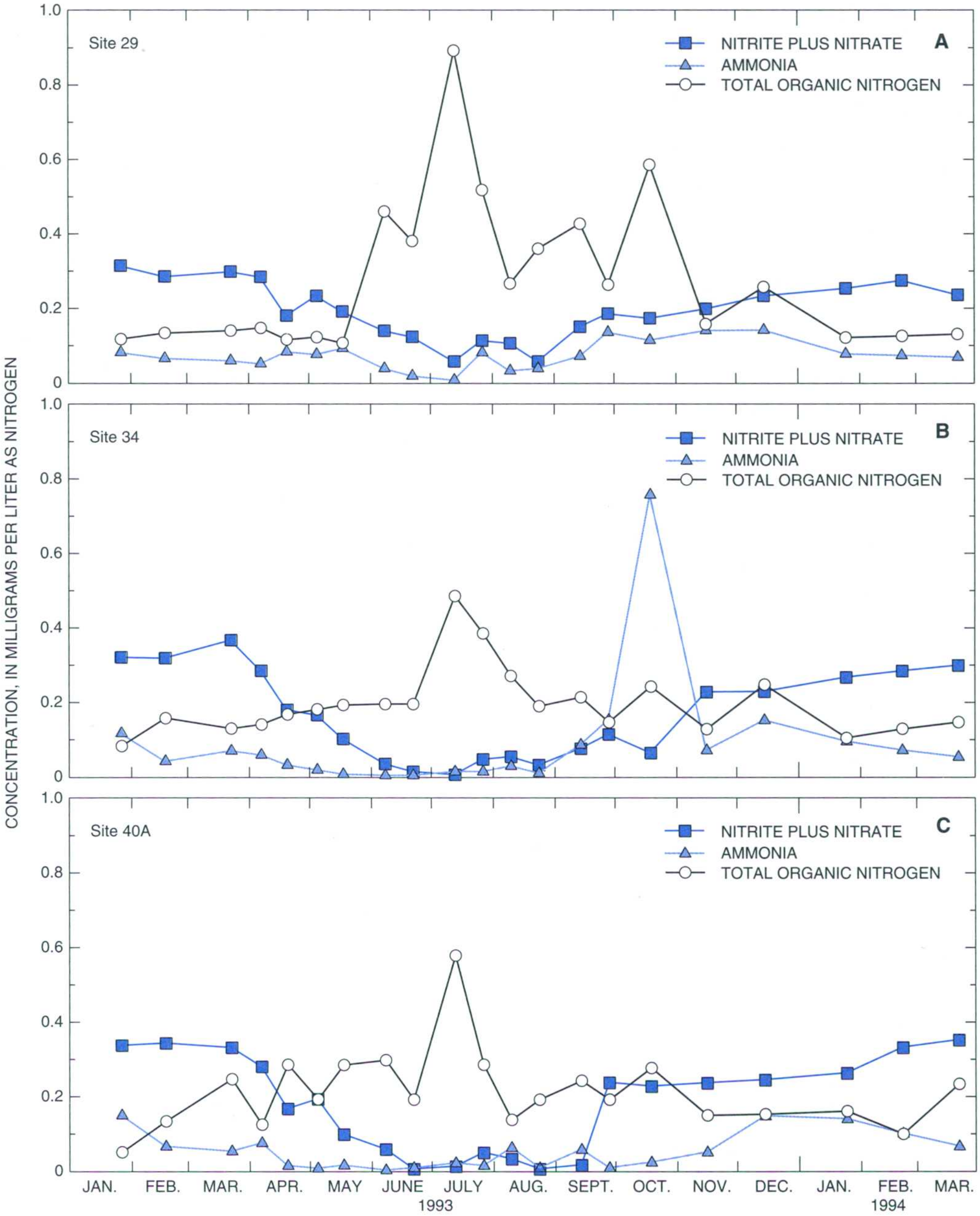


Figure 11. Concentrations of nitrite plus nitrate, ammonia, and total organic nitrogen in surface samples from Lake Hickory sites (A) 29, (B) 34, and (C) 40A, January 1993 through March 1994.

(fig. 11B). The source of this elevated concentration is not known.

Nitrite plus nitrate also was the predominant nitrogen species in Lake Hickory bottom waters during winter and spring (fig. 12). During late summer, however, when concentrations of dissolved oxygen were low in the hypolimnion at sites 34 and 40A, concentrations of NO_2+NO_3 decreased and concentrations of NH_4 increased (fig. 12B and C). Under anaerobic conditions, such as in the hypolimnion of Lake Hickory, nitrite and nitrate are rapidly converted to nitrogen gas by bacterial denitrification. Elevated concentrations of NH_4 likely resulted from the decomposition of organic matter in both hypolimnetic waters and bed sediments under anaerobic conditions. A maximum NH_4 concentration of 0.828 mg/L was observed at site 40A on September 14, 1993 (fig. 12C; table 3). Concentrations of TON in

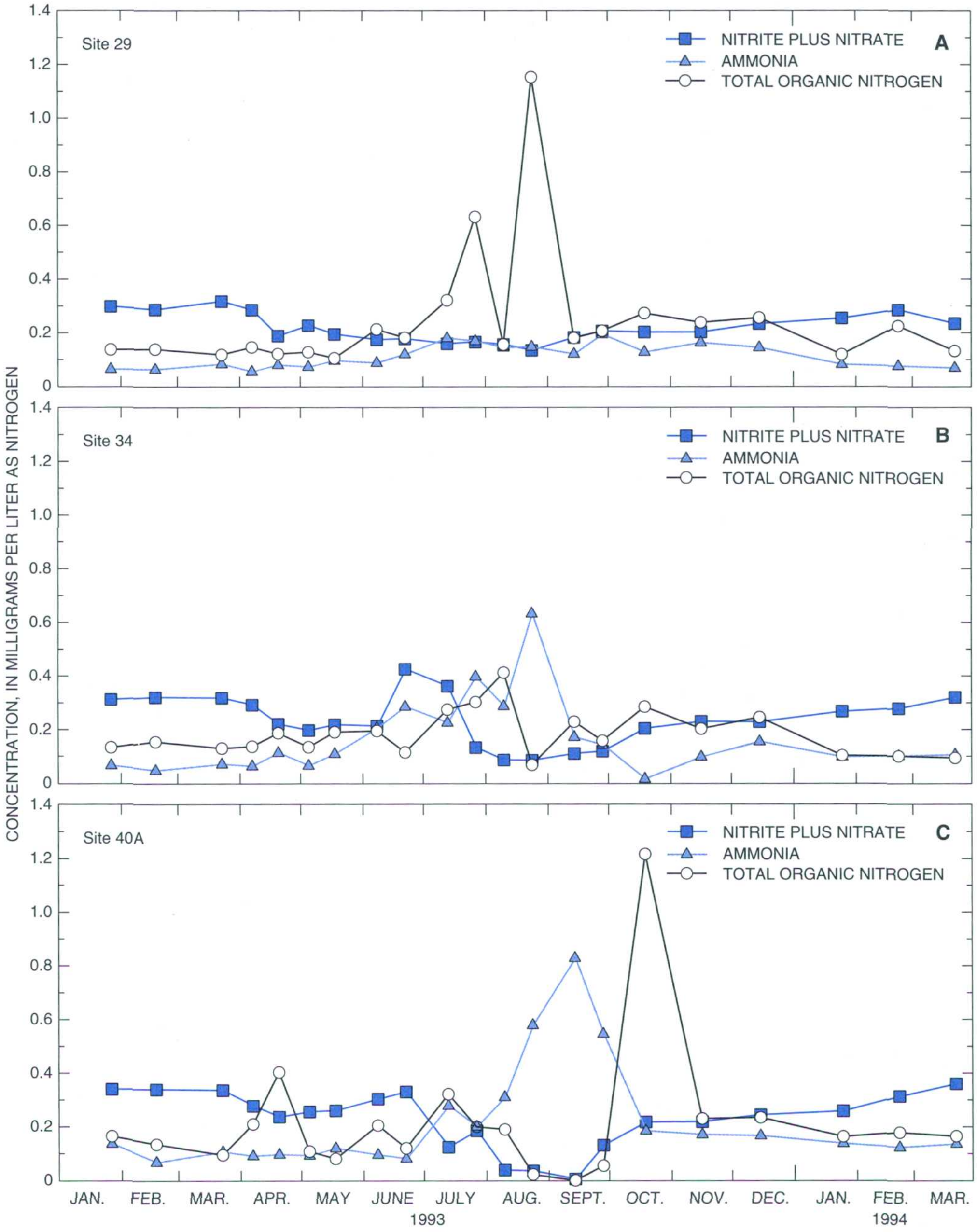


Figure 12. Concentrations of nitrite plus nitrate, ammonia, and total organic nitrogen in bottom samples from Lake Hickory sites (A) 29, (B) 34, and (C) 40A, January 1993 through March 1994.

reservoir bottom waters varied widely between sampling dates (fig. 12).

Similar seasonal and spatial variations in nitrogen species were observed in Rhodhiss Lake (Giorgino and Bales, 1997). Nitrite plus nitrate was the predominant nitrogen species in Rhodhiss Lake surface waters during cool months. During the summer of 1993, NO_2+NO_3 and NH_4 decreased and TON increased in surface waters. Meanwhile, concentrations of NH_4 increased in the hypolimnion at the forebay site (Giorgino and Bales, 1997).

North Carolina water-quality standards state that NO_2+NO_3 should not exceed 10 mg/L in surface waters classified for water supply (North Carolina Department of Environment, Health, and Natural Resources, 1997). All NO_2+NO_3 concentrations measured at Lake Hickory and its two tributary sites were well below that level (table 2).

Phosphorus

Phosphorus components measured during this investigation include total phosphorus as P (TP) and orthophosphate as P (PO_4). Total phosphorus includes both particulate and dissolved, and organic and inorganic phosphorus compounds. PO_4 is a dissolved, inorganic species of phosphorus that is readily available to phytoplankton.

Concentrations of TP at Upper Little River site 58 ranged from 0.006 to 0.107 mg/L and averaged 0.028 mg/L (table 2). At Middle Little River site 60, TP ranged from 0.009 to 0.139 mg/L and averaged 0.038 mg/L (table 2). Mean TP concentrations in these two tributaries were higher than the mean of 0.01 mg/L reported for an unpolluted stream in the same geochemical zone (Simmons and Heath, 1982). Furthermore, the median TP concentration of 0.027 mg/L at site 60 was higher than medians of 0.01 and 0.02 mg/L reported for minimally disturbed streams in forested watersheds near the Lake Hickory study area (Caldwell, 1992). The highest concentrations of TP at both sites were observed in conjunction with heavy rainfall and high streamflows in March 1993 and February 1994 (fig. 13). Concentrations of PO_4 in these two tributaries generally were very low

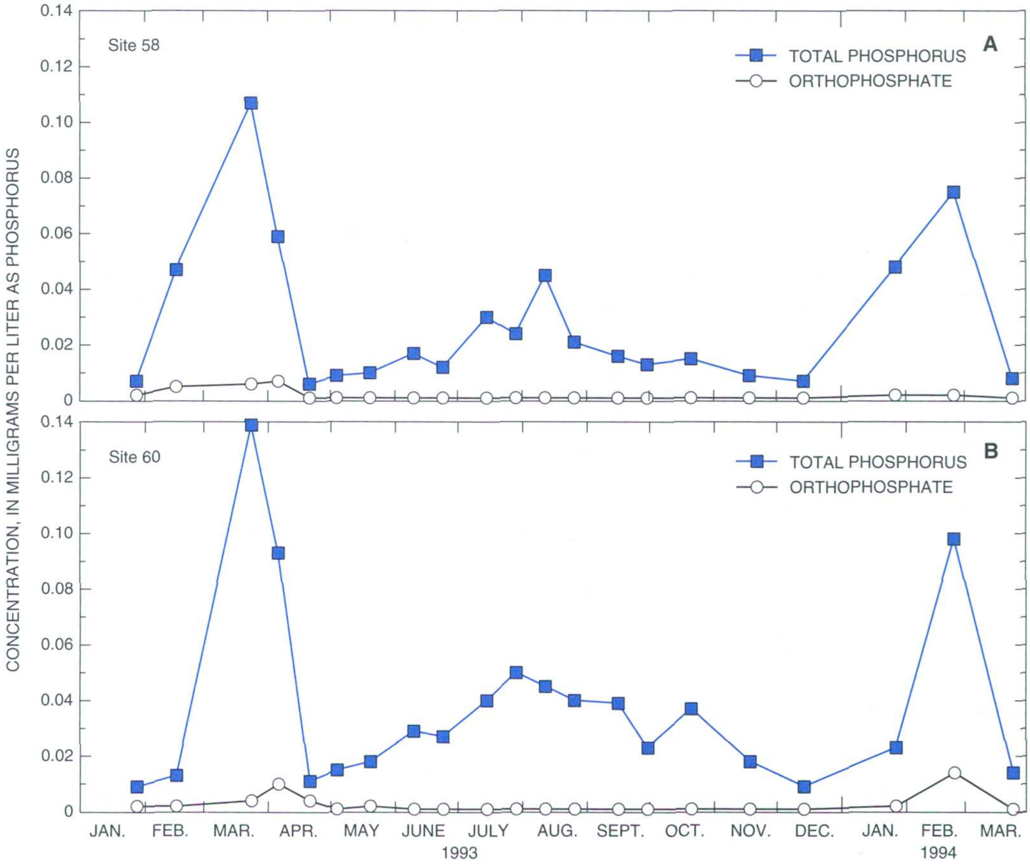


Figure 13. Concentrations of total phosphorus and orthophosphate at (A) Upper Little River site 58 and (B) Middle Little River site 60, January 1993 through March 1994.

throughout the study, with medians less than or equal to 0.001 mg/L (fig. 13; table 2). During high flows, PO_4 concentrations remained low, indicating that most of the increase in TP was associated with particulate matter or organic compounds.

Median concentrations of TP in Lake Hickory surface samples decreased from 0.047 mg/L in the headwaters to 0.031 mg/L at mid-reservoir and 0.019 mg/L in the forebay (table 2). Concentrations near the surface and bottom at site 29 were quite variable and followed no clear seasonal pattern (figs. 14A and 15A). This variability is not surprising because this site is largely influenced by regulated releases from Rhodhiss Lake, which are based on hydropower demands, and discharges of wastewater from municipal and industrial facilities.

At mid-reservoir site 34 and forebay site 40A, surface concentrations of TP were lowest during the

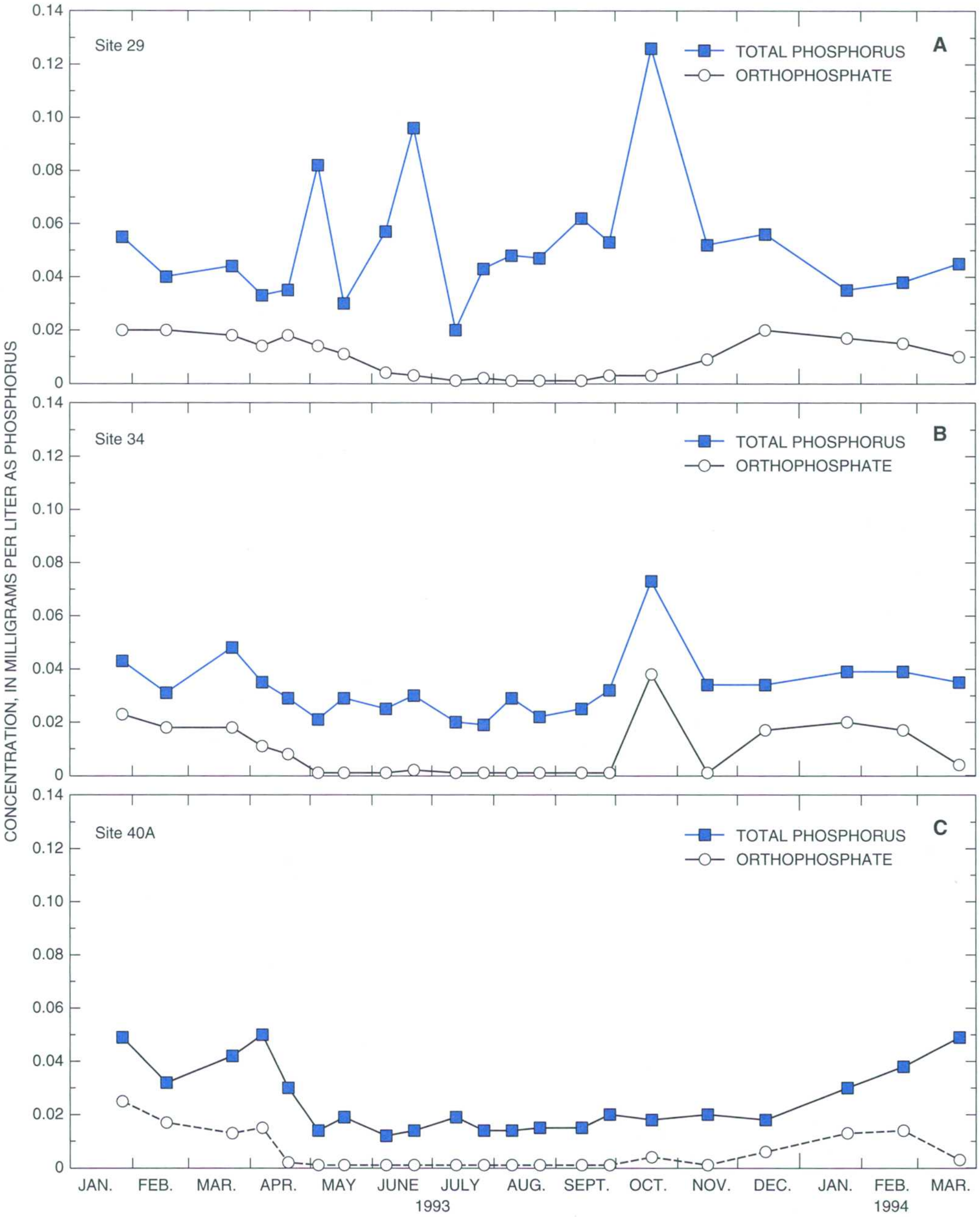


Figure 14. Concentrations of total phosphorus and orthophosphate in surface samples from Lake Hickory sites (A) 29, (B) 34, and (C) 40A, January 1993 through March 1994.

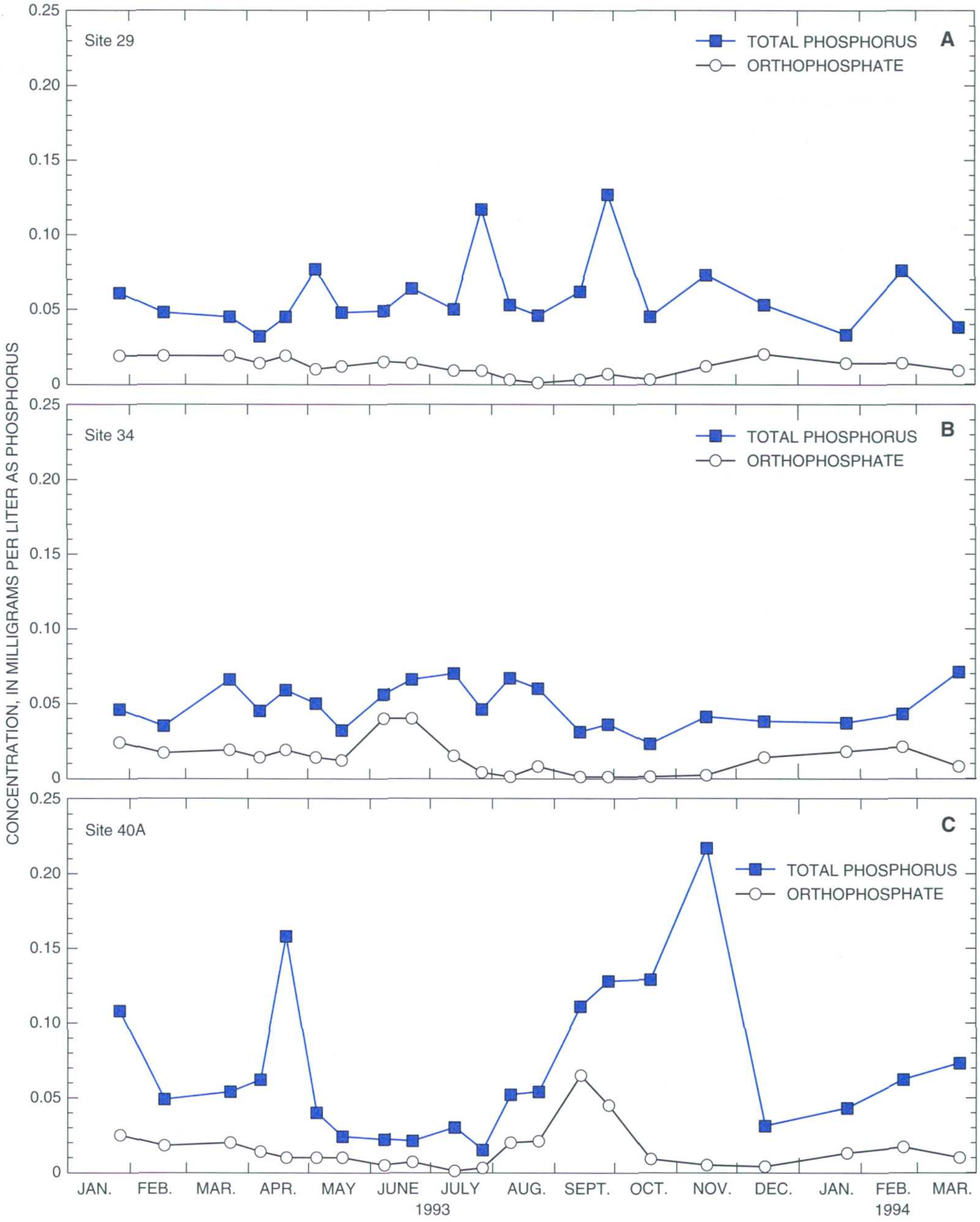


Figure 15. Concentrations of total phosphorus and orthophosphate in bottom samples from Lake Hickory sites (A) 29, (B) 34, and (C) 40A, January 1993 through March 1994.

summer (fig. 14B and C), and were lower than concentrations in bottom waters on most sampling dates (fig. 15B and C). These vertical differences were most pronounced at site 40A, where TP in the hypolimnion increased steadily from late July through November, when a maximum concentration of 0.217 mg/L was observed (fig. 15C; table 3). It is common for the hypolimnion of a productive lake to contain elevated phosphorus levels, particularly during later stages of stratification, because anoxic conditions promote the release of sediment-bound phosphorus into the water column (Wetzel, 1983).

Median concentrations of PO_4 in Lake Hickory surface samples were highest in the headwaters (0.010 mg/L) and decreased to 0.002 mg/L at site 34 and 0.001 mg/L at site 40A (table 3). Epilimnetic PO_4 concentrations decreased at all three sites during the summer of 1993 (fig. 14), probably as a result of algal uptake. Surface concentrations of PO_4 were less than 0.005 mg/L from June through October at site 29 and less than or equal to 0.002 mg/L from May through September 1993 at sites 34 and 40A. On October 19, 1993, a peak surface PO_4 concentration of 0.038 mg/L was observed at site 34 (fig. 14B; table 3), coinciding with a peak NH_4 concentration (fig. 11B). Sources of these elevated nutrient concentrations are not known.

At all reservoir sites, PO_4 was higher in bottom waters than in surface waters during the summer (figs. 14 and 15). A maximum PO_4 concentration of 0.065 mg/L was observed near the bottom at site 40A on September 14, 1993 (fig. 15C; table 3).

Similar to that in Lake Hickory, median concentrations of TP and PO_4 in Rhodhiss Lake tended to be highest in the headwaters and decreased in a downstream direction. Surface concentrations of TP in Rhodhiss Lake followed no clear seasonal pattern and were lower than concentrations in bottom waters at the mid-reservoir and forebay sites. Surface PO_4 concentrations at the mid-reservoir and forebay sites in Rhodhiss Lake were low from May through September, and bottom PO4 concentrations were high in the forebay from mid-July through mid-August (Giorgino and Bales, 1997).

Chlorophyll *a*

Chlorophyll *a*, a measure of algal biomass, was measured at reservoir sites 29, 34, and 40A. At all three sites, concentrations of chlorophyll *a* were very low during winter months and higher from spring through fall (fig. 16). From May through September 1993, the mean chlorophyll *a* concentration was 25 µg/L at site 29, 12 µg/L at site 34, and 6.4 µg/L at site 40A. Based on the concentrations of total phosphorus and chlorophyll *a* observed in the epilimnion, Lake Hickory would be classified eutrophic from its

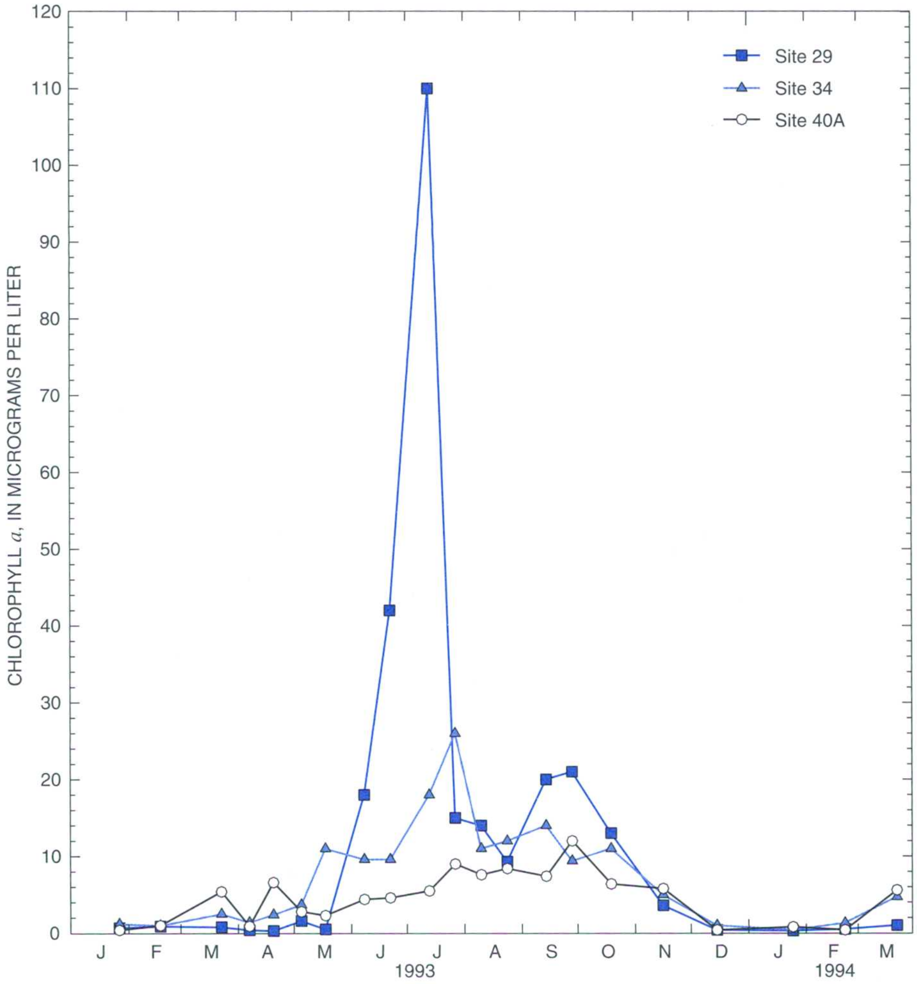


Figure 16. Concentrations of chlorophyll *a* in photic-zone samples from Lake Hickory sites 29, 34, and 40A, January 1993 through March 1994.

headwaters to mid-reservoir, and meso-eutrophic from mid-reservoir to the dam (Wetzel, 1983).

The maximum chlorophyll *a* concentration of 110 µg/L was observed at headwaters site 29 on July 13, 1993 (fig. 16; table 3). In all, two chlorophyll *a* values at site 29 exceeded the North Carolina water-quality standard of 40 µg/L (North Carolina Department of Environment, Health, and Natural Resources, 1997). No violations were observed at sites 34 or 40A.

In contrast, concentrations of chlorophyll *a* were low in the riverine headwaters of Rhodhiss Lake and peaked at the mid-reservoir site (Giorgino and Bales, 1997). A comparison of the two reservoirs indicates that the headwaters of Rhodhiss Lake had lower water transparency and more inorganic turbidity than the headwaters site in Lake Hickory (Giorgino and Bales, 1997). From May through September 1997, median Secchi transparency was 0.50 m in the headwaters of Rhodhiss Lake and 0.75 m in the headwaters of Lake Hickory. Moreover, median concentrations of nonvolatile suspended solids were 14 mg/L and 4 mg/L at headwater sites in Rhodhiss Lake and Lake Hickory, respectively. This suggests that some of the solids carried by the Catawba River were retained within Rhodhiss Lake. Lower concentrations of inorganic solids resulted in increased light availability in the headwaters of Lake Hickory which, in turn, supported phytoplankton growth.

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 cols./100 mL based on at least five samples collected during a 30-day period (North Carolina Department of Environment, Health, and Natural Resources, 1997). Sampling frequency during this investigation of Lake Hickory did not meet that criterion; therefore, the results could not be said to either meet or violate the State water-quality standard. Even so, repeated exceedences of this numerical criterion generally are undesirable and indicate a potential threat to human health.

At Upper Little River site 58, fecal coliform concentrations exceeded 200 cols./100 mL in 40 percent of the samples. Sixty percent of samples collected from Middle Little River site 60 contained more than 200 cols./100 mL, and both the median and geometric mean concentrations exceeded this criterion (table 2). Maximum concentrations of fecal coliform bacteria at sites 58 and 60 were 800 and 980 cols./100 mL, respectively (table 2). At both tributary sites, values greater than 200 cols./100 mL were observed across a range of flow conditions and during every season. Sources of the high bacterial concentrations in these two tributaries were not investigated but warrant further study.

At site 34 in Lake Hickory, a maximum concentration of 800 cols./100 mL was observed in January 1994 (table 3). No additional samples from any sites in Lake Hickory were greater than 200 cols./100 mL. In contrast, high fecal coliform concentrations were observed frequently at Rhodhiss Lake—in 16 to 37 percent of the samples collected at each reservoir site (Giorgino and Bales, 1997).

Loads of Solids and Nutrients for Selected Sites and Point Sources

Loads of total suspended solids, nitrogen, and phosphorus were estimated for Upper Little River site 58, Middle Little River site 60, Lake Hickory site 28, and for selected point-source discharges in the Lake Hickory watershed downstream from Rhodhiss Dam (fig. 2). Loads were estimated for April 1993 through March 1994—the period for which the CE-QUAL-W2 model was calibrated for Lake Hickory.

Simultaneous measurements of streamflow and constituent concentrations collected by the USGS during the investigation were used to develop regression relations for sites 58 and 60. Continuous records of streamflow then were used to compute hourly, daily, and monthly loads for each constituent. Monthly loads are summarized in table 4.

Peak-streamflow periods accounted for a large proportion of the TSS and TP loads at the two tributary sites. For example, high flows were measured at Upper Little River site 58 and Middle Little River site 60 during March 28–29, 1994, following heavy rainfall (fig. 4B and C). It was estimated that these 2 days accounted for 28 percent of the TSS load and 19 percent of the TP load at site 58 for the 12-month period of April 1993 through March 1994. However, this storm delivered only 8 percent of the total nitrogen (TN) load at site 58. At Middle Little River site 60, loads during these 2 days represented 44 percent of the

Table 4. Monthly loads of total suspended solids and nutrients at Upper Little River site 58 and Middle Little River site 60, April 1993 through March 1994

[Values were computed by summing hourly estimates and are listed to three significant figures]

Month	Monthly loads, in kilograms											
	Total suspended solids		Nitrite plus nitrate nitrogen, as N		Ammonia nitrogen, as N		Total nitrogen, as N		Orthophosphate phosphorus, as P		Total phosphorus, as P	
	Site 58	Site 60	Site 58	Site 60	Site 58	Site 60	Site 58	Site 60	Site 58	Site 60	Site 58	Site 60
April	239,000	232,000	2,110	3,410	253	460	3,960	7,010	20.4	38.7	296	271
May	75,200	109,000	1,350	2,500	114	332	2,480	4,950	7.59	19.4	142	165
June	31,500	79,800	838	1,620	50.5	215	1,510	3,110	3.65	6.82	68.9	114
July	27,300	71,200	721	1,180	41.4	173	1,300	2,310	3.22	4.54	56.5	103
August	20,100	48,000	609	684	32.2	115	1,120	1,390	2.57	3.07	46.4	94.5
September	17,500	45,400	413	624	24.9	105	821	1,310	2.18	3.58	35.0	94.1
October	10,600	28,200	249	355	14.3	68.6	568	814	1.50	2.24	23.0	83.0
November	32,600	63,700	420	615	31.8	103	858	1,330	3.03	5.77	43.1	123
December	64,600	86,700	671	993	44.8	154	1,250	2,010	4.64	6.95	70.0	139
January	216,000	169,000	1,510	1,890	152	261	2,860	3,790	14.5	17.2	226	234
February	170,000	167,000	1,070	1,990	98.8	265	2,010	3,950	10.4	17.5	153	218
March	755,000	1,300,000	2,340	3,120	284	425	4,760	6,590	36.9	81.4	610	827
Total	1,660,000	2,400,000	12,300	19,000	1,140	2,680	23,500	38,600	111	207	1,770	2,470

TSS load, 21 percent of the TP load, and 5 percent of the TN load for the same 12-month period. In the two tributaries, increased loads of TP and TSS during high-flow events resulted from both higher in-stream concentrations of these constituents and increases in streamflow. On the other hand, concentrations of TN did not increase during peak-flow events; therefore, higher streamflows alone accounted for increased loads.

NO_2+NO_3 and NH_4 accounted for approximately 57 percent of the TN load at Upper Little River site 58 and 56 percent at Middle Little River site 60. Loads of PO_4 accounted for 6 percent of the TP load at site 58 and 8 percent at Middle Little River site 60, indicating that most of the phosphorus load from these two tributaries was in the form of organic complexes or was associated with particulate matter (table 4). It should be noted that loads estimated for these two sites may not exactly represent the loads reaching the mainstem of Lake Hickory because some transformation or loss of material might have occurred.

Loads from Rhodhiss Dam into the headwaters of Lake Hickory (table 5) were estimated from daily mean flows recorded at site 28 and constituent concentrations measured at site 29 (fig. 2). Regression relations were developed and daily loads were computed for each constituent. As described later in this report, site 29 was located downstream from two small point-source discharges. However, these discharges probably had minimal influence on concentrations because they represented, on average, only 0.01 percent of the total flow into the headwaters of Lake Hickory.

Releases from Rhodhiss Dam delivered the majority of TSS, TN, and TP loads into Lake Hickory (table 5). Because flows at site 28 were regulated by hydropower operations, loads were not as strongly influenced by precipitation as in the Upper Little River and the Middle Little River. Similar to flow patterns at those two tributaries, high flows occurred at site 28 during March 28–29, 1994 (fig. 4). However, these 2 days accounted for only 2 percent of the TSS, TN, and TP loads at site 28 for the period of April 1993 through March 1994.

From April 1993 through March 1994, total loads of TN and TP into the headwaters of Lake Hickory were 17 and 16 percent higher, respectively, than loads into the headwaters of Rhodhiss Lake (Giorgino and Bales, 1997). This is not surprising because the contributing drainage area and, hence, the average flows for the Lake Hickory site are larger than for the Rhodhiss Lake site. In contrast, TSS loads at Lake Hickory site 28 were 29 percent lower than in the headwaters of Rhodhiss Lake, suggesting that substantial amounts of solids entering Rhodhiss Lake were retained in that reservoir.

Table 5. Monthly loads of total suspended solids and nutrients at Lake Hickory site 28, April 1993 through March 1994

[Values were computed by summing daily estimates and are listed to three significant figures]

Month	Monthly loads, in kilograms					
	Total suspended solids	Nitrite plus nitrate nitrogen, as N	Ammonia nitrogen, as N	Total nitrogen, as N	Orthophosphate phosphorus, as P	Total phosphorus, as P
April	3,520,000	72,700	22,300	139,000	3,460	13,900
May	2,870,000	57,700	19,700	122,000	2,640	12,100
June	1,090,000	18,900	10,200	60,900	748	6,030
July	710,000	12,600	7,880	46,000	419	4,540
August	670,000	11,900	7,440	43,400	396	4,280
September	472,000	9,380	6,000	34,500	245	3,390
October	405,000	8,500	5,490	31,400	198	3,080
November	640,000	12,100	7,130	41,600	388	4,090
December	1,060,000	18,900	10,000	59,500	739	5,890
January	2,030,000	37,000	15,600	95,300	1,680	9,470
February	2,380,000	48,800	16,900	104,000	2,090	10,400
March	3,240,000	64,800	21,000	130,000	3,180	13,000
Total	*19,100,000*	*373,000*	*150,000*	*908,000*	*16,200*	*90,100*

Six permitted point sources—one industry, four municipal wastewater-treatment plants (WWTP's), and one drinking-water treatment plant—discharged, on average, a combined total of 0.229 m^3/s into the Lake Hickory watershed downstream from Rhodhiss Dam. Although there were numerous additional point sources in the watershed, these six accounted for more than 99 percent of the permitted wastewater flow. None of these facilities were located in the Upper Little River or Middle Little River drainages. Two facilities discharged into Lake Hickory upstream from site 29, and one discharged immediately downstream from site 29. Effluent from the largest discharger, the Hickory-Northeast WWTP, entered Lake Hickory downstream from site 34. The remaining two facilities discharged into Gunpowder Creek (fig. 2).

The six facilities monitored volume of discharge and TSS concentrations in their effluents. In addition, the four municipal WWTP's and the industrial discharger monitored effluent concentrations of total nitrogen, ammonia, and total phosphorus. These data were provided to the USGS by the North Carolina Department of Environment, Health, and Natural Resources Compliance Monitoring System and were used to estimate loads of the various constituents during the period of April 1993 through March 1994. These estimates should be considered approximate, because monthly or quarterly monitoring data were used for the computations (table 6).

Load estimates indicate that these facilities were relatively minor sources of suspended solids, but they were potentially important sources of nitrogen and phosphorus (table 6). Loads from the four point sources downstream from site 28 were compared to total loads estimated for site 28 (table 5). These four facilities represented an additional 40-percent increase in NH_4 load, a 10-percent increase in TN load, and a 16-percent increase in TP load, but less than a 1-percent increase in TSS load. It should be noted that the discharged material probably did not all reach Lake Hickory, but might have been partially transformed or retained before reaching the reservoir mainstem. This is especially likely for effluent from the Lenoir-Gunpowder Creek WWTP, which discharged several kilometers upstream from Lake Hickory.

SIMULATION OF HYDRODYNAMICS, CONSTITUENT TRANSPORT, AND WATER QUALITY

A two-dimensional, laterally averaged hydrodynamic and water-quality model was constructed for Lake Hickory and was calibrated by using data collected during April 1993 through March 1994. The model simulates water level, currents, heat transport and temperature distribution, conservative material transport, and the transport and transformation of 11 chemical constituents. Objectives of the reservoir modeling were to develop the capability to simulate the movement and dispersion of spills or releases in the reservoir during stratified and unstratified conditions and the response of water quality (nutrient, algal, and DO concentrations) to possible changes in external loads and release patterns.

Table 6. Estimated total loads of total suspended solids and nutrients from selected point-source discharges in the Lake Hickory watershed, April 1993 through March 1994

[Facilities with daily mean discharges of at least 0.002 m^3/s were included. Loads are listed to three significant figures. m^3/s, cubic meters per second; WWTP, wastewater-treatment plant; —, no data]

Discharging facility	Average discharge (m^3/s)	Total loads, in kilograms			
		Total suspended solids	Ammonia nitrogen, as N	Total nitrogen, as N	Total phosphorus, as P
Rhodhiss WWTP	0.002	398	391	1,370	119
Huffman Finishing	.006	4,450	49.2	2,610	1,820
Hickory Water Plant	.017	5,030	—	—	—
Lenoir-Gunpowder Creek WWTP	.044	14,400	3,750	17,200	3,370
Granite Falls WWTP	.019	6,840	2,400	3,450	472
Hickory Northeast WWTP	.141	128,000	53,500	68,600	10,500
Total	0.229	159,000	60,100	93,200	16,300

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. Lake Hickory is relatively narrow, with a maximum width of less than 1 km. Temperature and DO data (fig. 17) suggest that a laterally averaged model formulation is appropriate for Lake Hickory. These functional requirements for the model were considered when selecting the model CE-QUAL-W2 for application to Lake Hickory. Complete details on model theory and structure, and an extensive bibliography for theoretical development and application are given by Cole

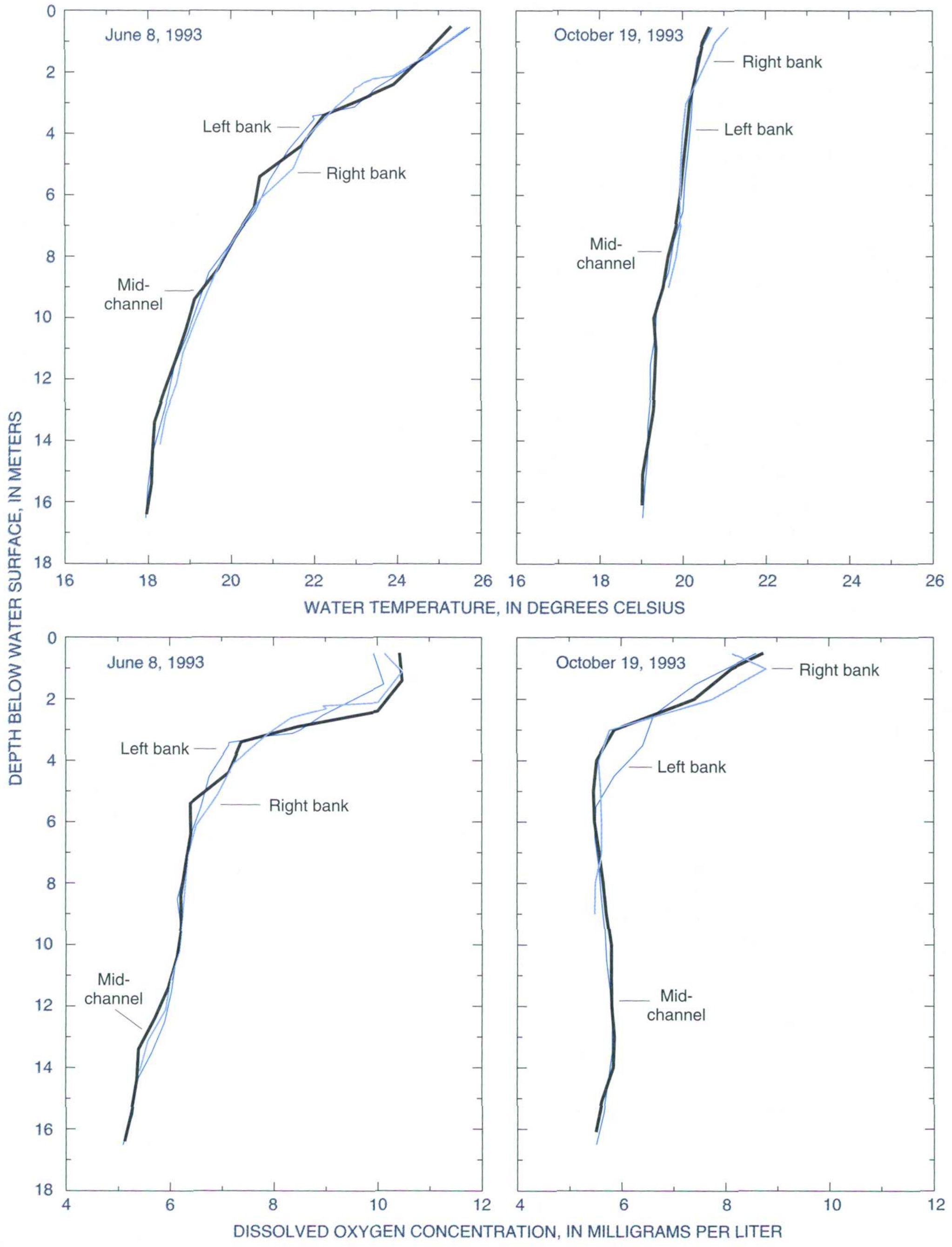


Figure 17. Vertical profiles of water temperature and dissolved oxygen concentration at Lake Hickory site 34 near left bank, mid-channel, and near right bank for stratified (June 8, 1993) and weakly stratified (October 19, 1993) conditions.

and Buchak (1995). Giorgino and Bales (1997) summarized features of the model relevant to the simulation of hydrodynamics, temperatures, and water-quality conditions in Rhodhiss Lake, which is upstream from Lake Hickory (fig. 1).

Model Implementation

Implementation of CE-QUAL-W2 for Lake Hickory included (1) development of the computational grid, (2) specification of boundary and initial conditions, and (3) preliminary selection of model parameter values. Model parameters from the calibrated Rhodhiss Lake model (Giorgino and Bales, 1997) were used as initial values for the Lake Hickory model. The model was calibrated and tested using data collected during April 1, 1993, through March 31, 1994.

Computational Grid

The model domain extends from U.S. Highway 321 bridge (site 29) to Oxford Dam (fig. 2), a distance of 22 km along the longitudinal axis of the reservoir. The domain encompasses five embayments—Gunpowder Creek, Upper Little River, Snow Creek, Middle Little River, and Long Shoals Creek. There are 43 computational segments along the mainstem of the reservoir, and each of the embayments is represented by a minimum of three segments (fig. 18). Volumes associated with smaller embayments not explicitly included in the model (shown as unnumbered segments in fig. 18) were added to nearby mainstem volumes so that the total volume of the reservoir was preserved by the computational grid. For example, the volume of the cove at the mouth of Falling Creek (fig. 2) was added to segment 22 (fig. 18). The mathematical formulation of the model suggests that segment lengths generally should exceed the maximum reservoir width in that segment. Segments ranged in length from 322 to 824 m. Each layer was 1 m thick. Distances from the elevation of the spillway crest to the bottom of the channel ranged from about 9 m near the U.S. Highway 321 bridge to 22 m at the dam. 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 in June 1994, when 43 cross sections were measured along the mainstem of

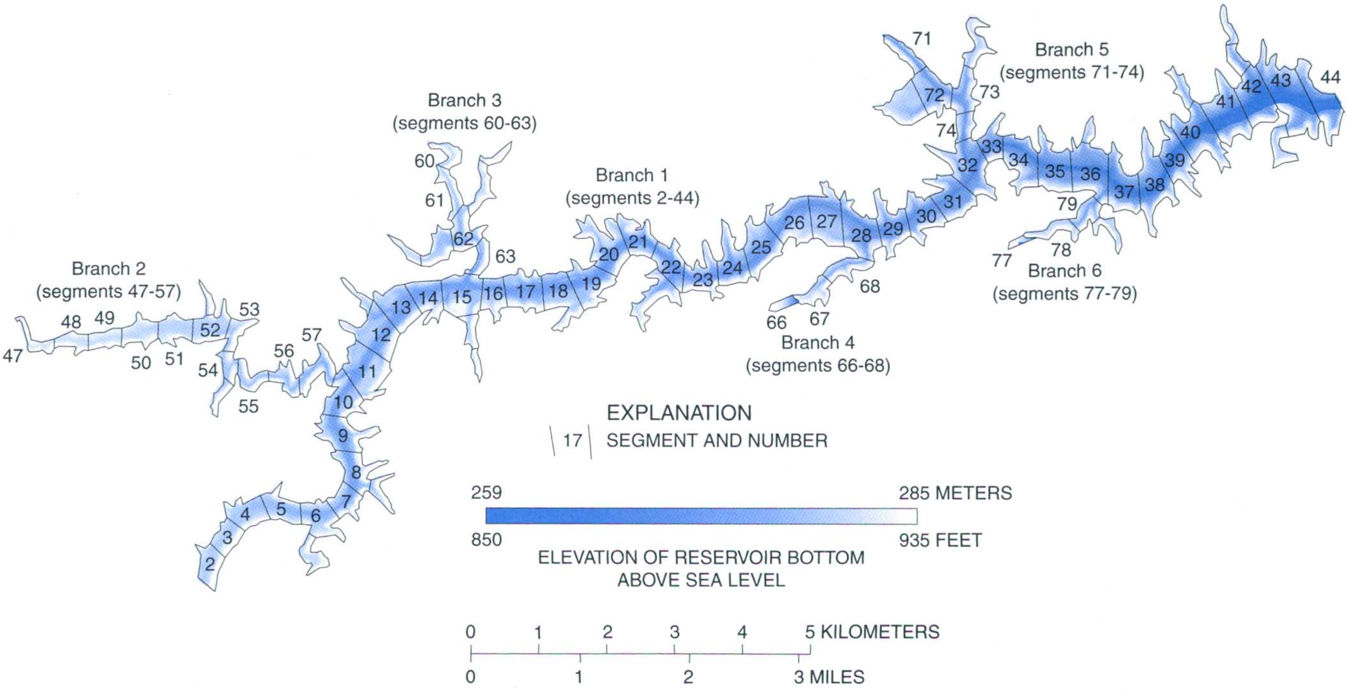


Figure 18. Lake Hickory computational segments.

the reservoir by 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. 18). Additional cross sections were measured in the coves and embayments around the reservoir. The water surface was about 1 m below the spillway crest when the cross sections were measured. The measured cross sections were extrapolated to an elevation equal to the elevation of the spillway crest 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.

An elevation-volume table (table 7) was developed for Lake Hickory from the computational grid, which is an approximation of the actual volume. The estimated full-pool volume is about 128 million m^3, which is somewhat less than a previously published value of 157 million m^3 (U.S. Environmental Protection Agency, 1975).

Table 7. Elevation-volume relation for Lake Hickory computational grid

Distance below spillway crest (meters)	Elevation (meters above sea level)	Total volume (million cubic meters)
0	285.05	127.823
1	284.05	113.443
2	283.05	100.529
3	282.05	88.684
4	281.05	77.644
5	280.05	67.374
6	279.05	57.904
7	278.05	49.286
8	277.05	41.604
9	276.05	34.851
10	275.05	28.908
11	274.05	23.697
12	273.05	19.129
13	272.05	15.108
14	271.05	11.664
15	270.05	8.889
16	269.05	6.774
17	268.05	5.202
18	267.05	4.023
19	266.05	3.072
20	265.05	2.280
21	264.05	1.637
22	263.05	1.112
23	262.05	.696
24	261.05	.387
25	260.05	.170
26	259.05	.029

Boundary and Initial Conditions

Boundaries of the Lake Hickory model include the reservoir bottom, the shoreline and tributary streams, the upstream boundary at the U.S. Highway 321 bridge, the downstream boundary at Oxford 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 with 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 Lake Hickory is unknown but is probably small relative to the flow in the reservoir. The reservoir bottom also is assumed to be immobile; that is, sediments are not resuspended by flow. The reservoir bottom causes resistance to the flow and thereby extracts energy from the mean flow. The Chezy formulation is used in CE-QUAL-W2 to describe this phenomenon and 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).

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 bottom 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 air temperature; a value of 15 °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 the Catawba River increases by 17 percent (from 2,830 to 3,416 km^2) between Rhodhiss Dam and Oxford Dam, with an increase of

Table 8. Hydraulic and thermal parameters specified as Lake Hickory model input

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

Parameter	Purpose	Value in Lake Hickory 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	7×10^{-8} $(watts/m^2)/°C$	Constant
Wind sheltering coefficient	Reduces measured wind speed to effective wind speed at water surface	0.85 (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

only 63 km^2 between Rhodhiss Dam and the U.S. Highway 321 bridge. For the purposes of the model, all of the tributary stream inflows from this 586-km^2 area into Lake Hickory were assumed to occur in the five branches of the model representing Gunpowder Creek, Upper Little River, Snow Creek, Middle Little River, and Long Shoals Creek (figs. 2 and 18). These tributaries are spatially distributed along the length of the reservoir and generally represent the largest tributary inflows to the reservoir.

The 586-km^2 contributing drainage area between Rhodhiss Dam and Oxford Dam was apportioned among the five branches (branches 2–6, fig. 18) based on the measured drainage areas of the streams. Inflows from the branches were estimated by using measured daily mean flows at Upper Little River and Middle Little River (sites 58 and 60, respectively, fig. 2) in cubic meters per second per square kilometer of drainage area, multiplied by the drainage area assigned to the individual branch. Inflows from the tributary streams were generally small relative to outflows from Rhodhiss Dam (fig. 4), and there was little justification for estimating inflows at a time interval of less than 1 day. The mean tributary inflow (total for all six branches) for the period April 1, 1993, through March 31, 1994, was estimated to be 9.6 m^3/s. The median total tributary inflow for the period was 7.3 m^3/s, and the total tributary inflow exceeded 19 m^3/s 10 percent of the time.

Hourly flows from Rhodhiss Dam were used as the upstream hydraulic boundary condition at the U.S. Highway 321 bridge. Flows from Rhodhiss Dam were obtained from Duke Power Company records of hourly power generation, and a relation between power generation and flow. Discharge measurements at two flows 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. 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. 19).

Hourly flows from Oxford Dam were used as the downstream hydraulic boundary condition. As with the Rhodhiss Dam flows, records of hourly power generation at Oxford Dam, obtained from Duke Power Company, were converted to flows using a relation between power generation and flow. Two measurements of flow downstream from Oxford Dam, made on April 21, 1994, were within 5 percent of the flow determined from power generation records. The median flow at Oxford Dam during the period April 1, 1993, through March 31, 1994, was 58.2 m^3/s; flows were greater than 108 m^3/s 20 percent of the time and were less than 1.2 m^3/s 10 percent of the time (fig. 19). Median flow during the period April 1993 through March 1994 at Oxford Dam was 16.6 percent greater than median flow at Rhodhiss Dam during the period. This increase in flow agrees well with the 17-percent increase in drainage area between Rhodhiss and Oxford Dams.

Water temperatures measured at the U.S. Highway 321 bridge at hourly intervals were used as the upstream thermal boundary condition. Water

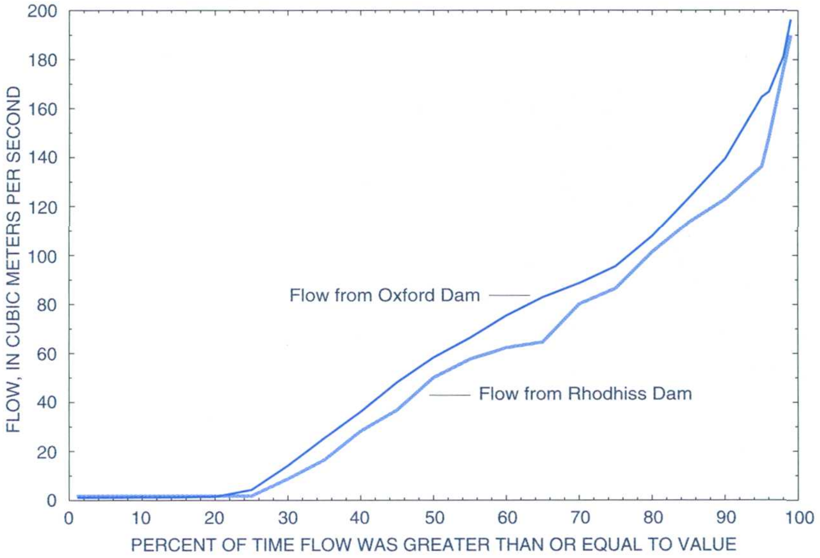


Figure 19. Flow durations for Catawba River at Rhodhiss Dam and Catawba River at Oxford Dam, April 1, 1993, through March 31, 1994.

temperature was measured near the water surface, at mid-depth, and near the streambed. Top-to-bottom differences in water temperature occasionally occurred (fig. 20), and these differences were greatest when water temperature was greater than about 18 °C. For this application, however, the upstream boundary condition was assumed to be isothermal, and the measured near-surface water temperature was used as the boundary condition. CE-QUAL-W2 allows a time series of vertically varying water temperature to be used as an upstream boundary condition, but this type of boundary condition requires about 10 times more thermal and water-quality boundary data than for the isothermal case because about 10 computational layers are at the upstream end of the reservoir. There was no indication that the assumption of an isothermal upstream boundary condition degraded the quality of the simulated reservoir temperatures. A downstream thermal boundary condition is not required. Daily mean water temperatures measured at the U.S. Highway 321 bridge were used as the required temperature boundary condition for each of the five tributary branches.

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 Oxford 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 relative to the total width of the dam (width of structure is less than 10 percent of total width of dam). The bottom of the withdrawal structure is at an elevation of 269.5 m, and the top of the structure is at 281.4 m (Jon Knight, written commun., September 1994). The top of the spillway is at an elevation of 285.04 m. The intake structures at Oxford Dam are located adjacent to the south 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.

Hydraulic 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 less than 2 km southwest of the headwaters of Lake Hickory, and were generally 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 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

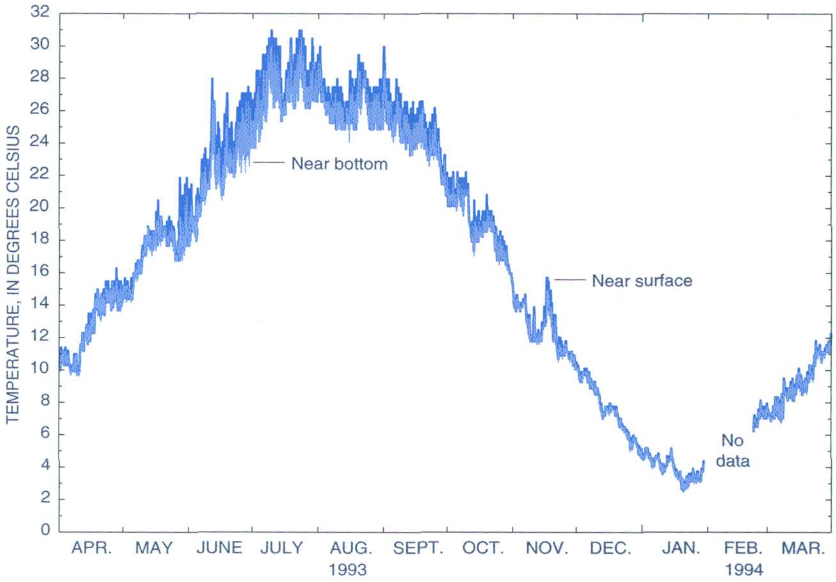


Figure 20. Measured near-surface and near-bottom water temperatures at the U.S. Highway 321 bridge, April 1, 1993, through March 31, 1994.

surface-water temperature also is required for the computation of surface heat exchange. Surface heat exchange can be computed from the heat exchange equation as the sum of seven separate terms, or by using a more simple linearization of the heat exchange 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 2 (Longview intake) and 3 (Hickory intake) (fig. 18). Withdrawal rates were assumed to be constant throughout the simulation period at 0.06 m^3/s (Longview) and 0.61m^3/s (Hickory), 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). Inputs from five permitted point sources, which are located in the Lake Hickory drainage downstream from Rhodhiss Dam, were included in the model (table 9). Data on flows and constituent concentrations from these dischargers were obtained from reported monthly compliance monitoring data (Sandra Gillaspie, North Carolina Department of Environment, Health, and Natural Resources, Raleigh, written commun., August 1994). Although the Rhodhiss WWTP discharge is located upstream from the model boundary, the discharge was assumed to be in model segment 2 (fig. 18) so that the effects of the discharge could be included in the reservoir model. Similarly, the Lenoir and Granite Falls outfalls are located in Gunpowder Creek upstream from Lake Hickory, but were placed in model segments 47 and 48, respectively (fig. 18). The temperature of each discharge was assumed to vary monthly and be approximately equal to the monthly mean air temperature. The Hickory Water Plant, which had an average daily discharge of 0.017 m^3/s (table 6), was not included in the model because nutrients and BOD were not monitored. It is likely that nutrient

Table 9. Permitted point sources in the Lake Hickory drainage basin downstream from Rhodhiss Dam

[WWTP, wastewater-treatment plant]

Facility	Model segment location of outfall (fig. 18)	Mean flow (cubic meters per second)
Rhodhiss WWTP	2	0.002
Huffman Finishing outfall	3	.01
Hickory Northeast WWTP	22	.14
Lenoir-Gunpowder Creek WWTP	47	.05
Granite Falls WWTP	48	.02

concentrations in the discharge were equal to or less than nutrient concentrations in Lake Hickory.

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 less than 0.2 percent of the April 1993 through March 1994 median outflow rate. Consequently, time series of precipitation on and evaporation from the reservoir were not included as model boundary conditions. The loss of heat from the reservoir resulting from evaporation was, however, included in the heat balance.

Chemical Boundary Conditions

Concentrations of the following constituents were simulated for Lake Hickory: labile dissolved organic matter, refractory dissolved organic matter, algae, particulate organic matter, NO_2+NO_3, NH_4, PO_4, DO, organic bottom sediments, iron, and BOD. These are the same 11 chemical constituents included in the Rhodhiss Lake model (Giorgino and Bales, 1997).

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 that are included in model simulations.

Chemical boundary conditions at the U.S. Highway 321 bridge (site 29, fig. 2) were based on monthly to semi-monthly measurements (Jaynes, 1994). As explained below, flow was not a good predictor of constituent concentrations at the U.S. Highway 321 bridge, but concentrations of NO_2+NO_3, NH_4, and PO_4 did have a fairly predictable temporal pattern (figs. 3, 11, 12, 14, and 15).

The relation between constituent concentrations and flow at the U.S. Highway 321 bridge was not as strong as that at the headwaters of Rhodhiss Lake (Giorgino and Bales, 1997) for several reasons. First, flows into Lake Hickory were controlled by releases from Rhodhiss Dam rather than natural runoff conditions. High flows from the dam did not necessarily coincide with storm events when constituent concentrations should be elevated. However, because the volume of Rhodhiss Lake was fairly small relative to the inflow, releases from Rhodhiss Dam did vary seasonally in a manner similar to an unregulated river (fig. 4, site 28). Somewhat higher, more sustained releases occurred in the winter and early spring compared to the rest of the year. More hydropower peaking occurred in the summer and early fall, resulting in short periods of high flows but lower daily mean flows compared to the rest of the year.

The second reason that flows were not a good predictor of concentrations at the U.S. Highway 321 bridge was that summertime withdrawals from Rhodhiss Lake were from the hypolimnion of the reservoir. Constituent concentrations in the hypolimnion of Rhodhiss Lake were controlled primarily by oxygen dynamics and, to a lesser extent, water temperature. Consequently, Rhodhiss Lake hypolimnetic waters were high in NH_4 and PO_4, and low in NO_2+NO_3 concentrations during the summer and fall (Giorgino and Bales, 1997).

Finally, reservoir processes appeared to affect concentrations in Lake Hickory at the U.S. Highway 321 bridge. At this site, measured concentrations of NO_2+NO_3, NH_4, and PO_4 were essentially the same near the surface and near the bottom except during May through September 1993 when near-surface concentrations generally were less than near-bottom concentrations (figs. 11, 12, 14, and 15). This observation suggests that several processes may have occurred as water moved from Rhodhiss Dam to the U.S. Highway 321 bridge. The colder Rhodhiss Lake releases, with high summer concentrations of PO_4 and NH_4 (Giorgino and Bales, 1997), may have plunged beneath the warmer surface waters at U.S. Highway 321, where there was generally less than a 2-°C difference between near-surface and near-bottom water temperature. Phytoplankton production may have utilized near-surface nutrients during the summer; chlorophyll *a* values generally were high at site 29 (fig. 16). NH_4 and PO_4 may have been released from bottom sediments, although near-bottom oxygen concentrations were never measured to be less than 2.6 mg/L at the site. Finally, NH_4 may have been oxidized in surface waters.

Boundary data at the upstream end of the reservoir and the tributaries included algae, detritus (particulate organic matter), NO_2+NO_3, NH_4, and PO_4, DO, iron, and BOD. Concentrations of TSS, TN, NH_4, TP, and DO were available at monthly intervals from the five point sources (table 9). Because no PO_4 data were available for the point sources, PO_4 concentrations in these discharges initially were assumed to be equal to reported TP concentrations.

Exchange of carbon dioxide and DO occurs at the air-water interface. Exchange rates are governed by wind speed, water temperature, reservoir elevation, and the molecular diffusivity of the gases. The CE-QUAL-W2 surface-exchange algorithm was modified for this application to increase reaeration rates because simulated DO concentrations near the surface were generally 1 to 2 mg/L less than measured values before the algorithm was modified. The algorithm used for the Lake Hickory application was the same as that used for the Rhodhiss Lake application (Giorgino and Bales, 1997).

It was assumed that atmospheric inputs of all simulated constituents to the reservoir were negligible. Data from the National Atmospheric Deposition Program indicated that the deposition of NH_4 on Lake Hickory was approximately 6,630 kilograms (kg) as N, and deposition of NO_3 was approximately 4,230 kg as N from April 1993 through March 1994. Inorganic nitrogen deposition was about 12 percent of the point-source TN loads (table 6) and about 1 percent of the TN load at site 28 (table 5) during the same period. No deposition data for phosphorus were available. Future applications of the model might benefit from including atmospheric inputs. Constituent inputs from the bottom boundary generally were computed within the model and were based on the value of selected parameters (table 10) and the concentration of the constituent in the overlying waters.

Initial Conditions

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

Table 10. Rate coefficients used in water-chemistry simulations and specified as model input

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

Parameter	Computational purpose	Value in Rhodhiss Lake model	Value in Lake Hickory model
Light extinction coefficient for water	Amount of solar radiation absorbed in the surface layer	0.5/m	0.45/m
Light extinction coefficient for organic solids	Amount of solar radiation absorbed in the surface layer	0.2 $(m^3/m)/g$	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*	0.4*
Suspended solids settling rate	Settling rates and sediment accumulation on reservoir bottom	2 m/d	1 m/d
Algal growth rate	Maximum gross algal production rate, uncorrected for respiration, mortality, excretion, or settling; temperature dependent	1.9/d	1.2/d
Algal mortality rate	Maximum algal mortality rate; temperature dependent	0.09/d	0.075/d
Algal excretion rate	Maximum algal photorespiration rate, which becomes labile dissolved organic matter	0.005/d	0.010/d
Algal dark respiration rate	Maximum algal dark respiration rate	0.005/d	0.010/d
Algal settling rate	Representative settling velocity for algal assemblages	0.1 m/d	0.2 m/d
Saturation light intensity	Saturation light intensity at maximum algal photosynthesis rate	150 watts/m^2	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*	0.8*
Lower temperature for algal growth	Algal growth rate as a function of water temperature	10 °C	8 °C
Fraction of algal growth at lower temperature	Algal growth rate as a function of water temperature	0.1*	0.2*
Lower temperature for maximum algal growth	Algal growth rate as a function of water temperature	22 °C	24 °C
Fraction of maximum growth at lower temperature	Algal growth rate as a function of water temperature	0.99*	0.99*
Upper temperature for maximum algal growth	Algal growth rate as a function of water temperature	22.5 °C	30.1 °C
Fraction of maximum growth at upper temperature	Algal growth rate as a function of water temperature	0.95*	0.95*
Upper temperature for algal growth	Algal growth rate as a function of water temperature	35 °C	35 °C
Fraction of algal growth at upper temperature	Algal growth rate as a function of water temperature	0.1*	0.1*
Labile dissolved organic matter decay rate	Dissolved oxygen loss and production of inorganic carbon, ammonia, and phosphate from algal decay; temperature dependent	0.04/d	0.09/d

Table 10. Rate coefficients used in water-chemistry simulations and specified as model input—Continued

[m, meters; (m^3/m)/g, cubic meter per meter per gram; *, dimensionless parameter; m/d, meters per day; d, day; watts/m^2, watts per square meter; °C, degrees Celsius; (g/m^2)/d, grams per square meter per day; BOD, biochemical oxygen demand; g/m, grams per meter; mg/L, milligrams per liter]

Parameter	Computational purpose	Value in Rhodhiss Lake model	Value in Lake Hickory model
Labile to refractory decay rate	Transfer of labile to refractory dissolved organic matter	0.005/d	0.002/d
Maximum refractory dissolved organic matter decay rate	Dissolved oxygen loss and production of inorganic carbon, ammonia, and phosphate from decay of refractory dissolved organic matter; temperature dependent	0.001/d	0.001/d
Detritus decay rate	Dissolved oxygen loss and production of inorganic carbon, ammonia, and phosphate from decay particulate organic matter; temperature dependent	0.002/d	0.010/d
Detritus settling velocity	Loss of particulate organic matter to bottom sediment	2.5 m/d	1.0 m/d
Lower temperature for organic matter decay	Organic matter decay as a function of temperature	5 °C	5 °C
Fraction of organic matter decay at lower temperature	Organic matter decay as a function of temperature	0.05*	0.05*
Lower temperature for maximum organic matter decay	Organic matter decay as a function of temperature	25 °C	25 °C
Fraction of maximum organic matter decay at lower temperature	Organic matter decay as a function of temperature	0.95*	0.95*
Sediment decay rate	Decay rate of organic matter in bed sediments	0.015/d	0.005/d
Sediment oxygen demand	Zero-order sediment oxygen demand for each computational segment	0 (g/m^2)/d	0.1–1.9 (g/m^2)/d
5-day BOD decay rate	Effects of BOD loading on dissolved oxygen	0.15/d	0.25/d
BOD temperature rate coefficient	Adjusts 5-day BOD decay rate at 20 °C to ambient temperature	1.047*	1.047*
Ratio of 5-day BOD to ultimate BOD	Effects of BOD loading on dissolved oxygen	1.20*	1.85*
Release rate of phosphorus from bottom sediments	Phosphorus balance; computed as a fraction of the sediment oxygen demand	0.005*	0.0001*
Phosphorus partitioning coefficient	Describes sorption of phosphorus on to suspended solids	3.0*	2.0*
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	0.005 g/m
Release rate of ammonia from bottom sediments	Nitrogen balance; computed as a fraction of the sediment oxygen demand	0.003*	0.040*
Ammonia decay rate	Rate at which ammonia is oxidized to nitrate	0.2/d	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*	0.014*
Lower temperature for ammonia decay	Ammonia nitrification as a function of temperature	5 °C	10 °C
Fraction of nitrification at lower temperature	Ammonia nitrification as a function of temperature	0.1*	0.1*
Lower temperature for maximum ammonia decay	Ammonia nitrification as a function of temperature	25 °C	25 °C
Fraction of maximum nitrification at lower temperature	Ammonia nitrification as a function of temperature	0.99*	0.99*
Nitrate decay rate	Rate at which nitrate is denitrified; temperature dependent	0.15/d	2.0/d
Lower temperature for nitrate decay	Denitrification as a function of temperature	5.0 °C	5.0 °C
Fraction of denitrification at lower temperature	Denitrification as a function of temperature	0.1*	0.2*
Lower temperature for maximum nitrate decay	Denitrification as a function of temperature	25 °C	30 °C
Fraction of maximum denitrification at lower temperature	Denitrification as a function of temperature	0.99*	0.99*
Iron release rate from bottom sediments	Iron balance; computed as fraction of sediment oxygen demand	1.0*	1.0*
Iron settling velocity	Particulate iron settling velocity under oxic conditions	2.0 m/d	2.0 m/d
Oxygen stoichiometric equivalent for ammonia decay	Relates oxygen consumption to ammonia decay	4.0*	4.57*
Oxygen stoichiometric equivalent for organic matter decay	Relates oxygen consumption to decay of organic matter	1.5*	1.4*
Oxygen stoichiometric equivalent for dark respiration	Relates oxygen consumption to algal dark respiration	0.9*	1.4*
Oxygen stoichiometric equivalent for algal growth	Relates oxygen production to algal growth	3.0*	3.0*
Stoichiometric equivalent between organic matter and phosphorus	Relates phosphorus release to decay of organic matter	0.009*	0.011*
Stoichiometric equivalent between organic matter and nitrogen	Relates nitrogen release to decay of organic matter	0.08*	0.08*
Dissolved oxygen limit	Dissolved oxygen concentration below which anaerobic processes such as nitrification and sediment nutrient releases occur	0.1 mg/L	0.1 mg/L

at the U.S. Highway 321 bridge for the day simulations were to begin. The time required for the effects of the estimated initial conditions to disappear from the model domain was generally 3 to 10 days; however, the time is less if model initial conditions closely represent actual reservoir conditions.

Model Parameters

Parameters are used to describe physical and chemical processes that are not explicitly modeled and 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 that 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 are 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 10). Most of the coefficients were based on suggestions given by Cole and Buchak (1995), 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 Oxford Dam which remain constant between measured values.

Model Calibration and Testing

Model calibration was achieved by adjusting 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 the U.S. Highway 321 bridge were not available until March 11, 1993. Moreover, high flows during late March resulted in flows over the spillway at Oxford Dam, and flow data at the dam were not available for these periods when the water level was above the crest of the spillway. The April 1, 1993, through March 31, 1994, calibration and testing period also is consistent with the period used for the Rhodhiss Lake model (Giorgino and Bales, 1997).

Hydrodynamics and Heat Transport

There was good agreement between measured and simulated water levels at Oxford Dam (fig. 21). Some adjustments in Oxford outflow rates were required to achieve the good agreement between measured and simulated water levels; the mean of the absolute value of the adjustments was about 7 percent. The root mean square difference between measured and simulated water levels was 0.046 m. The differences between measured and simulated water levels ranged from 0.18 to -0.20 m, and the mean difference was -0.012 m. Ninety percent of the differences between measured and simulated water levels were between 0.06 and -0.08 m. The total range in measured water level during the simulation period was 1.14 m.

Simulated water temperature is a function of simulated concentration of organic solids (algae and detritus). The organic solids affect light penetration and, consequently, heat distribution in the reservoir. Hence, water temperature calibration cannot be performed independently of 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.

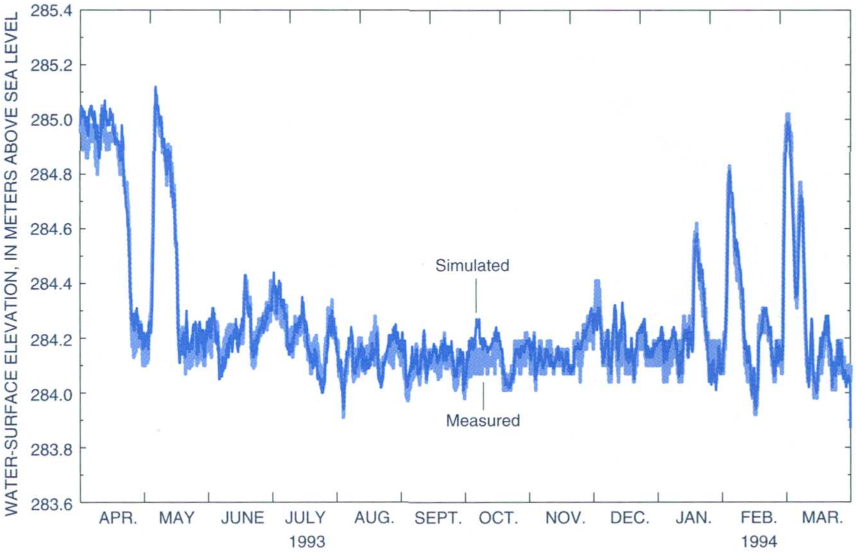


Figure 21. Measured and simulated water levels at Oxford Dam, April 1, 1993, through March 31, 1994.

Simulated water temperatures reproduced seasonal variations in near-surface and near-bottom water temperature measured in Lake Hickory (fig. 22) from the headwaters (fig. 22A) to the forebay (fig. 22D). Simulated near-surface and near-bottom water temperatures were generally within 1 °C of corresponding measured values.

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 are typically separated by a thermocline, in which there is a relatively large change in temperature over a small depth. A strong

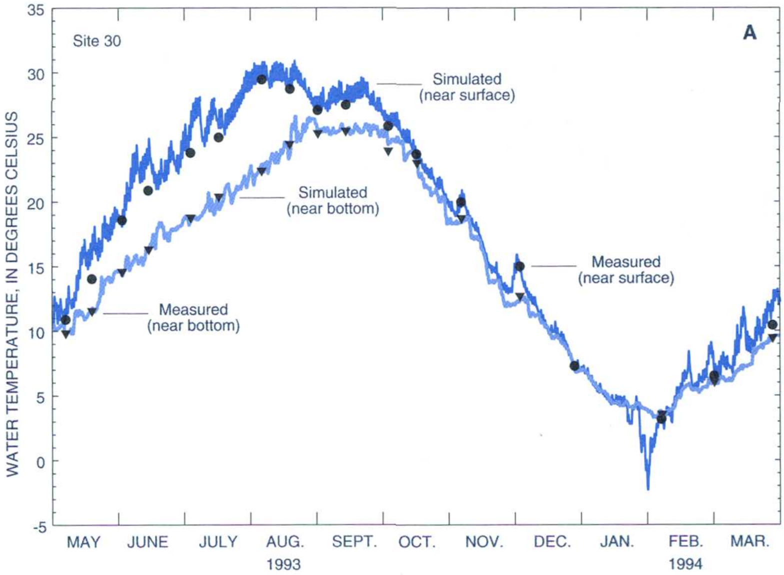


Figure 22. Measured and simulated near-surface and near-bottom water temperatures for May 1, 1993, through March 25, 1994, at Lake Hickory sites (A) 30, (B) 34, (C) 36, and (D) 40.

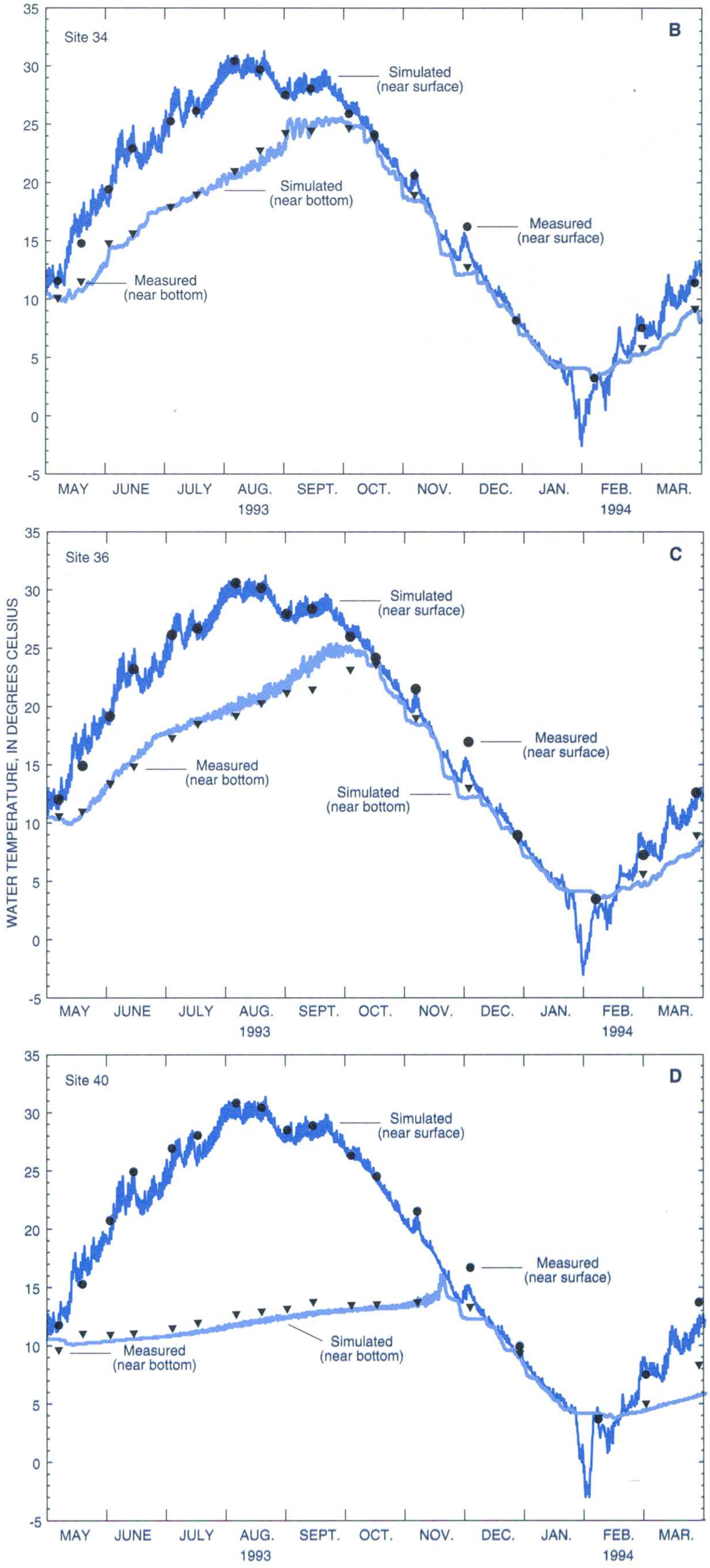


Figure 22. (Continued) Measured and simulated near-surface and near-bottom water temperatures for May 1, 1993, through March 25, 1994, at Lake Hickory sites (A) 30, (B) 34, (C) 36, and (D) 40.

thermocline rarely existed in Lake Hickory (fig. 23), although top-to-bottom water temperature differences exceeded 15 °C at times near the dam (for example, July 13, 1993 (fig. 23C)). During April 20 through October 19, 1993, vertical temperature distributions at sites 34 (upper mid-reservoir), 36 (lower mid-reservoir), and 40 (forebay) varied from nearly isothermal conditions to strongly stratified, and back to very small top-to-bottom differences in water temperature (fig. 23). Simulated vertical distributions agreed quite closely with measured distributions, even for complex temperature profiles (for example, June 8, 1993, at site 40 (fig. 23C)).

All measured water temperatures (288 observations) for April 1993 through March 1994 were compared with corresponding simulated values for site 36. Measured water temperatures at site 36 ranged from about 4 to 31 °C, although simulations indicated a greater range (fig. 22C). The mean difference between measured and simulated water temperatures at site 36 was -0.05 °C, and the median difference was zero, indicating no bias toward over or underprediction of temperatures at this site during the calibration period. Eighty-five percent of the simulated temperatures were within 1 °C of the measured temperature.

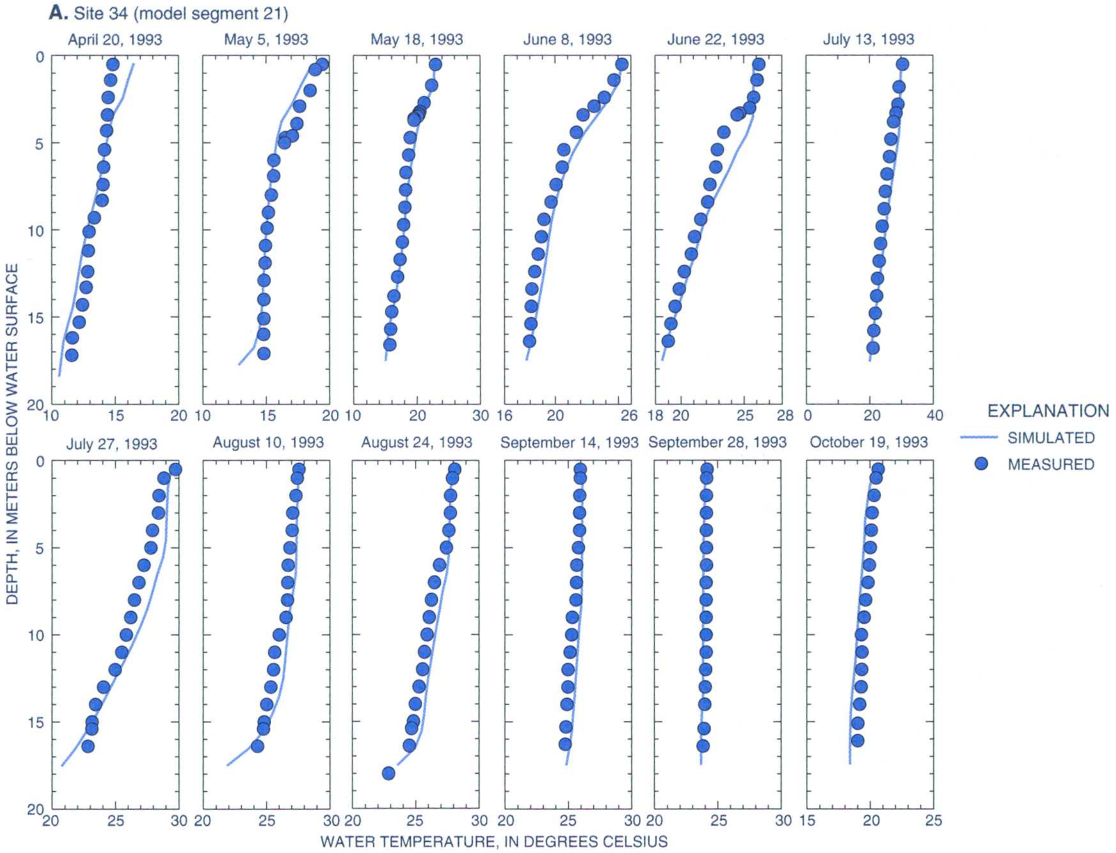
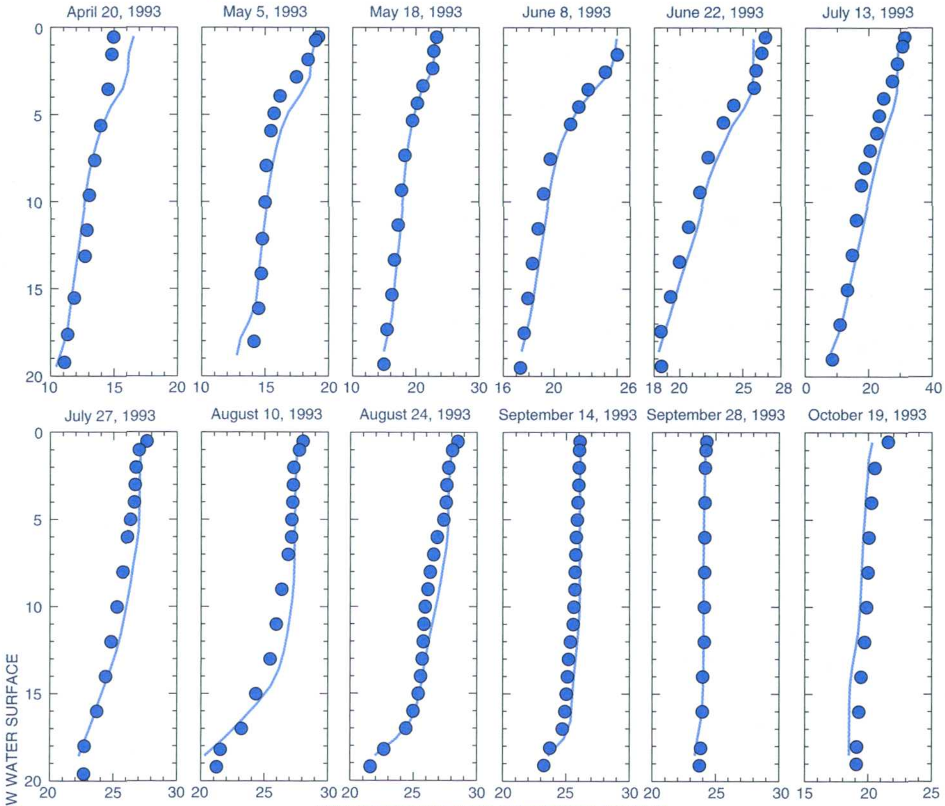


Figure 23. Measured and simulated vertical profiles of water temperature at Lake Hickory sites (A) 34, (B) 36, and (C) 40, April 20 through October 19, 1993.

B. Site 36 (model segment 31)



C. Site 40 (model segment 41)

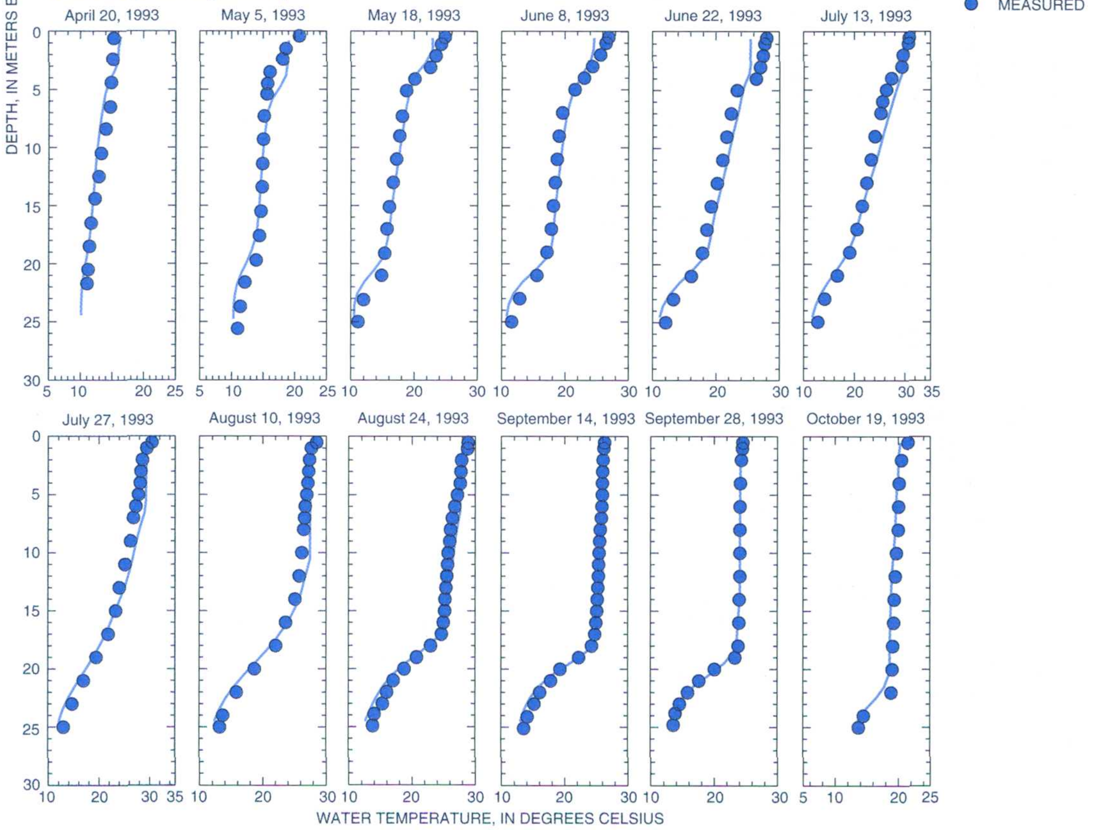


Figure 23. (Continued) Measured and simulated vertical profiles of water temperature at Lake Hickory sites (A) 34, (B) 36, and (C) 40, April 20 through October 19, 1993.

For site 36, simulated water temperatures generally were overpredicted more often when measured temperatures were greater than about 23 °C (fig. 24A), and were somewhat overpredicted in the fall of 1993 (fig. 24B). Errors in simulated temperatures were not related to the depth at which measurements were made (fig. 24C). In general, the calibrated model provides an excellent simulation of water temperature in Lake Hickory, with most of the simulated values within plus or minus 1 °C of the actual value. Water temperature simulations for Lake Hickory were somewhat better than similar simulations for Rhodhiss Lake (Giorgino and Bales, 1997).

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 suggest that the reservoir did not become thermally mixed at site 40 until perhaps mid-December. The simulations, however, indicate that the reservoir was mixed in early November. A thermal

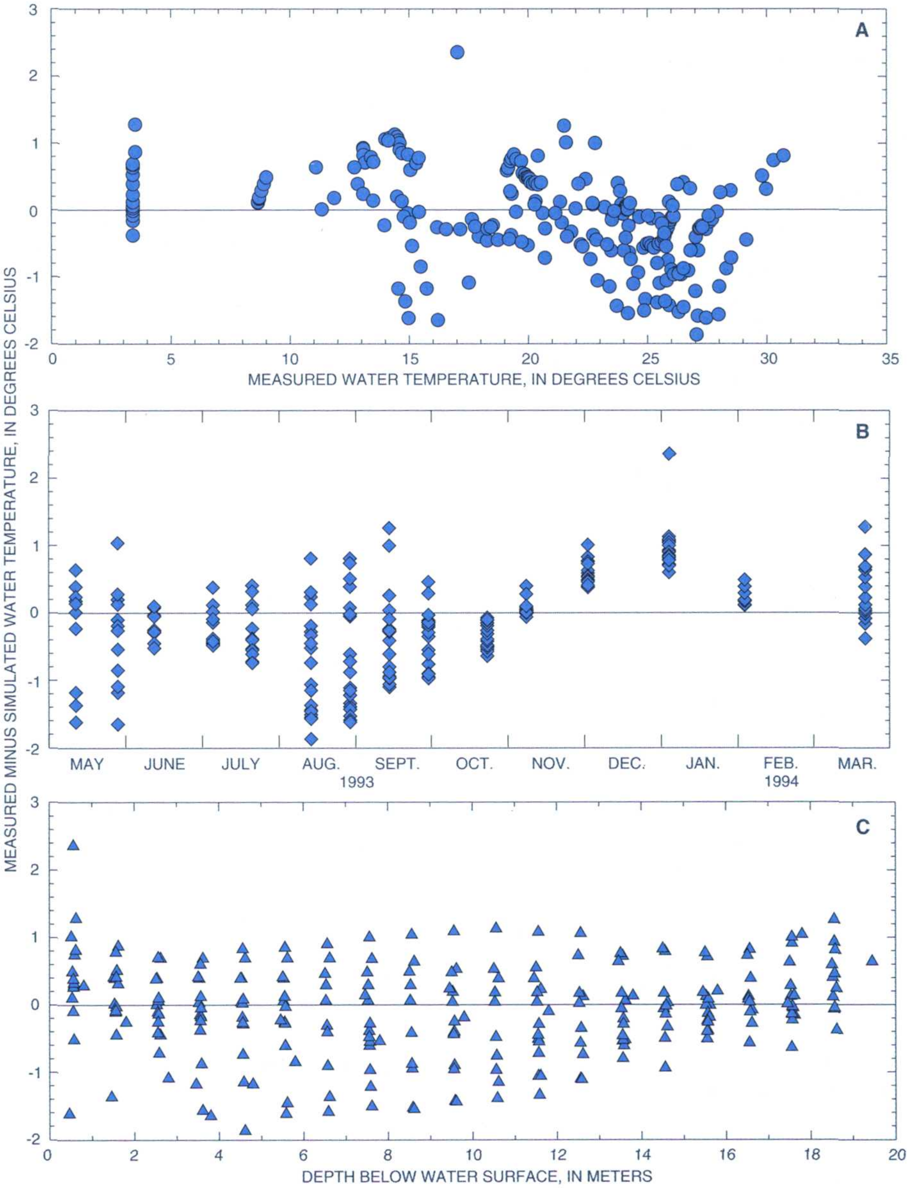


Figure 24. Relation of difference between measured and simulated water temperatures at Lake Hickory site 36 to (A) measured water temperature, (B) date, and (C) water depth, May 1, 1993, through March 25, 1994.

inversion (near-surface water temperature less than near-bottom water temperature) which occurred in January 1994 (fig. 22) 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 much more gradually than those in the shallower reaches (fig. 22).

Water Chemistry

The simulated complex biochemical reactions affecting the 11 water-chemistry constituents and physical transport processes in the Lake Hickory model are aggregated in the simulated dissolved oxygen concentrations. The North Carolina ambient water-quality standard for DO is a daily average of at least 5 mg/L and a minimum instantaneous value of 4 mg/L (North Carolina Department of Environment, Health, and Natural Resources, 1997). Consequently, results of the DO simulations are emphasized in the following discussion. Of the other 10 simulated water-chemistry parameters, only 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, 1997). Algal biomass also is used as an indicator of the trophic state of a reservoir. Hence, additional information on simulated algae is presented. Algal growth in Lake Hickory appears to be limited by PO_4 concentrations and light availability. Nitrate and ammonia concentrations appear to have only a limited effect on DO and algal concentrations in Lake Hickory, although nitrate concentrations decrease during periods of high phytoplankton growth. Oxygen is used during the decay of labile and refractory dissolved organic matter, BOD, organic sediments, and detritus. Decay processes also release phosphate and ammonia to the system, and particulate matter affects light penetration and heat distribution.

Dissolved Oxygen

Simulated DO concentrations for Lake Hickory exhibited the same general patterns and magnitudes as measured values (fig. 25). The onset of low DO condition and the recovery to higher DO levels were well simulated throughout the reservoir (fig. 25).

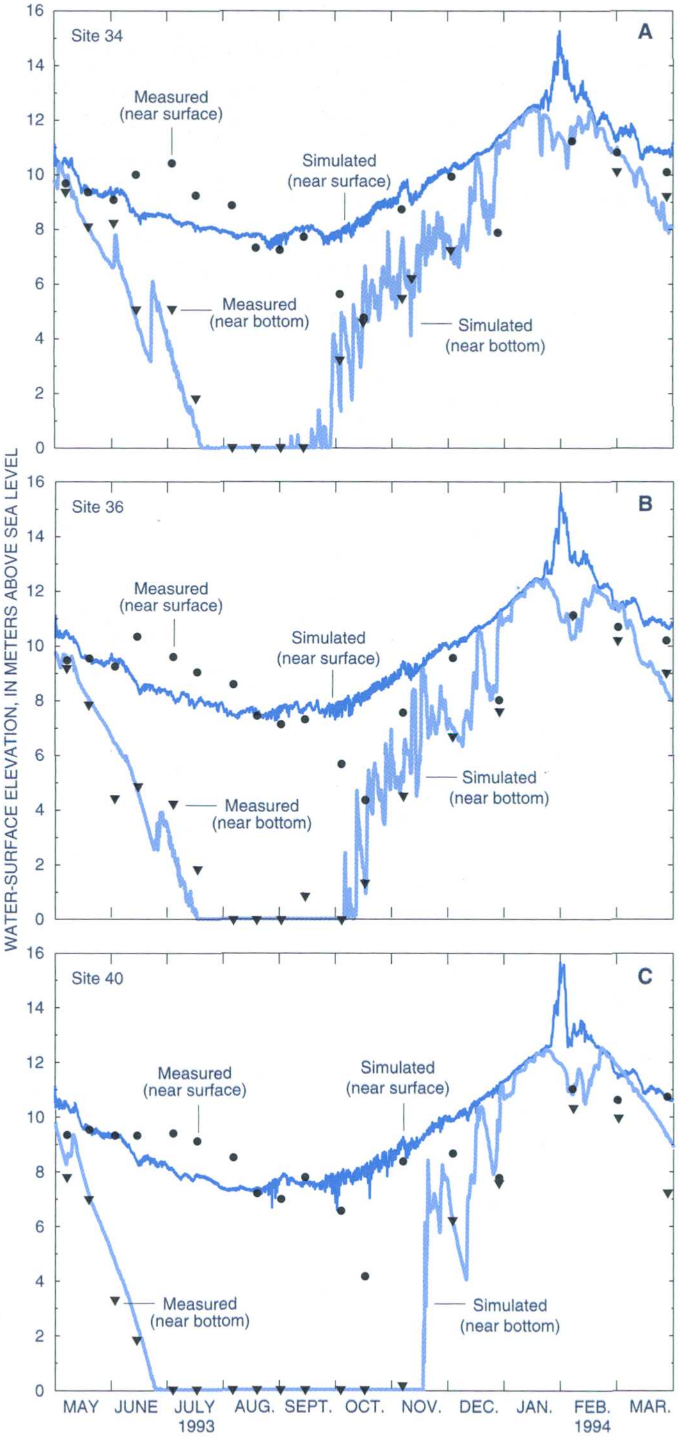


Figure 25. Measured and simulated dissolved oxygen concentrations, May 1, 1993, through March 25, 1994, at Lake Hickory sites (A) 34, (B) 36, and (C) 40.

Near-surface DO concentrations were undersimulated by about 1 to 1.5 mg/L for late May through early July. Near-surface DO concentrations were oversimulated for both of the September 1993 measurements, as well as the December 1993 measurement. Measured near-surface DO concentrations were somewhat low on these three dates (fig. 26). DO concentrations were generally less than 80 percent of saturation concentration on September 14, and generally

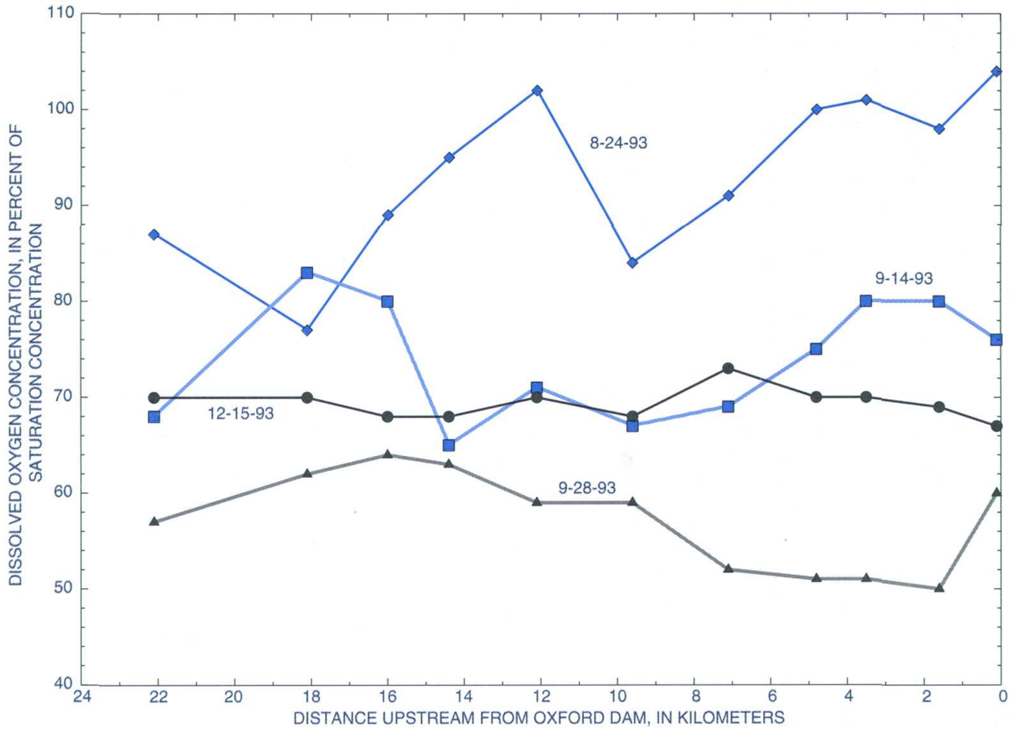


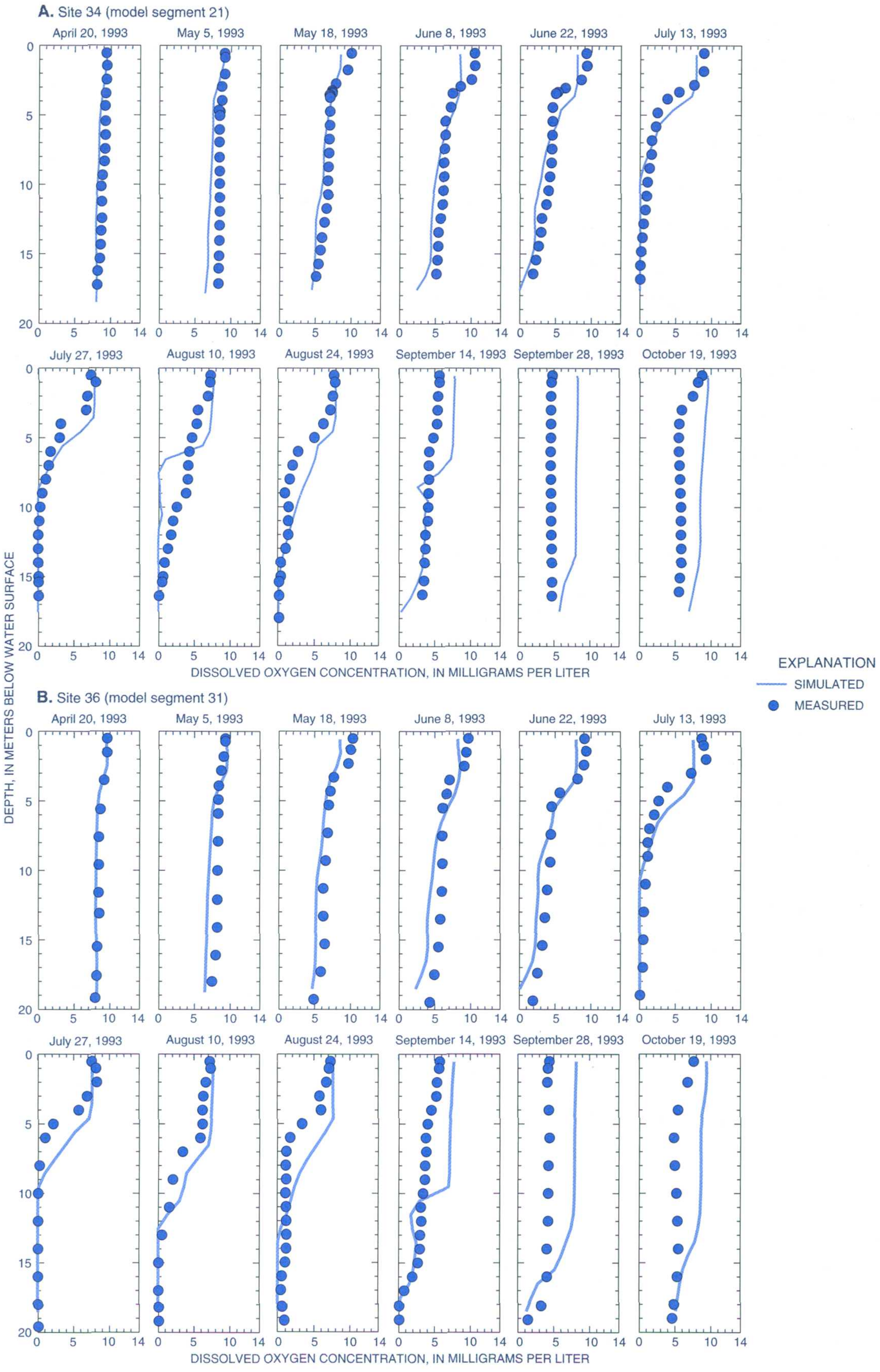
Figure 26. Measured longitudinal profile of surface dissolved oxygen concentration, in percent of saturation concentration, August 24, September 14, September 28, and December 15, 1993.

less than 60 percent of saturation on September 28, whereas concentrations were greater than 90 percent of saturation on August 24. Chlorophyll *a* concentrations for the August 24 and September 14 observations were about equal, and chlorophyll *a* concentrations on September 28 were fairly high; somewhat higher DO concentrations generally would be expected in daylight hours during periods of higher chlorophyll *a* levels because of increased photosynthetic activity.

Temperature data suggest that vertical mixing occurred in September throughout much of the reservoir. In the upstream two-thirds of the reservoir, near-surface and near-bottom water temperatures were nearly the same by September 28. However, the reservoir remained strongly stratified in about the downstream one-third of the lake (fig. 22D).

Given these observations, at least two possible reasons exist for the oversimulation of near-surface DO concentrations. First, it is possible that there was an undetected, large influx of oxygen-demanding material to Lake Hickory which was not included as part of the model boundary conditions. Second, it is possible that the hypolimnetic volume of the reservoir was underestimated by the computational grid of the model, although the total volume of the reservoir was represented fairly well. This underestimation of the hypolimnetic volume would lead to an underestimation of the volume of low DO hypolimnetic water which mixed with the epilimnion in late September; thus, by mass balance, DO concentrations in the epilimnion would be oversimulated. It is of note that hypolimnetic DO concentrations during September and December (fig. 25) were correctly simulated, although near-surface DO concentrations were oversimulated. The underestimation of hypolimnetic volume could explain the oversimulation of near-surface DO concentrations in late September but could not explain the December oversimulation because the reservoir appears to have remained thermally mixed during much of November and December.

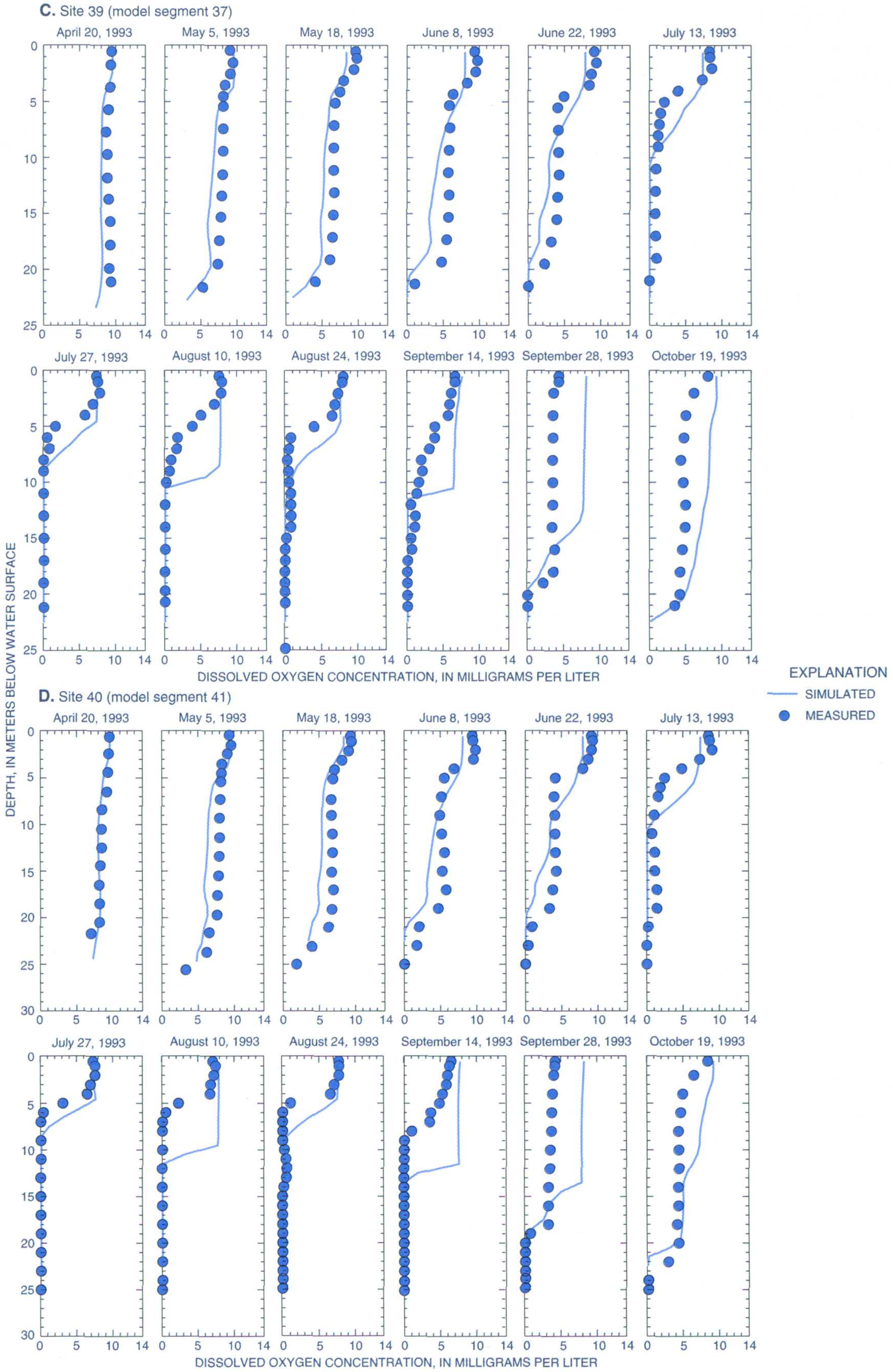
Simulated vertical profiles of DO concentrations generally agreed with measurements (fig. 27). Agreement was better at upstream sites 34 and 36 (fig. 27A and B, respectively) than near the dam (site 40, fig. 27D). Simulated values had the greatest departures from measured values for the late summer and early fall (generally July 27 through October 19, 1993). Although a clearly defined thermocline was



A. Site 34 (model segment 21)

B. Site 36 (model segment 31)

Figure 27. Measured and simulated vertical profiles of dissolved oxygen concentrations at Lake Hickory sites (A) 34, (B) 36, (C) 39, and (D) 40, April 20 through October 19, 1993.



C. Site 39 (model segment 37)

D. Site 40 (model segment 41)

Figure 27. (Continued) Measured and simulated vertical profiles of dissolved oxygen concentrations at Lake Hickory sites (A) 34, (B) 36, (C) 39, and (D) 40, April 20 through October 19, 1993.

seldom measured, strong oxyclines were evident on a number of occasions throughout the reservoir (for example, July 27, 1993, at site 40 (figs. 23C and 27D)). However, simulated oxyclines were typically more abrupt than measured oxyclines, particularly at mid-reservoir sites 36 and 39 (figs. 27B and 27C).

Differences between measured DO concentrations at site 36 and simulated DO concentrations at the corresponding time and depth were calculated for the period April 20 through December 15, 1993. These differences were compared to the corresponding measured DO concentrations, to the measurement date, and to the depth of measurement (fig. 28). There appeared to be a greater tendency toward overprediction than underprediction, except when DO values were greater than about 8 mg/L (fig. 28A). The most biased results and most of the large differences between measured and simulated DO concentrations occurred in the late summer and fall of 1993 (fig. 28B). The magnitude of the error between measured and simulated DO concentrations did not appear to be related to the depth below the water

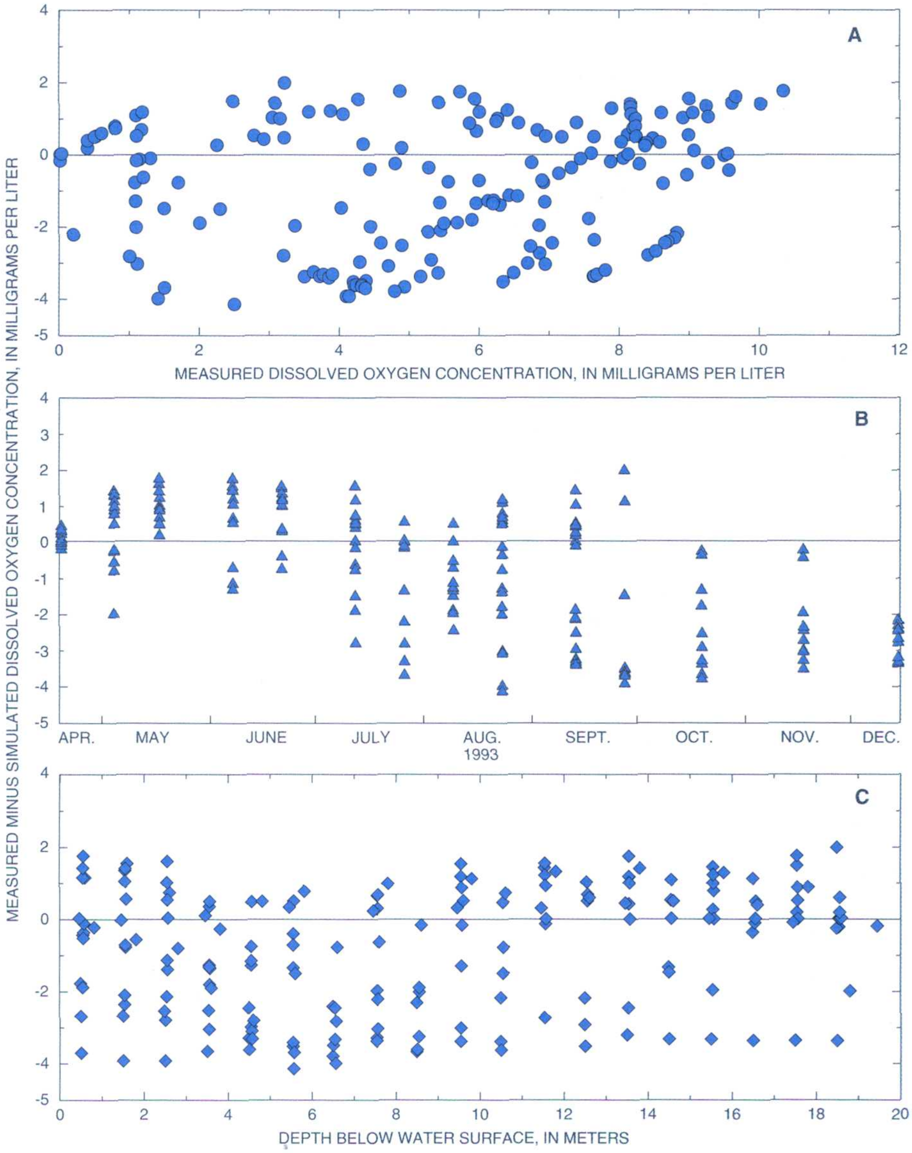


Figure 28. Relation of difference between measured and simulated dissolved oxygen concentrations at Lake Hickory site 36 to (A) measured dissolved oxygen concentration, (B) date, and (C) water depth, April 20 through December 15, 1993.

surface at which the comparison between measured and simulated values was made (fig. 28C). In general, DO simulations for Lake Hickory were better than those for Rhodhiss Lake (Giorgino and Bales, 1997), particularly with regard to the overall magnitude of the errors and to the simulation of vertical distribution of DO concentrations.

The cumulative frequency of occurrence of all measured DO concentrations at site 36 was determined and plotted (fig. 29). A similar analysis was performed using the simulated DO values which correspond to times and locations at which DO measurements were made. The frequency of occurrence of DO concentrations less than 4 mg/L was almost the same for measured (32 percent of the time) and simulated (28 percent of the time) conditions. For DO concentrations between about 4 and 11 mg/L, simulated DO values were greater than measured DO values, as already demonstrated. For example, about 60 percent of the time, simulated DO values were less than or equal to about 7.5 mg/L, but measured DO values were less than or equal to about 6.5 mg/L. There generally is more interest on the part of water-resources managers in the occurrence of DO concentrations less than 4 or 5 mg/L than in higher values.

From an analysis of simulated hourly near-bottom DO concentrations, concentrations less than 4 mg/L occurred about 32 percent of the time during April 1993 through March 1994 at site 34, and about 52 percent of the time at site 40 near Oxford Dam (fig. 30). Similarly, near-bottom DO concentrations of 1 mg/L or less occurred about 22 percent of the time at site 34, 28 percent of the time at site 36, and 46 percent of the time at site 40 (fig. 30). Near-bottom DO concentrations of about 8 mg/L and greater occurred about the same amount of time throughout the reservoir, regardless of location.

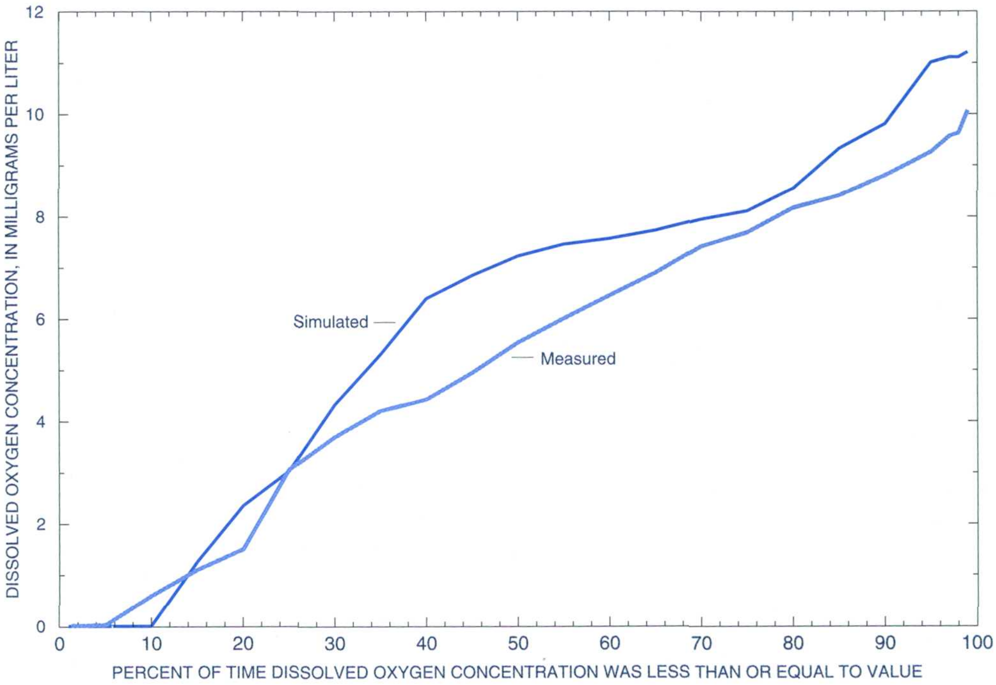


Figure 29. Measured and simulated frequencies of occurrence of dissolved oxygen concentrations at Lake Hickory site 36, April 20 through December 15, 1993.

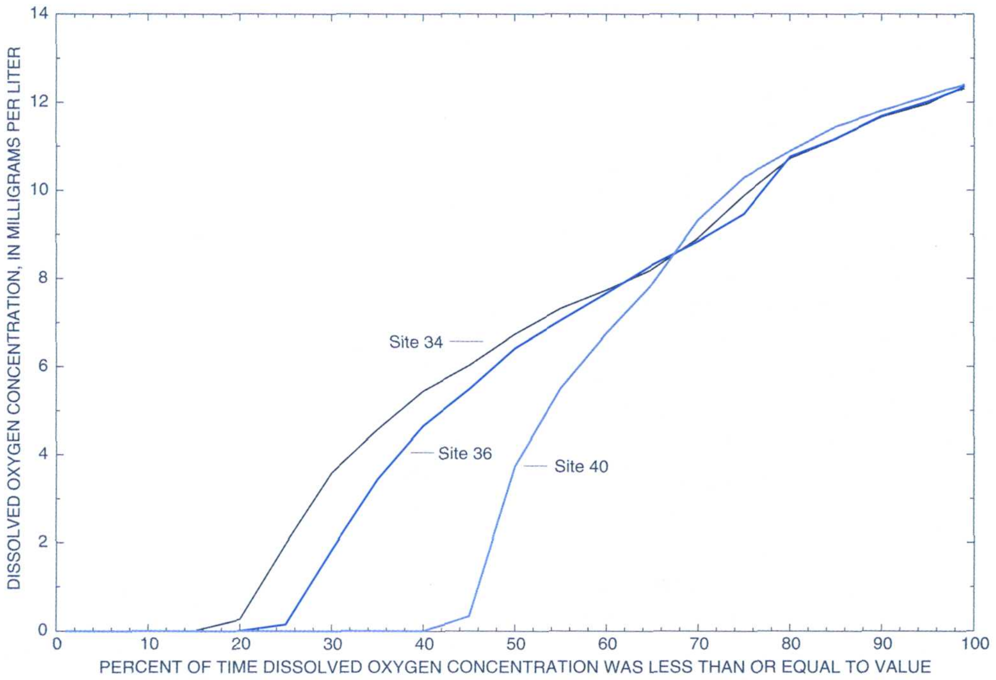


Figure 30. Simulated frequencies of occurrence of near-bottom dissolved oxygen concentrations at Lake Hickory sites 34, 36, and 40 as determined from simulated hourly dissolved oxygen concentrations, April 1993 through March 1994.

Algae and Phosphorus

Simulated algal concentrations were similar to measured values at site 34, with two exceptions (fig. 31A). Algal concentrations at the mid-reservoir site were undersimulated in July 1993, although bottom PO_4 and surface NO_2+NO_3 and NH_4 concentrations were oversimulated. It might be expected that an oversimulation of nutrient concentrations would lead to an oversimulation of algal biomass rather than an undersimulation. In addition, measured DO concentrations at site 34 in July 1993 were not particularly high, as might be expected during an algal bloom, and simulated DO concentrations agreed very closely with measured values for these dates (fig. 25A). Algal concentrations were simulated well at site 40A near Oxford Dam. The only significant differences between measured and simulated concentrations were for the May observations, when algal concentrations were oversimulated, and September, when one value was oversimulated and the other measured value was undersimulated (fig. 31B). The simulated timing of the late April bloom was about 2 weeks late, but the magnitude of the simulated biomass was fairly accurate.

Accurate simulations of algal concentrations are very difficult for several reasons. First, algae are not uniformly distributed vertically or horizontally in the reservoir but usually occur in patches. Consequently, obtaining a representative sample can be difficult. For example, on November 17, 1993, in Rhodhiss Lake, USGS data indicated that the chlorophyll *a* concentration was 52 µg/L, but a sample collected by the North Carolina Division of Water Quality (DWQ, formerly known as the Division of Environmental Management) on the same day and within 100 m of the USGS sampling location contained a chlorophyll *a* concentration of 13 µg/L (Giorgino and Bales, 1997). Although different sampling methodologies and analytical procedures were used by the USGS and DWQ, these differences do not explain the large difference in measured chlorophyll *a*. Second, phytoplankton are simulated by the Lake Hickory 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

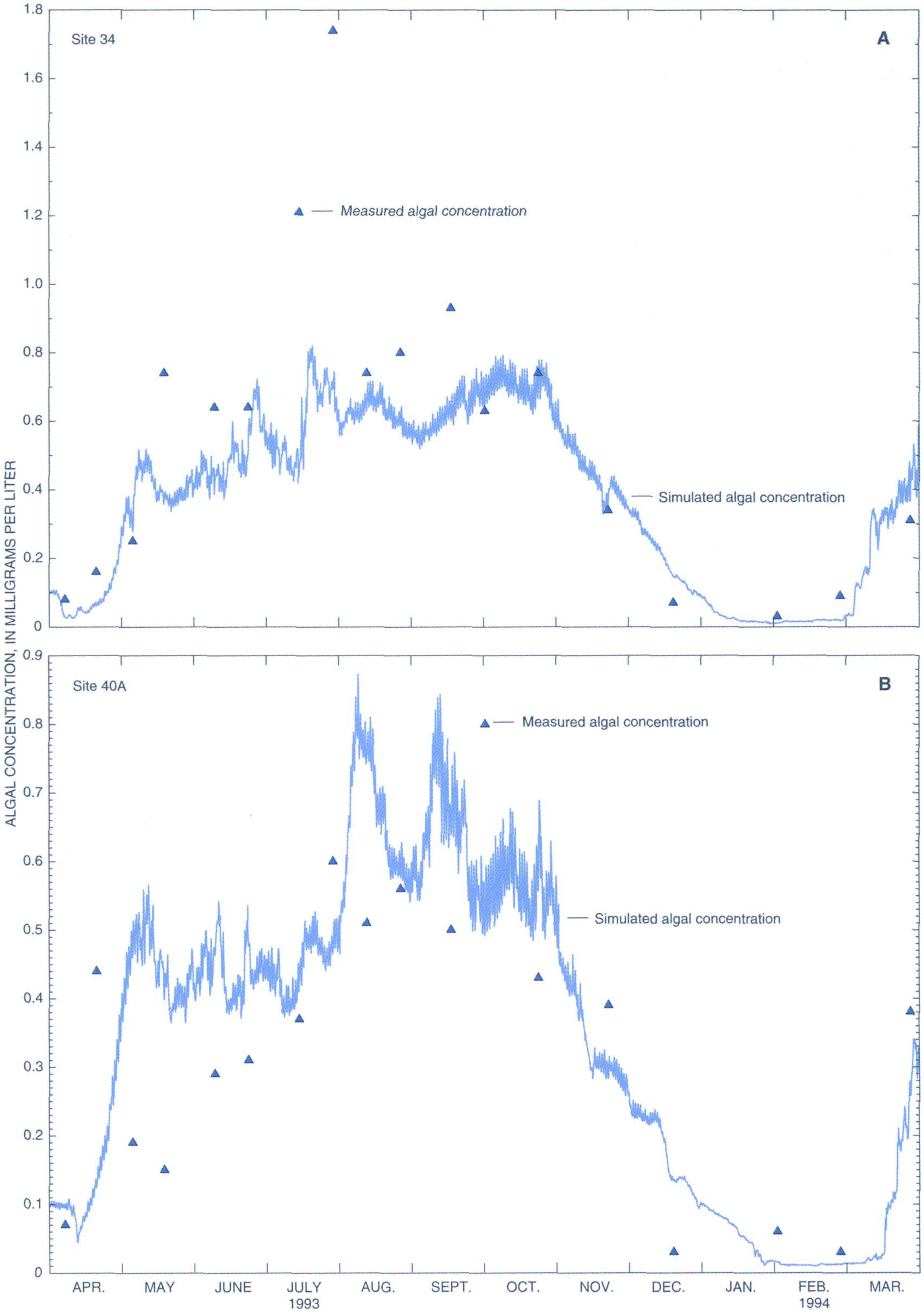


Figure 31. Measured and simulated algal concentrations at Lake Hickory sites (A) 34 and (B) 40A, April 1993 through March 1994.

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 Lake Hickory. Other processes that may be important to the accurate simulation of algal concentrations, such as self shading and consumption of algae by zooplankton and fish, 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 Lake Hickory, therefore, depend on accurate simulations of DO. The release rate of PO_4 from bottom sediments used in the Lake Hickory model during anoxic conditions is considerably lower than values suggested by Cole and Buchak (1995). Even with the low release rate, PO_4 concentrations were generally overpredicted for July through October 1993. 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.

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 Lake Hickory model was not conducted; however, about 25 water-quality simulations were made after the hydrodynamic component of the model was calibrated. In addition, more than 50 water-quality simulations were performed previously during the development of the Rhodhiss Lake model (Giorgino and Bales, 1997), and the models for the two reservoirs behaved similarly. 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 Lake Hickory model computational grid. Sub-grid scale processes represented by the hydraulic parameters were relatively unimportant, so model results did not change significantly with changes in these parameters.

The vertical distribution of temperature in Lake Hickory (fig. 23) was simulated much more accurately than in the Rhodhiss Lake model. Vertical distributions of DO concentration were less accurately simulated than vertical distributions of temperature but were simulated more accurately by the Lake Hickory model than the Rhodhiss Lake model. Two factors which 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 temperatures in the upper layers of the reservoir were somewhat affected by the coefficients governing light extinction and absorption of solar radiation. These upper layer temperatures, in turn, affected the density distribution and, thus, vertical mixing. Model results were relatively insensitive to the parameters controlling heat exchange at the reservoir bottom, partly because the total area of the bottom-layer computational grids was small relative to the entire surface of the sediment-water interface.

Nearly 60 model parameters that affect chemical kinetics were specified as input to the Lake Hickory model (table 10). Initial Lake Hickory model simulations were made by using parameter values from the calibrated Rhodhiss Lake model, which then were adjusted within ranges suggested by Cole and Buchak (1995). Parameter values for both models are presented in table 10 for comparison.

Algal concentrations in Lake Hickory 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. Algal concentrations also were controlled by the amount of PO_4 available in the upper layers of the reservoir.

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. 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. 27, July 27, 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. Although a number of parameters control production of organic matter, including 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 Lake Hickory model as a first-order process (decay of organic matter in the bottom layer) and varied longitudinally along the reservoir.

According to model simulations, algal growth in the upper layers of the reservoir was limited by PO_4 and, to a lesser extent, light availability. Consequently, PO_4 concentrations in the upper layers were generally quite low and most sensitive to the algal growth rate. PO_4 is released from the bottom sediments when the DO concentration in the bottom layer is less than 0.1 mg/L. The PO_4 concentrations in the mid- to lower layers of the reservoir and algal concentrations in the upper layers were sensitive to changes in the PO_4 release rate from the sediments. Changes in other parameters controlling phosphorus kinetics resulted in only small changes in simulated PO_4 and algal concentrations.

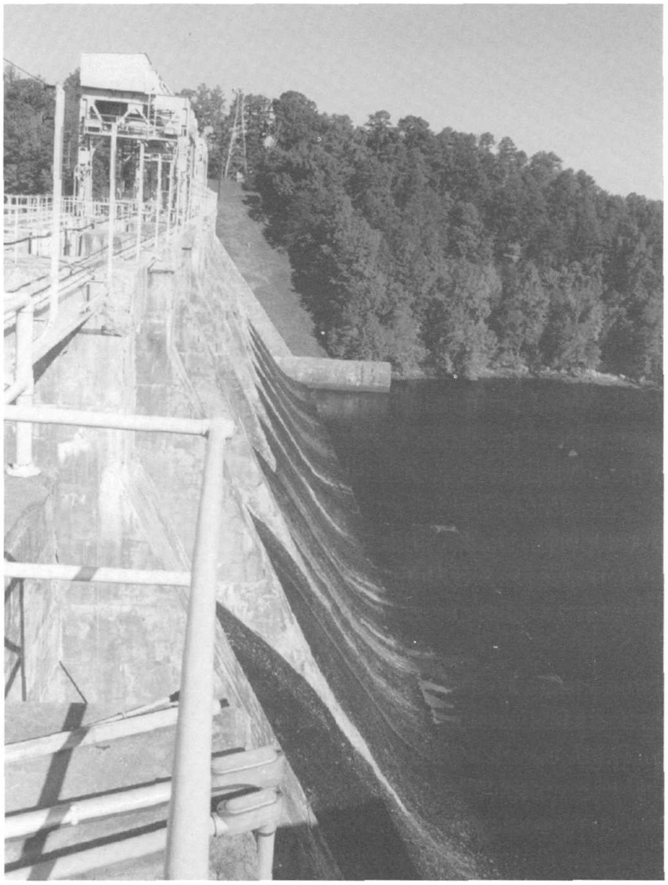
Model Application

The calibrated Lake Hickory model was used to simulate the movement of a neutrally buoyant conservative material through the reservoir. Simulations also were made to evaluate the effects on DO concentrations and algae as a result of changes in PO_4 loads from Catawba River inflows (nonpoint sources) and from bottom sediments.

Simulation of Conservative Material Transport

Transport of a conservative, neutrally buoyant material, or tracer, released at the U.S. Highway 321 bridge was simulated for summer and winter conditions. The material was released on July 31 through August 1 (summer) and November 15 through 16 (winter). Inflows during both the summer and the winter releases were affected by Rhodhiss Dam hydropower peaking operations and ranged from 2 to 140 m^3/s during the summer period and from 2 to 100 m^3/s during the winter. Likewise, outflows at Oxford Dam were quite variable because of hydropower peaking operations. 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.

Inflow water temperature during the summer release was colder than surface waters in the reservoir. Consequently, the material released at the U.S. Highway 321 bridge sank fairly rapidly as the material moved into the reservoir (fig. 32). One day after the release and about 2 km downstream from the release location, the highest concentration of material was at about 7 m below the water surface. Two days after the release, the highest concentrations were at least 10 m below the water surface (fig. 32).



RHODHISS DAM AT THE UPSTREAM END OF LAKE HICKORY.

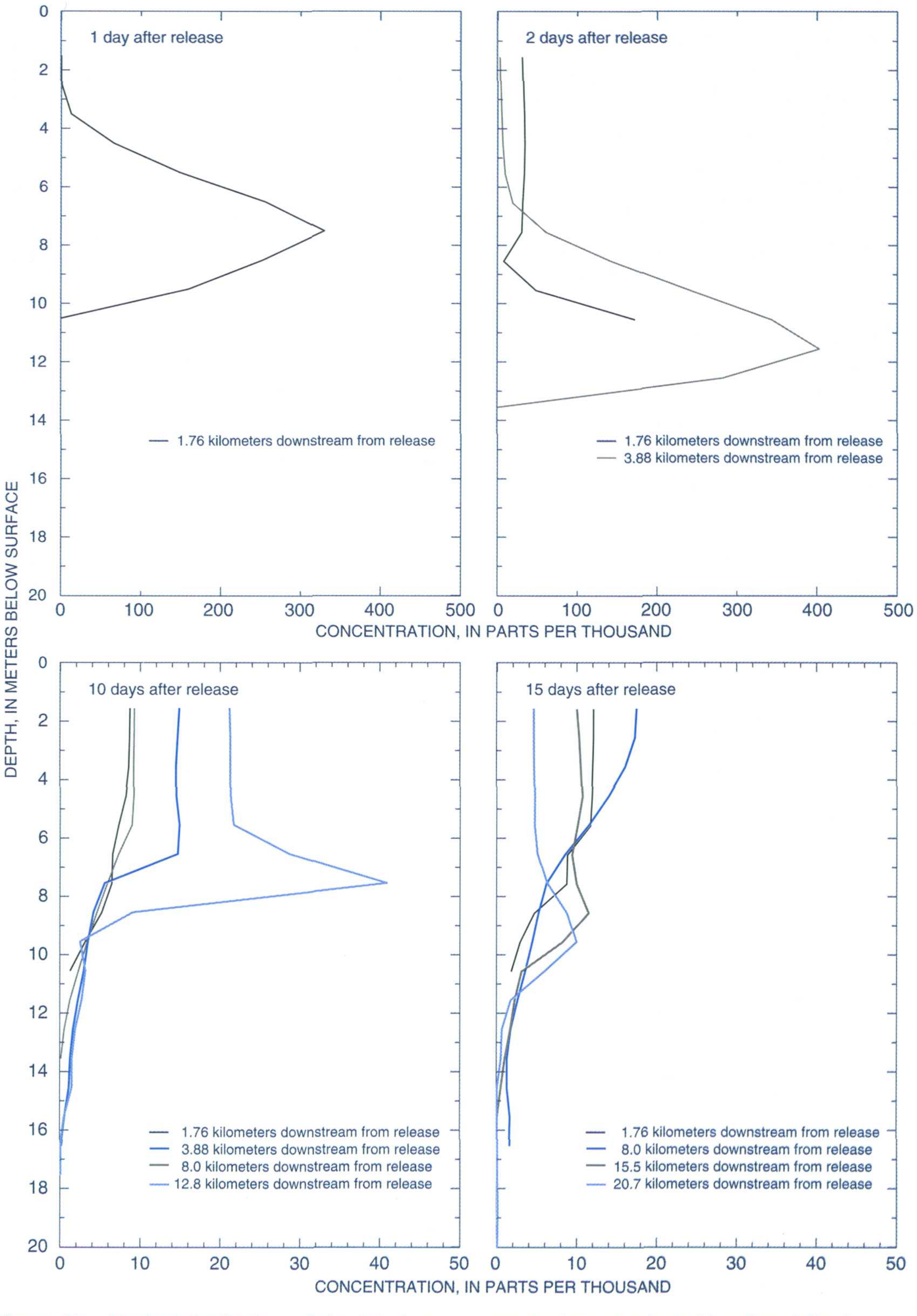


Figure 32. Vertical distributions of simulated tracer concentrations at selected locations following a summer release of a neutrally buoyant tracer.

The maximum concentration of the tracer at the water surface occurred between 6 and 7 days after the summer release at a location 12.8 km downstream from the release (fig. 33), indicating that the travel time for the peak concentration was on the order of 2 kilometers per day (km/d). The maximum surface concentration in the downstream reaches of the reservoir was between 1 and 3 percent of the initial concentration, and the maximum near-bottom concentration was less than 1 percent of the initial concentration. However, the maximum concentration at a particular location likely occurred somewhere near mid-depth (fig. 32), rather than at the surface or bottom of the reservoir. Concentrations remained at about 0.05 percent of the initial concentration for at least 60 days after the release.

Concentrations 2 days following the winter release were greater than corresponding concentrations following the summer release (figs. 32 and 34). The tracer was more vertically mixed 15 days after the winter release than after the summer release.

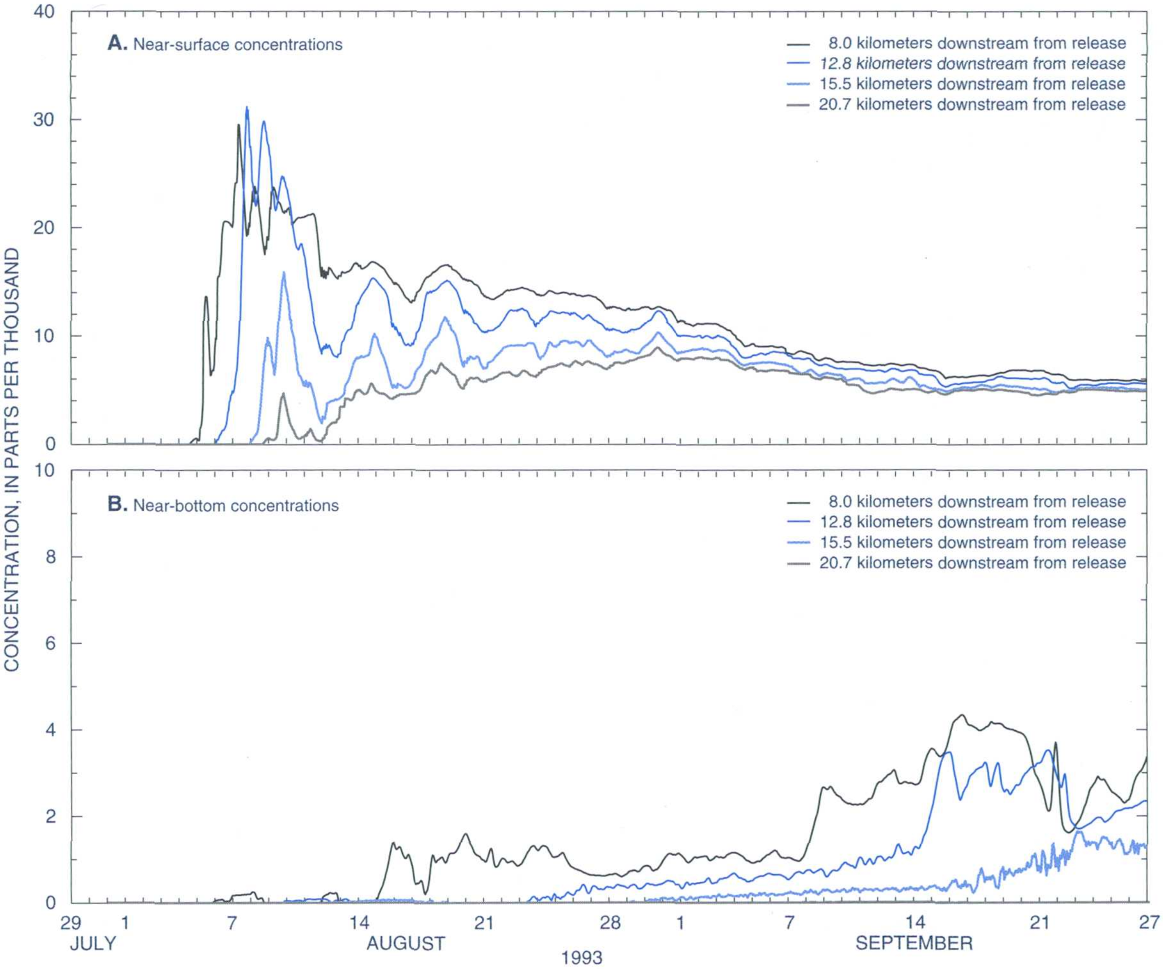


Figure 33. Temporal distributions of simulated tracer concentrations (A) near the water surface and (B) near the reservoir bottom at selected locations following a summer release of a neutrally buoyant tracer.

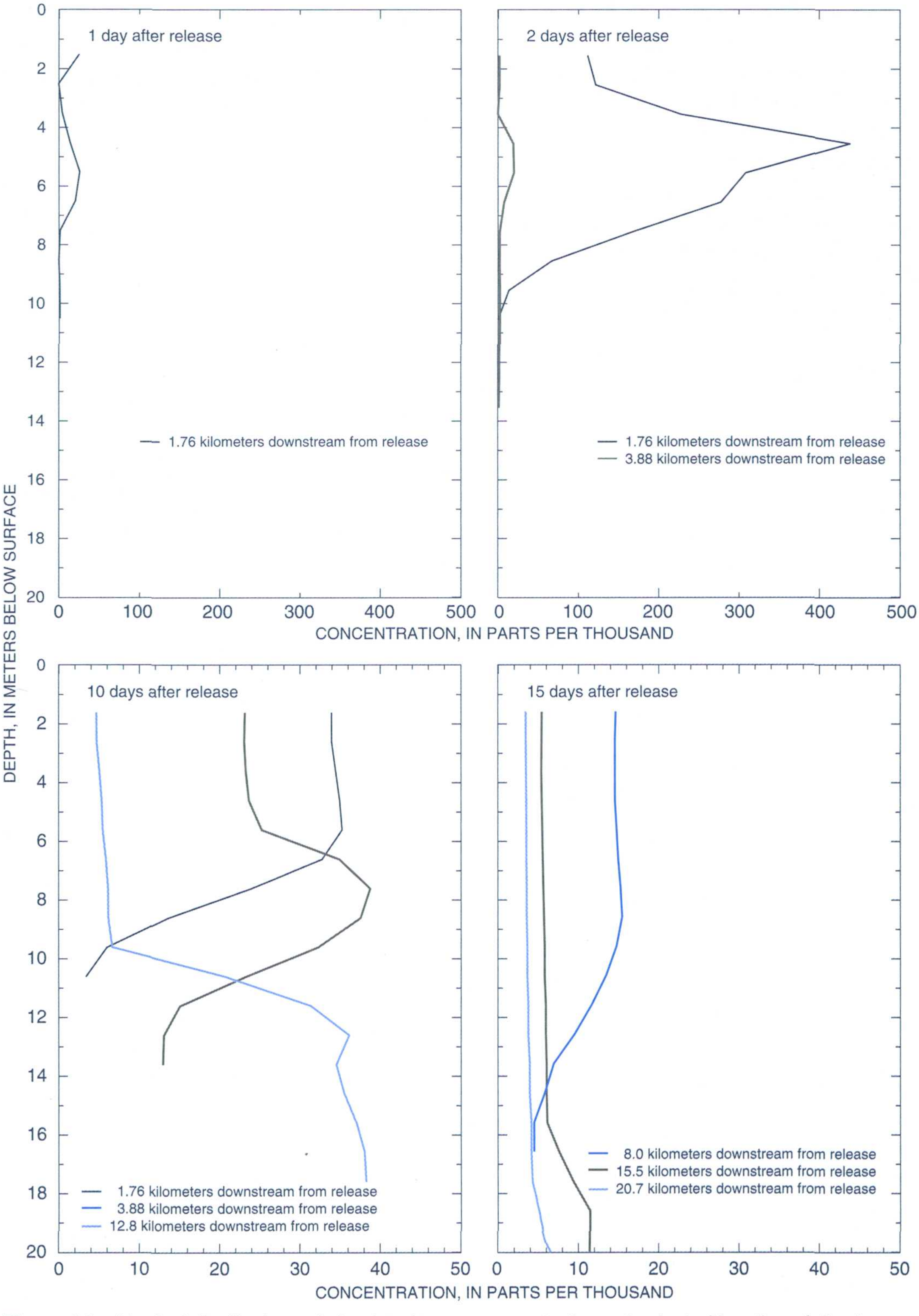


Figure 34. Vertical distributions of simulated tracer concentrations at selected locations following a winter release of a neutrally buoyant tracer.

The maximum tracer concentration near the water surface reached Oxford Dam more than 20 days after the winter release of the tracer. However, about 12 days were required for the maximum tracer concentration near the bottom to reach Oxford Dam (fig. 35). The pulsating nature of the concentration distributions at the surface for the summer release and near the bottom for the winter release likely result from the effects of hydropower peaking operations on flow and transport in the reservoir.

These examples of the transport of a conservative material demonstrate (1) the utility of the Lake Hickory model in evaluating the movement of a short-term or a long-term release into the reservoir, (2) the general 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 material through Lake Hickory.

Simulation of Effects of Changes in Nutrient Loads

Concentrations of NO_2+NO_3, NH_4, and PO_4 at the U.S. Highway 321 bridge were lowered by 30 percent to simulate the possible effects of future nutrient management actions on algal and DO concentrations. Reductions in the inflow concentrations resulted in a fairly small change in algal

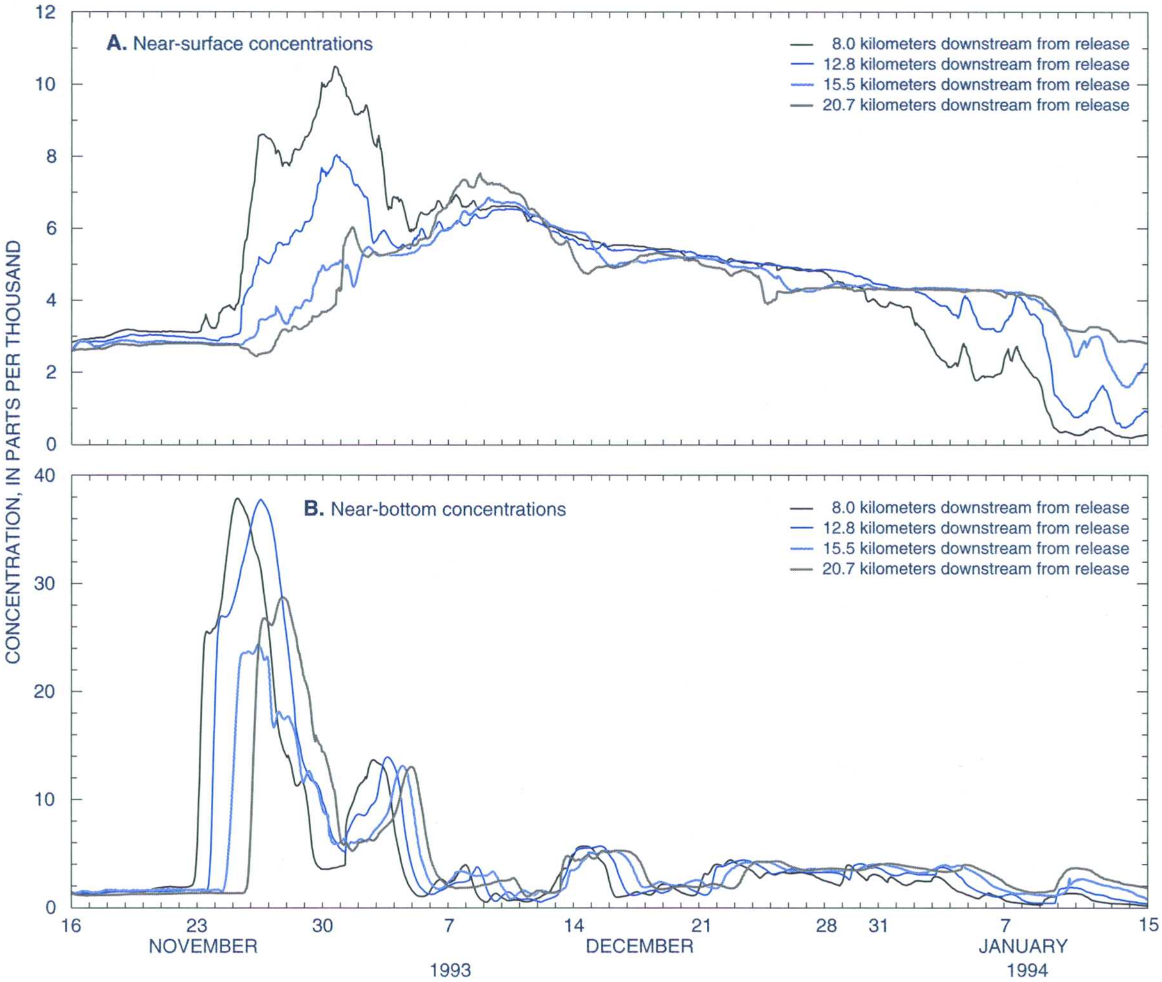


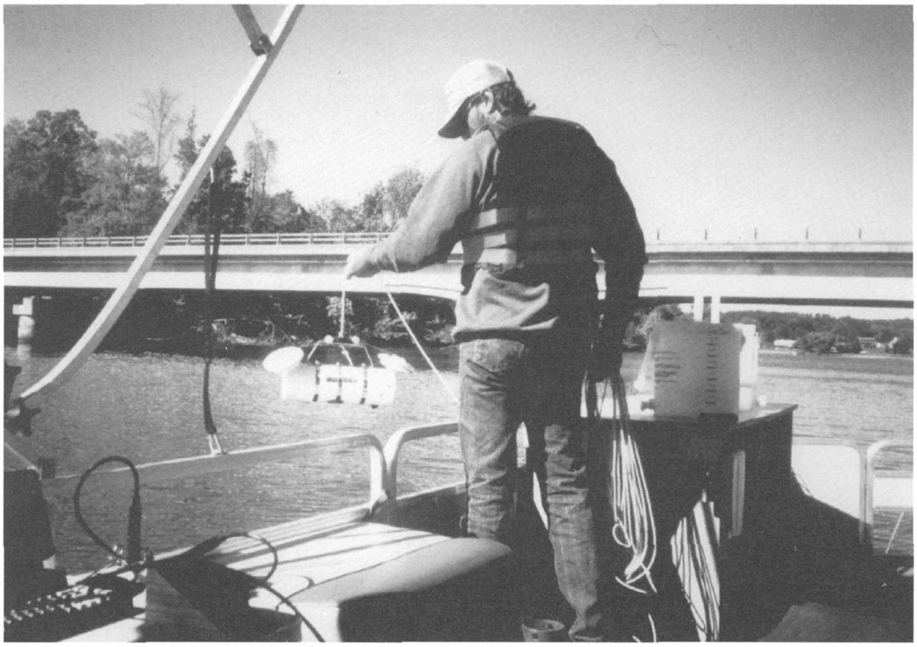
Figure 35. Temporal distributions of simulated tracer concentrations (A) near the water surface and (B) near the reservoir bottom at selected locations following a winter release of a neutrally buoyant tracer.

concentrations, with the greatest effect being a delay of about 2 weeks of initial growth of algae in late April and early May. Otherwise, simulated algal and DO concentrations did not seem to be particularly sensitive to a 30-percent reduction in nutrients in the inflow to Lake Hickory at the U.S. Highway 321 bridge.

The simulated release rates of PO_4 and NH_4 from bottom sediments, which are calculated as a fraction of the sediment oxygen demand, were lowered—the PO_4 release rate was lowered to zero, and the NH_4 rate was lowered an order of magnitude to 0.004 (table 10). The reduction in the nutrient release rates resulted in only a fairly small reduction in simulated algal concentrations and very little change in DO concentrations. This result is similar to the result obtained for Rhodhiss Lake (Giorgino and Bales, 1997).

In order to simulate the effects of shoreline development on Lake Hickory water quality, constituent loads in Snow Creek were increased. Snow Creek was used as the test case because the creek is near the midpoint of the reservoir and fairly near the city of Hickory. Concentrations of NO_2+NO_3, NH_4, and PO_4 in Snow Creek were set at 0.6 mg/L, 0.3 mg/L, and 0.15 mg/L, respectively. These values are based on median concentrations of these constituents measured in urban streams in Charlotte, N.C. (Robinson, Hazell, and Garrett, 1998). For the base (undeveloped, or 1993–94) simulation, NO_2+NO_3 concentrations in Snow Creek ranged from 0.053 to 0.320 mg/L; NH_4 concentrations ranged from 0.002 to 0.048 mg/L; and PO_4 concentrations ranged from 0.001 to 0.005 mg/L. Flows in Snow Creek ranged from 0.3 to 4.47 m^3/s, with a median flow of 0.61 m^3/s during the simulation period.

Algal concentrations in the Lake Hickory simulations were affected more by the changed loads from Snow Creek than by reductions in loads from bottom sediments or at the U.S. Highway 321 bridge (fig. 36). Algal concentrations at site 34, upstream from the mouth of Snow Creek, increased about 100 percent relative to base conditions in May 1993 when streamflow was high (fig. 36A). Less pronounced increases in algal concentrations at site 34 also were predicted for June and mid-August through mid-September 1993 (fig. 36A). Likewise, simulated peak algal concentrations at site 40A increased about 100 percent relative to base conditions during May 1993 and also increased in September 1993 (fig. 36B). From these results, it appears that Lake Hickory water-quality conditions may be sensitive to increases in loads of nutrients delivered by streams directly into the middle reaches of the reservoir.



WATER-QUALITY SAMPLING ON LAKE HICKORY NEAR THE U.S. HIGHWAY 321 BRIDGE.

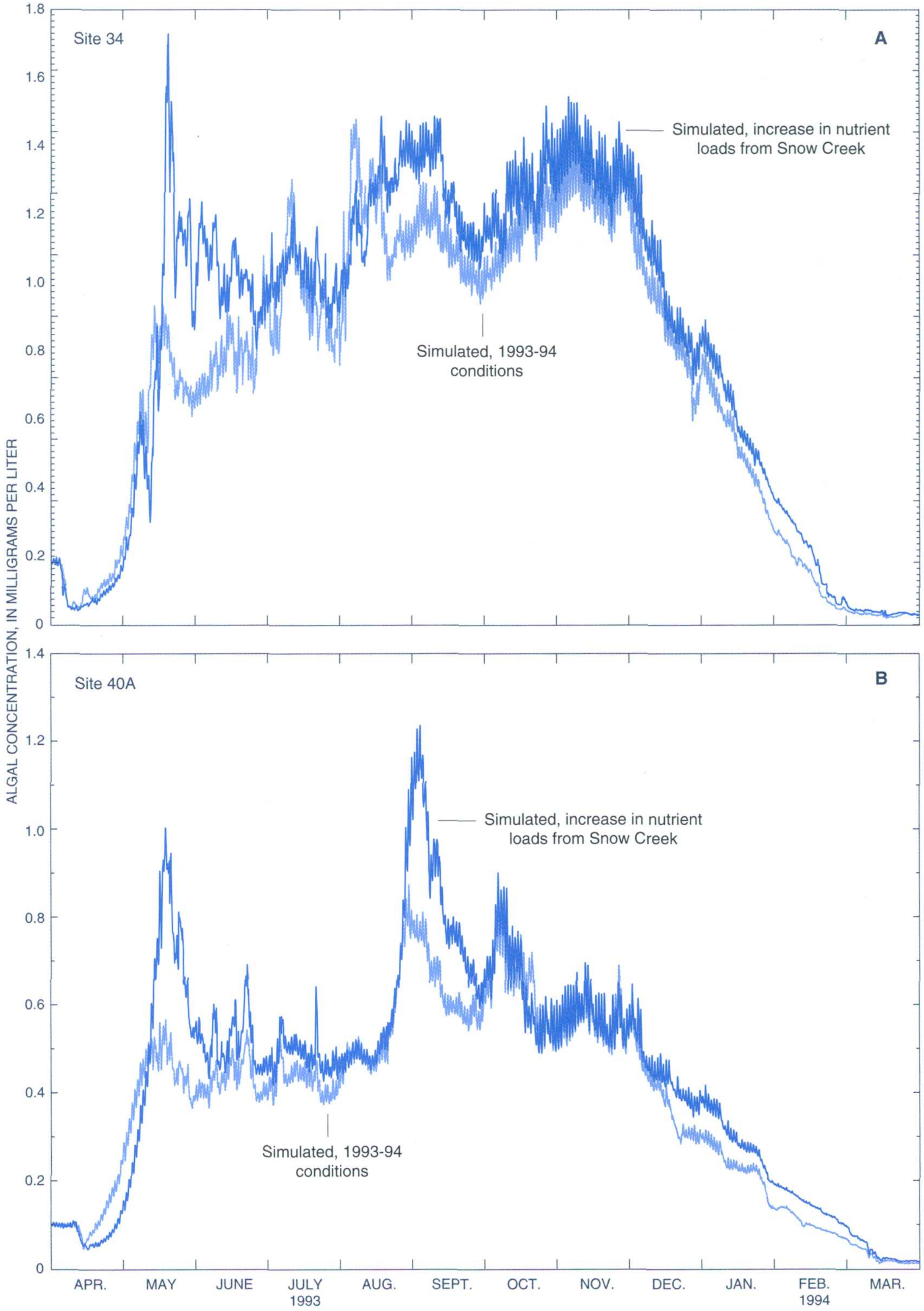


Figure 36. Simulated algal concentrations at Lake Hickory sites (A) 34 and (B) 40A for 1993–94 conditions and for increased nutrient loads from Snow Creek.

SUMMARY

Hydrologic and water-quality data were collected from Lake Hickory and two of its tributaries to describe ambient conditions from January 1993 through March 1994 and to calibrate a water-quality model for the reservoir. Data also were used to estimate loads of solids and nutrients from selected point sources and tributaries to Lake Hickory.

Results from the investigation of Lake Hickory were compared to results from a similar study of Rhodhiss Lake, a mainstem reservoir on the Catawba River immediately upstream from Lake Hickory. Water-quality characteristics in the two reservoirs were closely linked, and models developed for the two reservoirs behaved similarly. However, Rhodhiss Lake was strongly affected by watershed dynamics, whereas water-quality conditions in Lake Hickory were strongly influenced by inputs from Rhodhiss Lake.

Distributions of temperature and dissolved oxygen (DO) revealed complex patterns of circulation in Lake Hickory which, in turn, affected the distribution of other water-quality constituents. Circulation patterns varied seasonally as the reservoir underwent thermal stratification and varied spatially from the headwaters to Oxford Dam. The part of Lake Hickory upstream from the U.S. Highway 321 bridge was unstratified during much of the study period. Daily mean temperatures near the surface and bottom of the water column at this location generally agreed within 1 °C. Downstream from the U.S. Highway 321 bridge to Oxford Dam, Lake Hickory thermally stratified from May through mid-September 1993. During stratification, cool, sub-surface releases from Rhodhiss Dam plunged beneath the warmer surface waters of Lake Hickory and moved through the reservoir as interflow.

Distributions of DO in Lake Hickory were strongly influenced by thermal stratification and mixing. During the summer, stratification curtailed vertical mixing downstream from the U.S. Highway 321 bridge. Dissolved oxygen was rapidly depleted from Lake Hickory bottom waters beginning in May, and the hypolimnion of Lake Hickory was anoxic from July through mid-September 1993. Summertime anoxia was associated with increased specific conductance, ammonia, and orthophosphate, and decreased nitrite plus nitrate in the hypolimnion. During the same period, concentrations of nitrite plus nitrate, ammonia, and orthophosphate were low in the epilimnion, presumably from algal assimilation.

Constituent concentrations in the mainstem of Lake Hickory were strongly influenced by upstream releases from Rhodhiss Dam, which represented the dominant inflow to Lake Hickory. In particular, specific conductance at the U.S. Highway 321 bridge closely followed seasonal variations observed in Rhodhiss Lake upstream from Rhodhiss Dam. In general, water clarity increased, and concentrations of suspended solids, phosphorus, and summertime chlorophyll *a* decreased in a downstream direction from the U.S. Highway 321 bridge to Oxford Dam. Two chlorophyll *a* samples collected near the U.S. Highway 321 bridge exceeded the North Carolina water-quality standard of 40 μg/L during the investigation.

Concentrations of fecal coliform bacteria exceeded 200 cols./100 mL in one of 60 samples collected from Lake Hickory. In contrast, concentrations of fecal coliform bacteria exceeded 200 cols./100 mL in 40 percent of samples collected from Upper Little River, and in 60 percent of samples collected from Middle Little River, two tributaries to Lake Hickory. Concentrations exceeding this criterion were observed in 16 to 37 percent of samples collected from sites in Rhodhiss Lake.

Load estimates for the period April 1993 through March 1994 indicated that releases from Rhodhiss Dam accounted for most of the suspended solids, nitrogen, and phosphorus entering the headwaters of Lake Hickory. Loads of nitrogen and phosphorus from selected point-source discharges were potentially important, but loads of suspended solids from these point sources were relatively insignificant. Water-quality data and load estimates suggested that Rhodhiss Lake retained a substantial amount of the inorganic solids transported by the Catawba River. Concentrations of total suspended solids and nonvolatile suspended solids were lower in the headwaters of Lake Hickory than in the headwaters of Rhodhiss Lake. Furthermore, loads of total suspended solids into the headwaters of Lake Hickory were 29 percent lower than total suspended solids loads into the headwaters of Rhodhiss Lake during April 1993 through March 1994.

The hydrodynamic and water-quality computer model CE-QUAL-W2 was used to simulate flow, transport, and water-quality conditions in Lake Hickory for 1993–94 conditions and for selected hypothetical conditions. The model domain extended from the U.S. Highway 321 bridge to Oxford Dam—a

distance of 22 km—and included five embayments or coves.

There was good agreement between measured and simulated water levels at Oxford Dam. The root mean square difference between measured and simulated water levels was 0.046 m, and the actual differences ranged from 0.18 m to -0.20 m. Ninety percent of the differences between measured and simulated water levels were between 0.06 and -0.08 m. The total range in measured water level during the simulation period was 1.14 m.

At the mid-reservoir measurement site, water temperature during April 1993 through March 1994 ranged from about 4 to about 31 °C. The mean difference between the measured and corresponding simulated water temperatures at the mid-reservoir site was -0.05 °C; 85 percent of the differences were within 1 °C of measured values. All of the results of the water temperature simulations suggest that (1) the calibrated model provides a reasonable simulation of water temperature in Lake Hickory, with most of the simulated values within 1 °C of the actual value; (2) water temperatures were overpredicted for the fall of 1993; and (3) Lake Hickory water temperature simulations were more accurate than those for Rhodhiss Lake.

Eleven water-chemistry constituents were included in the Lake Hickory model. Analysis of results focused on DO and algae because ambient water-quality standards exist for DO and algae. For much of the year, simulated vertical profiles of DO concentrations were within 1 to 1.5 mg/L of measurements, but agreement was better at upstream sites than near the dam. Agreement also was better during the spring and summer than during September and October. 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. There was a greater tendency toward oversimulation than undersimulation of DO values at the mid-reservoir site. The most biased results and most of the large differences between measured and simulated values occurred in the fall of 1993.

The calibrated model provided a reasonable simulation of DO concentration in Lake Hickory. The onset and end of low DO conditions were simulated very accurately throughout the reservoir. However, DO concentrations were oversimulated in the fall when low DO waters mixed with surface waters, indicating perhaps an underestimation of the hypolimnetic volume of the reservoir.

The cumulative frequency of occurrence of measured DO concentrations at the mid-reservoir site was compared with the corresponding distribution for measured DO values. The frequency of occurrence of DO concentrations less than 4 mg/L was almost the same for measured and simulated DO. For DO concentrations between about 4 and 12 mg/L, simulated DO values were greater than measured DO values. However, the model should still be useful to water-quality managers, who generally have more interest in the occurrence of DO concentrations less than 5 mg/L than in higher values. Dissolved oxygen simulations for Lake Hickory were substantially better than those for Rhodhiss Lake, particularly with regard to the overall magnitude of the errors and to the accurate simulation of vertical distribution of DO concentrations.

The calibrated Lake Hickory model was applied using 1993–94 boundary data to simulate the movement of a neutrally buoyant conservative material through the reservoir. Results from these simulations demonstrate (1) the utility of the Lake Hickory 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 on transport processes.

Simulations were made of the effects on DO and algal concentrations as a result of changes in orthophosphate loads from inflows and from bottom sediments. A 30-percent reduction in inflow concentrations at the U.S. Highway 321 bridge resulted in only a small change in DO and algal concentrations. Likewise, a change in the release rates of orthophosphate and ammonia from bottom sediments had very little effect on simulated DO and algal concentrations. Concentrations of nitrite plus nitrate, ammonia, and orthophosphate in Snow Creek were increased to levels similar to those in streams draining developed areas in Charlotte, N.C. Maximum simulated algal concentrations increased about 100 percent relative to base conditions as a result of these increased nutrient loads.

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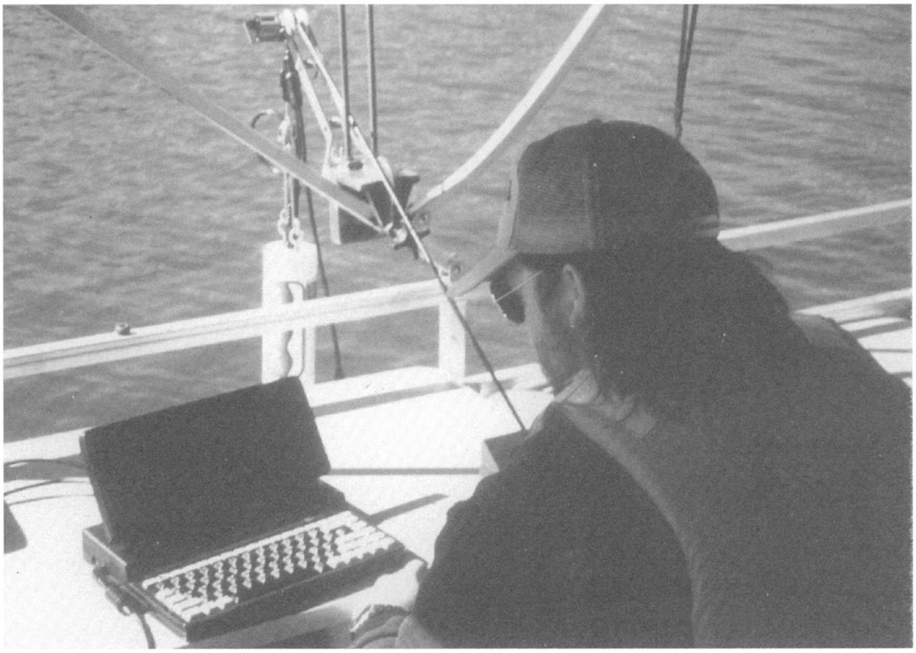

LOGGING WATER-QUALITY DATA ON LAKE HICKORY.

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