

**WATER QUALITY AND EVALUATION OF RAW-WATER-ROUTING
SCENARIOS, CHICKAHOMINY, DIASCUND CREEK, AND LITTLE
CREEK RESERVOIRS, SOUTHEASTERN VIRGINIA, 1983-86**

By Dennis D. Lynch

U.S. GEOLOGICAL SURVEY

Water-Resources Investigations Report 92-4034



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CITY OF NEWPORT NEWS, VIRGINIA

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MANUEL LUJAN, Jr., Secretary

U.S. GEOLOGICAL SURVEY

Dallas L. Peck, Director

For additional information write to:

District Chief
U.S. Geological Survey, WRD
600 West Broad Street
Room 606
Richmond, Virginia 23230

Copies of this report can be purchased from:

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

<i>Multiply</i>	<i>By</i>	<i>To obtain</i>
<i>Length</i>		
inch (in.)	2.54	centimeter
foot (ft)	0.3048	meter
mile (mi)	1.609	kilometer
<i>Area</i>		
square mile (mi ²)	259	hectare
square mile (mi ²)	2.590	square kilometer
square foot (ft ²)	0.09290	square meter
<i>Volume</i>		
million gallons (Mgal)	3,785	cubic meters
billion gallons (Bgal)	3,785,000	cubic meters
<i>Flow</i>		
cubic foot per second (ft ³ /s)	0.02832	cubic meter per second
inch per year (in/yr)	25.4	millimeter per year
million gallons per day (Mgal/d)	0.003785	million cubic meters per day
million gallons per yr (Mgal/yr)	0.003785	million cubic meters per year
mile per hour (mi/h)	1.609	kilometer per hour
<i>Mass</i>		
pound (lb)	453.6	gram
pound (lb)	0.4536	kilogram
gram (g)	0.03527	ounce, avoirdupois
kilogram (kg)	2.205	pound, avoirdupois
ton	0.9072	megagram

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

In this report, chemical concentrations in water are expressed in milligrams per liter (mg/L) or micrograms per liter (µg/L); 1,000 µg/L = 1 mg/L. Specific conductance of water is expressed in microsiemens per centimeter at 25 degrees Celsius (µS/cm). This unit is equivalent to micromhos per centimeter at 25 degrees Celsius. Temperature in degrees (°C) can be converted to degrees Fahrenheit (°F) by use of the following equation:

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

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Water Quality and Evaluation of Raw-Water-Routing Scenarios, Chickahominy, Diascund Creek, and Little Creek Reservoirs, Southeastern Virginia, 1983-86

By Dennis D. Lynch

ABSTRACT

The Department of Public Utilities, city of Newport News, provides water to more than 330,000 people and many industries on the York-James Peninsula in southeastern Virginia. Raw water for this supply is withdrawn primarily from Chickahominy, Diascund Creek, and Little Creek Reservoirs. Withdrawals from the reservoirs amounted to 33.7 Mgal/d (million gallons per day) during 1983-86. The purpose of this report is to characterize the quantity and quality of water in these reservoirs and to describe how quality differs temporally and spatially. The report focuses on water-quality problems commonly encountered by water suppliers in the Coastal Plain physiographic province, such as high concentrations of iron, manganese, nutrients, and algae.

Algal growth in all three reservoirs is phosphorus-limited, as evidenced by nitrogen-to-phosphorus ratios that consistently exceeded 7.0 (by weight). On the basis of mean total-phosphorus concentrations of 0.049 mg/L (milligram per liter) for Chickahominy Reservoir and 0.029 mg/L for Diascund Creek Reservoir, these reservoirs are classified in the Carlson (1977) trophic state index as eutrophic and mesotrophic/eutrophic, respectively. During draw-down, however, Diascund Creek Reservoir is classified as hypereutrophic, on the basis of a mean total-phosphorus concentration of 0.09 mg/L. The trophic level of Little Creek Reservoir depends primarily on the quantity and quality of inputs from Chickahominy and Diascund Creek Reservoirs. Without water from the other reservoirs, Little Creek Reservoir would be classified as oligotrophic. This classification is based on an estimated total-phosphorus concentration of 0.005 mg/L. Because

inflows averaged 11.2 Mgal/d during 1983-86, the phosphorus concentration in Little Creek Reservoir averaged 0.014 mg/L, which classified it as mesotrophic. During 1983-86, chlorophyll-*a* concentrations averaged 21, 14, and 8 micrograms per liter in Chickahominy (base-flow periods only), Diascund Creek, and Little Creek Reservoirs, respectively. These values are approximately proportional to the mean total-phosphorus concentrations in the three reservoirs.

Anaerobic bottom sediments of the three reservoirs are a primary source of iron and manganese to the hypolimnions. By late summer, dissolved-iron and manganese concentrations near the sediment-water interface typically exceed 10 and 1 mg/L, respectively. Elevated hypolimnial concentrations do not cause severe water-supply problems because these constituents are quickly lost from the water column during fall overturn. Contrary to findings of similar investigations, anaerobic bottom sediments are not a net source of phosphorus to the hypolimnions of Chickahominy, Diascund Creek, and Little Creek Reservoirs. The lack of phosphorus release from bottom sediments can result from sorption reactions involving clay or iron complexes, incorporation of phosphorus in organic matter, and (or) formation of an iron-phosphate mineral, such as vivianite.

On the basis of a detailed mass-balance analysis, Little Creek Reservoir is a significant source of uptake (sink) for several constituents. Raw water routed through Little Creek Reservoir lost 20 and 70 percent of its nitrogen and phosphorus loads, respectively, during 1983-86. Loads of dissolved sil-

ica, iron, and manganese were reduced 60, 55, and 35 percent, respectively. Raw-water quality is substantially improved with respect to standards established by the U.S. Environmental Protection Agency when withdrawals from Chickahominy and Diascund Creek Reservoirs are routed through Little Creek Reservoir. Results from an empirical phosphorus model indicate that the load of phosphorus delivered to terminal reservoirs could be reduced about 50 percent if all raw water were routed through Little Creek Reservoir.

INTRODUCTION

The Department of Public Utilities, city of Newport News, provides water to more than 330,000 people and many industries on the York-James Peninsula in southeastern Virginia (figs. 1 and 2). The city's raw-water supply is withdrawn primarily from Chickahominy, Diascund Creek, and Little Creek Reservoirs (fig. 3), which are located about 25 mi west of the city of Newport News. From 1983 through 1986 these reservoirs supplied 77 percent of the average 44 Mgal/d of water used by Newport News; the remaining 23 percent came from smaller reservoirs located closer to the city. Chickahominy, Diascund Creek, and Little Creek Reservoirs supplied an average of 17.5, 12.6, and 3.6 Mgal/d of the 44 Mgal/d of raw water, respectively; however, depending on operational strategies, daily pumpage from these reservoirs differed considerably from these average figures. Water from the Chickahominy, Diascund Creek, and Little Creek Reservoirs is pumped through a 25-mi pipeline to two terminal reservoirs, where it is stored until treatment. A schematic of the routing of the Newport News water supply is shown in figure 4.

Little Creek Reservoir is important for storage of raw water pumped from the Chickahominy and Diascund Creek Reservoirs. About 69 percent of the water pumped into Little Creek Reservoir from 1983 through 1986 came from Chickahominy Reservoir, whereas the remainder came from Diascund Creek Reservoir. About 25 percent of the water in Little Creek Reservoir came from its small drainage basin during the study.

Chickahominy Reservoir was created by the construction of Walkers Dam in 1942 to supply drinking water for military personnel. The reservoir was later purchased by the city of Newport News to expand its raw-water supply. Newport News constructed Diascund Creek Reservoir in the early

1960's to again expand its raw-water supply. Little Creek Reservoir was constructed in the late 1970's to provide off-line storage of raw water to supplement water from Chickahominy and Diascund Creek Reservoirs during dry periods and times of peak demand. Little Creek Reservoir also serves as a backup supply when water from other sources cannot be used because of poor water quality or mechanical problems.

In order for the Department of Public Utilities to optimize their operations and plan effectively for the future, the Department needs information about each component of their water-supply system. To obtain information on the hydrology of the reservoir system, the U.S. Geological Survey, in cooperation with the Newport News Department of Public Utilities, began a study to characterize the water quality of Chickahominy, Diascund Creek, and Little Creek Reservoirs and to provide the beginnings of a long-term data base for future studies.

Water-quality sampling of Chickahominy, Diascund Creek, and Little Creek Reservoirs began in August 1983 and continued through October 1986. During the first year, sampling was intensive (15 onsite visits to 8 sampling sites) so that the temporal and spatial variability of water quality could be described in detail. The early emphasis of the program was on characterizing the water quality of Little Creek Reservoir because it was the least understood reservoir in the Newport News water-supply system. By the final year of data collection, sampling frequency was reduced to 10 times per year, and the number of sampling sites was reduced to 5. The program emphasis in the final year shifted toward developing a balanced, cost-effective, long-term data base.

Water samples were analyzed for nutrients and chlorophyll-*a* because algal productivity and biomass are linked to many supply problems, including filter clogging, objectionable tastes and odors, and **elevated concentrations**¹ of trihalomethanes in finished water. Samples were also analyzed for iron and manganese because elevated concentrations were observed in all three reservoirs. Elevated concentrations of iron and manganese clog treatment filters, impart a bad taste to drinking water, and

1. In this report, the term "elevated concentrations" refers to a concentration that exceeds the drinking-water regulation established by the U.S. Environmental Protection Agency (U.S. Environmental Protection Agency, 1986a, b).

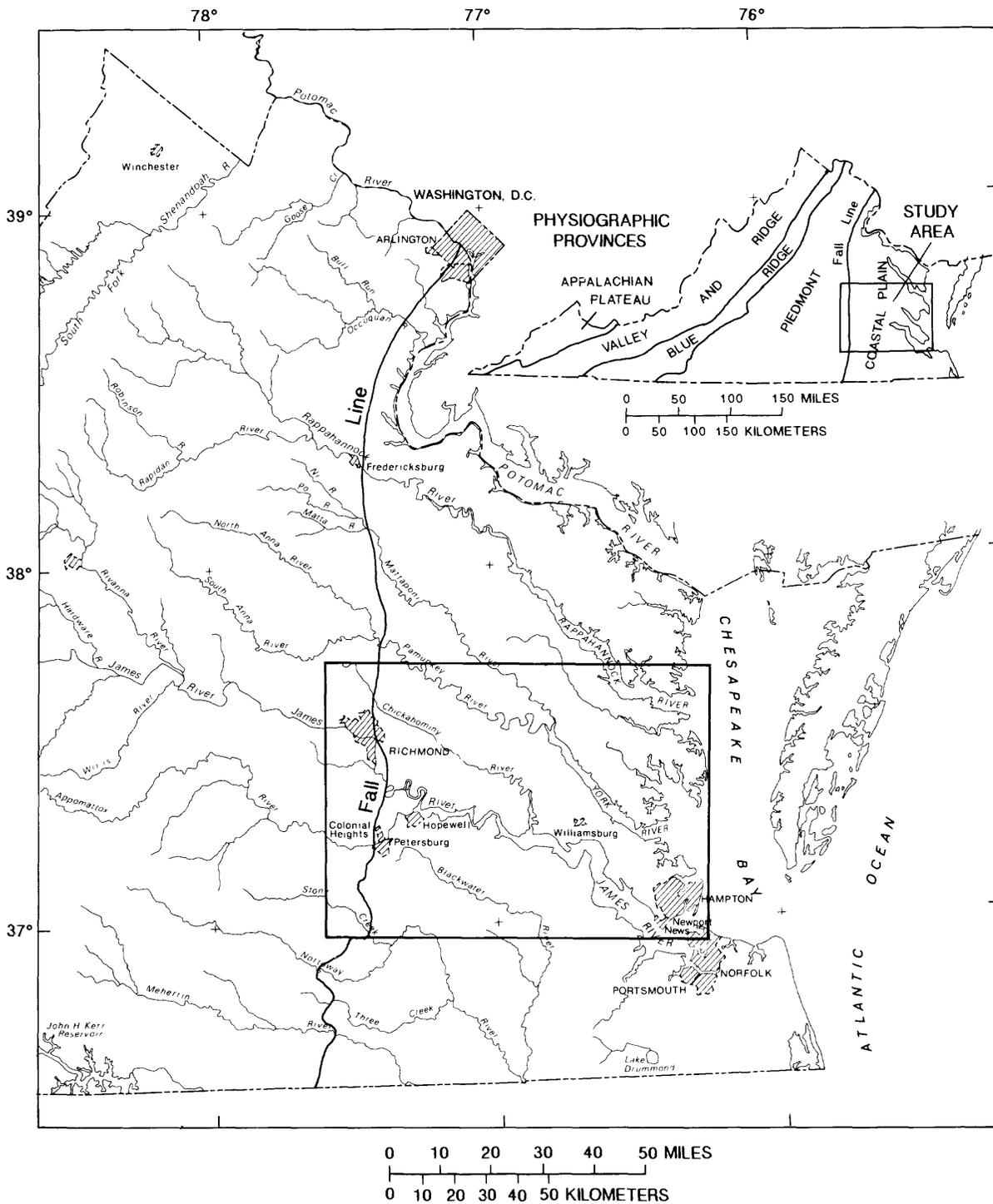


Figure 1.--Physiographic provinces of Virginia and study area.

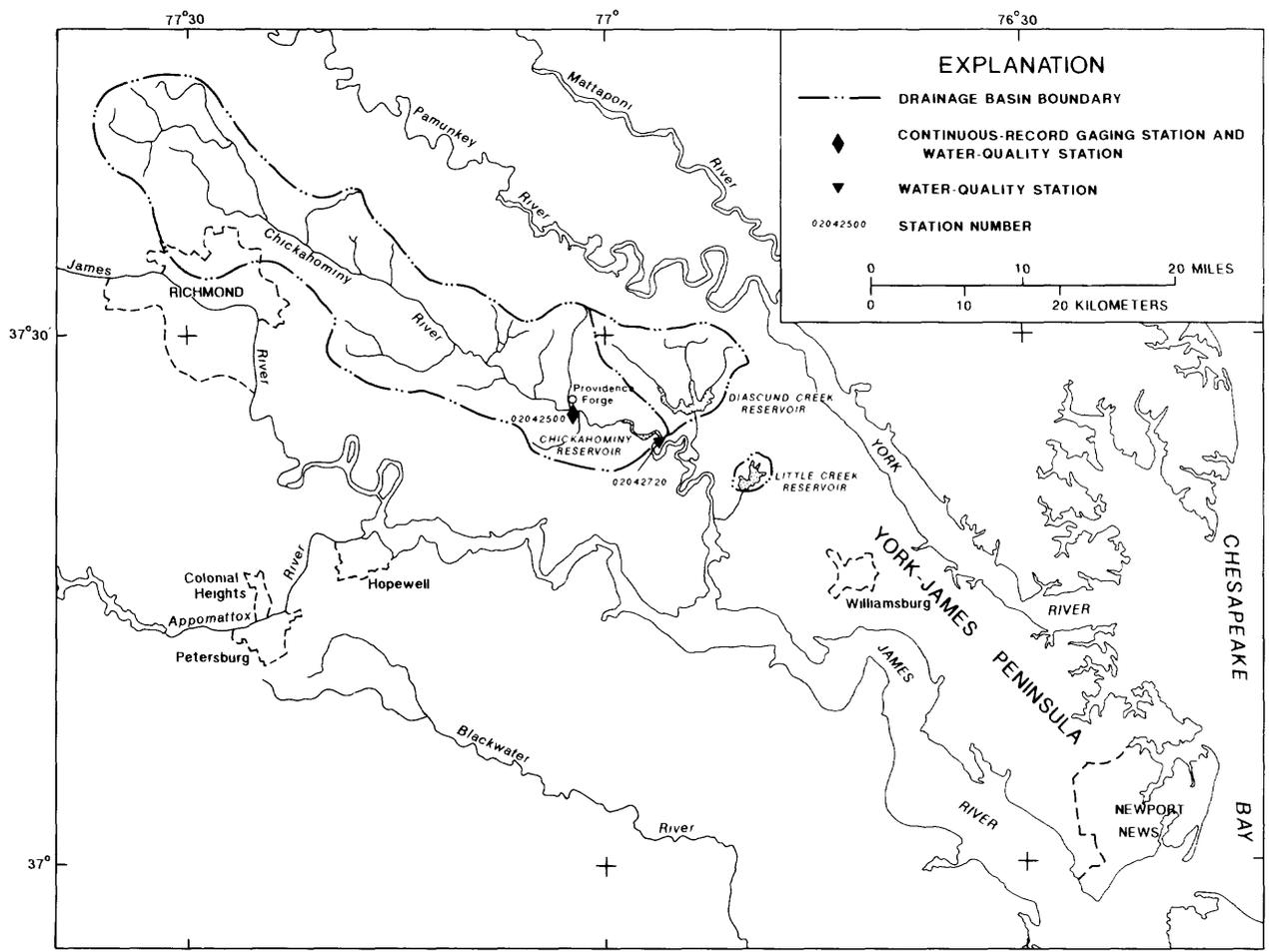


Figure 2.--Chickahominy, Diascund Creek, and Little Creek Reservoir drainage basins and sites in study area.

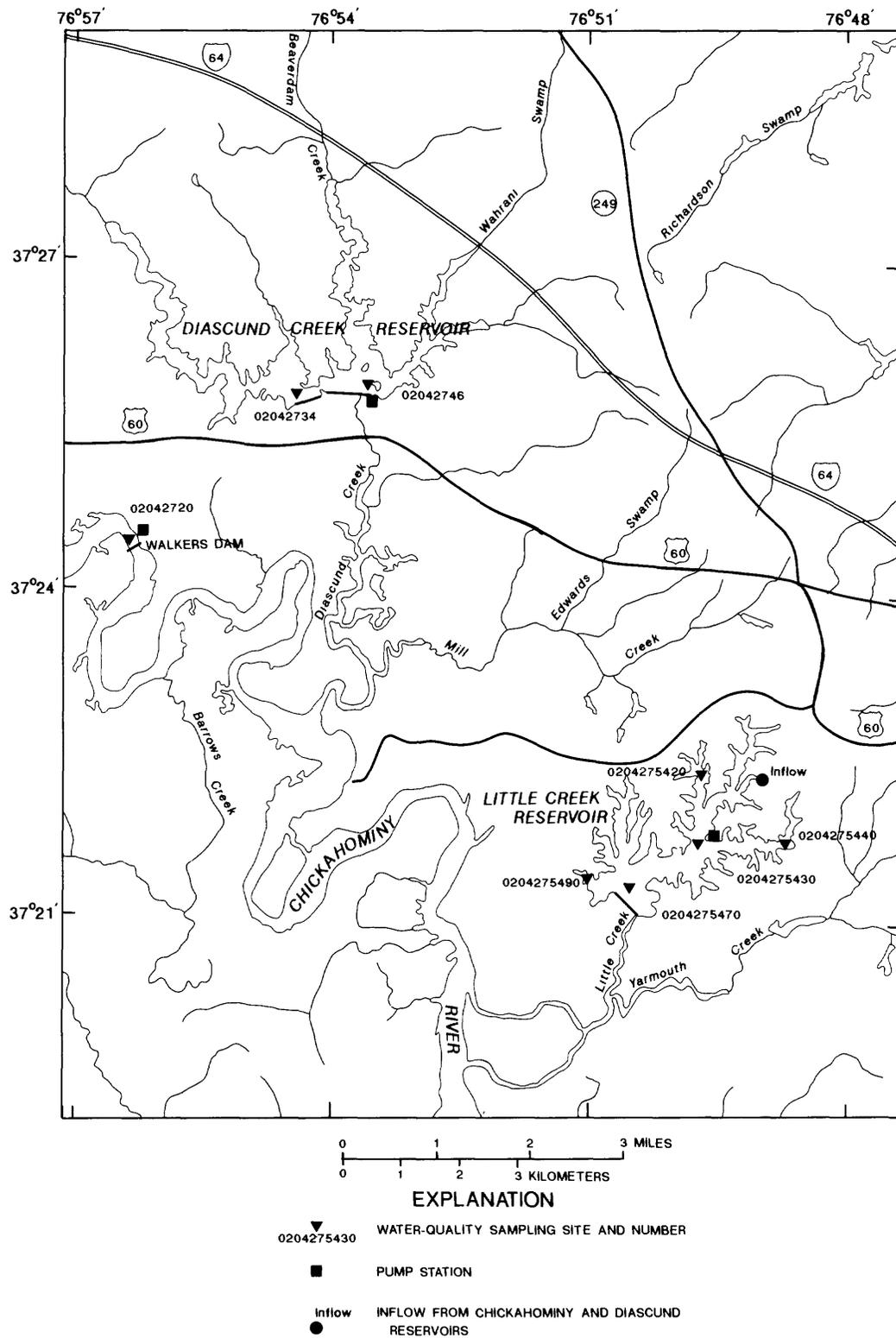


Figure 3.--Sampling sites in Chickahominy, Diascund Creek, and Little Creek Reservoirs.

CITY OF NEWPORT NEWS
WATER SUPPLY, 1983-86

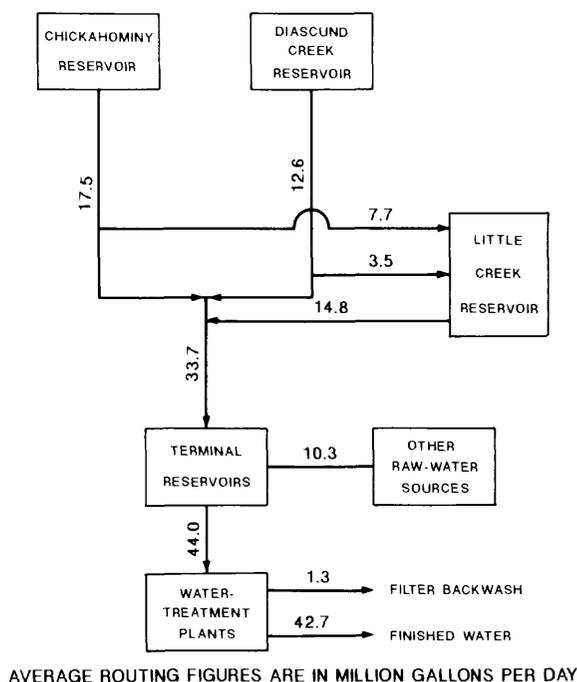


Figure 4.--Raw-water routing and average daily volumes used by city of Newport News, 1983-86.

stain plumbing fixtures and laundered clothes. Vertical gradients of temperature and dissolved oxygen (DO) were measured at several locations in Diascund Creek and Little Creek Reservoirs during onsite visits to determine reservoir mixing and stratification patterns, which affect the concentration, distribution, and speciation of many constituents. In addition, samples from selected sites were analyzed for major ions, trace metals, and silica to further characterize the chemical quality of water in these reservoirs.

Purpose and Scope

The purpose of this report is to characterize the quality and quantity of water in each reservoir, to present background data and interpretation that could be useful for future studies, and to evaluate potential effects of several raw-water-routing scenarios on the quality of water that reaches terminal reservoirs. The report focuses on water quantity and quality problems encountered by water suppliers in the Atlantic Coastal Plain physiographic prov-

ince. Hydrologic and hydrochemical factors affecting concentrations of iron, manganese, nutrients, and chlorophyll-*a* are discussed in detail.

Information concerning quantity of reservoir inflow and outflow is presented in this report. The ability of a reservoir to provide adequate quantities of good quality water is largely affected by the sources of inflow water and reservoir hydraulic retention times.

Acknowledgments

The staff of the Department of Public Utilities, city of Newport News, Virginia, are gratefully acknowledged for their help and cooperation. Particularly, the Director of Public Utilities, Charlie C. Crowder, deserves special recognition for his enthusiastic support, without which this cooperative study would not have been possible.

DESCRIPTION OF STUDY AREA

Chickahominy, Diascund Creek, and Little Creek Reservoirs are located on the York-James Peninsula in southeastern Virginia. The Chickahominy Reservoir drainage basin encompasses 305 mi² and includes parts of four counties and the northern part of the city of Richmond (fig. 2). The Diascund Creek Reservoir drainage basin encompasses 43.6 mi²; the Little Creek Reservoir drainage basin encompasses only 4.5 mi².

Land Use

Land development in the Chickahominy River basin has been scattered during the past 100 years. Development has been slow in the basin east of the city of Richmond. Most of the numerous farms that existed at the turn of the century are still in operation. These farms made up about 20 to 25 percent of the land use in 1920, and the remainder was forest and wetland (U.S. Geological Survey, 1920a, b). Although some of the farms have expanded in the basin and a few new ones have developed, the overall change in farmland acreage has been small. The most significant change in land use in the Chickahominy River basin east of Richmond has been the recent (since the 1960's) development of small suburban communities within commuting distance of Richmond and the construction of two major roads that link the city of Richmond to the Tidewater area of Virginia.

In contrast to the slow development east of Richmond, the Chickahominy River basin near the city of Richmond has changed dramatically in the past 100 years. In 1890, urban development in the basin near Richmond was negligible. Land use was limited to small farms and scattered residences, and the remainder was forest and wetland (U.S. Geological Survey, 1892; 1895). By the 1930's, the city of Richmond had expanded into the southern headwaters of the Chickahominy River (U.S. Geological Survey 1934, 1938a, 1938b, 1939, 1941, 1942). This expansion was largely residential development with a small amount of light industrial development along rail lines.

This pattern of development of the Chickahominy River basin continued into the 1960's (U.S. Geological Survey, 1964). Large residential communities grew to the west, north, and east of Richmond. Industrial development along rail lines intensified. Six small (0.25 to 1 Mgal/d) waste water-treatment

plants were built in the 1960's (Frank Miller, Chief Engineer, Department of Public Utilities, Henrico County, Va., oral commun., 1987). The effluent from these plants was discharged into the Chickahominy River during the 1960's and the early 1970's. These plants were phased out between 1969 and 1977 as sewer lines from a large waste-water treatment plant in Richmond were extended to these communities.

During the 1970's through the early 1980's, rapid suburban and industrial growth continued in the headwaters of the Chickahominy River. This development included the construction of housing projects, hospitals, warehouses, schools, an industrial park, a large chicken-processing plant, and numerous roads, including about 20 mi of Interstate 295.

Perhaps most striking about the development of the Chickahominy River basin is the lack of riverbank disturbance. This land, immediately adjacent to the Chickahominy River, is of little value for housing or farming because the broad flood plain and extensive wetlands make development risky and costly. The limited riverbank development and the preservation of wetlands has helped to minimize the effects of land development in other parts of the river basin.

Development in the Diascund Creek Reservoir drainage basin (43.6 mi²) has been slow. The basin contains several small communities and a few scattered farms and small logging operations. The single largest land-use change in the basin was the construction of 11 mi of Interstate 64 through the center of the basin during the late 1960's and early 1970's.

Little Creek Reservoir covers about one-third of the land in its drainage basin. The remainder of the basin is a mixture of forest, small farms, and residential areas. Although a relatively large percentage of the Little Creek Reservoir drainage basin is developed, the effect of this development on reservoir water quality is relatively minor; however, the steady development along the shoreline since Little Creek Reservoir was created could eventually affect the quality of water in the reservoir.

Geology

The upper 30 percent of the Chickahominy River basin lies in the Piedmont physiographic province and is underlain by microcline, biotite granite, and chlorite granodiorite of the Petersburg Granite. The Piedmont physiographic province is characterized by gently rolling hills and well-defined stream channels. Overburden thickness is variable, generally ranging from 20 to 80 ft. The lower 70 percent of the Chickahominy Reservoir drainage basin and the drainage basins of both Diascund and Little Creek Reservoirs are in the relatively flat Coastal Plain physiographic province. These areas are underlain by unconsolidated, layered sedimentary deposits, including the Yorktown, Eastover, and Calvert Formations (Mixon and others, 1989). The Yorktown and Eastover Formations are the principal exposed sedimentary deposits in the Coastal Plain, and are made up of gray, yellow, and brown sand and silt with differing amounts of intermixed and interbedded blue-to-gray clays. These formations are commonly glauconitic and phosphatic, and variably shelly (Mixon and others, 1989). The Calvert Formation is a thick (40 to 60 ft), resistant gray clay (montmorillonite and iron-rich illite) that dips gently eastward in the study area and underlies the Yorktown and Eastover Formations. The Calvert Formation is exposed or nearly exposed along many stream channels. The presence of this poorly permeable clay layer probably accounts for the broadly cut stream valleys and large wetlands in the study area. Presumably, low-energy streams, such as the Chickahominy River, cannot easily erode the clay of the Calvert Formation, and therefore cut laterally into the less resistant overlying sand and silt. This process creates wide, flat valley bottoms that do not readily transmit water.

Climate

The climate in the study area is dominated by winter storms (November through April) that track from the west and north, and by moist, tropical air masses from the Gulf of Mexico and southwest Atlantic Ocean during the remainder of the year. Large low-pressure systems dominate during the winter, producing gentle precipitation lasting as long as several days over the entire study area. In contrast, summer precipitation typically occurs as intense local showers and thunderstorms of short duration (8 hours or less).

Normal (1951-80 mean) annual precipitation across the study area ranges from about 44 to 48 in. Normal annual precipitation in Richmond, near the western edge of the study area, and Williamsburg, near the eastern edge, is 47.07 and 44.57 in., respectively (National Oceanic and Atmospheric Administration, 1986). Precipitation amounts are distributed relatively evenly throughout the year. Normal monthly amounts are greatest in July and August, averaging 5.01 in., and least from November through April, averaging 3.38 in. (table 1). Annual snowfall averages 10 in.

Table 1.--Normal (1951-80 mean) monthly precipitation at Richmond and Williamsburg

	Precipitation, in inches		
	Richmond	Williamsburg	Average
January	3.23	3.74	3.48
February	3.13	3.49	3.31
March	3.57	4.22	3.90
April	2.90	2.98	2.94
May	3.55	4.40	3.98
June	3.60	4.21	3.90
July	5.14	5.23	5.18
August	5.01	4.68	4.84
September	3.52	4.42	3.97
October	3.74	3.56	3.65
November	3.29	3.21	3.25
December	3.39	3.43	3.41
TOTAL	47.57	44.07	45.81

Air temperature in the study area varies considerably, with maximums occurring in July and minimums in January. The mean monthly maximum and minimum temperatures in July are 31 and 20 °C. In January, the mean monthly maximum and minimum temperatures are 10 and -1 °C (Virginia Department of Conservation and Economic Development, 1970).

Physical Characteristics of the Reservoirs

Forty percent of the raw water for the city of Newport News is obtained from a shallow (mean depth 5.5 ft) reservoir created by Walkers Dam on the Chickahominy River. The raw-water intake for the city is about 12 ft below the reservoir surface and is located in the center of the original river channel, 500 ft upstream of Walkers Dam. The surface area of the reservoir is 1.7 mi², and the full-pool volume is about 1,900 Mgal/d (table 2). The drainage area upstream of Walkers Dam is 305 mi².

Walkers Dam is a low-head, salinity-control structure extending about 800 ft across a tidal reach of the river. The water level upstream from the dam is typically several feet higher than downstream; however, this water-level difference changes frequently, depending on tide, river discharge, amount of pumping, and weather conditions. Occasionally, flow reversals occur with tidal water moving upstream through locks in the dam or overtopping the spillway. These flow reversals can have a substantial effect on the quality of this raw-water supply.

Diascund Creek Reservoir was created in 1963 by the construction of a earth-fill dam stretching 1 mi across a broad tidal flat of Diascund Creek. This reservoir, with its 43.6 mi² drainage basin and 3,900 Mgal of total storage, is the second largest raw-water source for the city of Newport News. Cur-

rently (1986), Diascund Creek Reservoir accounts for about 29 percent of the raw-water supply. At full-pool elevation, the surface area of Diascund Creek Reservoir is 2.0 mi², maximum depth is 24 ft, and mean depth is 9.3 ft (table 2). Full-pool elevation is governed by a concrete spillway set at 25.86 ft above mean sea level. Water from Diascund Creek Reservoir is pumped from intakes set at 12 and 22 ft below full-pool elevation. Water is usually drawn from the upper intake. As much as 33 Mgal/d can be pumped from Diascund Creek Reservoir; however, the amount withdrawn varies during the year as customer usage changes and operators adjust raw-water withdrawals in response to quantity and quality of water in each source.

Three major tributaries drain into Diascund Creek Reservoir: Diascund Creek, Beaverdam Creek, and Wahrani Swamp. Inundation of these stream channels produced the three main reservoir arms that can be seen in figure 3. A causeway built across Diascund Creek arm divides Diascund Creek Reservoir into two parts, connected by a narrow (about 50 ft) passage. This causeway impedes mixing of water in the reservoir, which can affect the spatial variation of water quality in the reservoir. Quality of water from sampling site 02042746, in the eastern part of the reservoir, probably represents a mixture of inputs from all three major tributaries. Two of the major tributaries empty directly into the eastern part of the reservoir, and water from the third major tributary, Diascund Creek, readily

Table 2.--Physical characteristics of Chickahominy, Diascund Creek, and Little Creek Reservoirs

	Total drainage area, in square miles	Reservoir surface area, in square miles	Maximum depth, in feet	Mean depth, feet	Volume, in millions of gallons
Chickahominy Reservoir	305	1.70	18	5.5	1,900
Diascund Creek Reservoir:					
At full-pool elevation	43.6	2.01	24	9.3	3,900
Drawn down 11 feet	43.6	.78	13	4.7	770
Little Creek Reservoir	4.50	1.48	58	24	7,480

moves west to east through the causeway passage because of the location of the pump station (fig. 3). In contrast, the quality of water at sampling site 02042734, in the western part of the reservoir, probably most represents inflows from Diascund Creek.

Little Creek Reservoir was created by the construction of an earth-fill dam stretching 2,000 ft across a tidal part of Little Creek (fig. 3). Before inundation, all vegetation was removed from the land to decrease biochemical oxygen demand during the first years of reservoir operation. Filling of the reservoir began in 1980, and full-pool elevation was achieved in mid-1982. A drop-inlet spillway, set at 60.0 ft above sea level, governs the full-pool elevation. At full pool, the surface area of Little Creek Reservoir is 1.48 mi², maximum depth is 58 ft, mean depth is 24 ft, and total storage capacity is 7,480 Mgal (table 2).

Water from Little Creek Reservoir is drawn from a triple-intake pump station 1.1 mi from the dam (fig. 3). The water intakes are set at different reservoir depths (15, 30, and 45 ft below full-pool elevation) to provide flexibility of operation. For example, during thermal stratification of water in the reservoir, the lower intakes are closed to prevent withdrawal of poor-quality water from the hypolimnion. The lower intakes are used when the reservoir level drops below the upper intake. During the study period, water was pumped almost exclusively from the upper intake. Pumpage from Little Creek Reservoir averaged 14.8 Mgal/d during the study period (1983-86); however, the pump station at Little Creek Reservoir currently (1987) can pump as much as 55 Mgal/d. With the completion of another pipeline and the planned installation of a third pump, the pumping capacity at Little Creek Reservoir will increase to 75 Mgal/d.

Reservoir Inflows and Outflows

The usefulness of water-supply reservoirs is affected by the quantity of inflows and outflows, by their seasonal variability, and by temporal trends. Ideally, a system of water-supply reservoirs should meet demands for water during all seasons and during periods of extreme drought. It is not the intention of this report to determine the adequacy of this system of reservoirs to supply raw water to Newport News, although some discussion is devoted to issues that should be considered when such a determination is made. The primary purpose of this section is to discuss the water budgets of

Chickahominy, Diascund Creek, and Little Creek Reservoirs because inflow and outflow rates affect reservoir water quality. For example, slowly flushed reservoirs (those that have a long hydraulic retention time) typically lose a larger percentage of their nutrient input (and other constituents as well) than do rapidly flushed reservoirs (those that have a short hydraulic retention time). Slowly flushed reservoirs allow considerable time for settling and sorption of material to bottom sediments; however, rapidly flushed reservoirs rarely have major algal problems because algae do not have ample time to grow.

Chickahominy Reservoir

Flows in the Chickahominy River respond slowly to climatic conditions. The low relief and extensive wetlands in the basin prevent rapid runoff of precipitation and increase ground-water infiltration and storage. As a result, instantaneous peak flows are generally attenuated and typically occur 3 to 5 days after precipitation. The recession following peak flows is gradual because water stored in adjacent wetlands sustains streamflows for many days; however, flows in the Chickahominy River are low during extended dry periods as ground-water contributions decline. This reduction of flow is especially critical in the summer when ground-water discharge in the basin is largely lost by evapotranspiration in wetlands before reaching stream channels.

Historical flows

Reservoir discharge has not been measured at Walkers Dam (02042720), but outflow from Chickahominy Reservoir can be estimated for this site from discharge records collected at Chickahominy River near Providence Forge (02042500), 11 miles upstream (fig. 2). This streamflow-gaging station has operated continuously since 1943, recording discharge from the upper 83 percent (252 mi²) of the Chickahominy River basin upstream from Walkers Dam and providing an indication of reservoir inflows. Although discharge can only be estimated for the Chickahominy River at Walkers Dam, the record from Providence Forge is an indication of the timing and relative magnitude of historical and recent flow conditions from the outlet of the dam.

A summary of flow characteristics for the Chickahominy River near Providence Forge for 1943-87 is listed in table 3. The maximum recorded flow of 7,710 ft³/s was on August 15, 1955, and the minimum recorded flow of 0.7 ft³/s was on July 7, 1977. The average daily discharge for this period is 264 ft³/s, which represents an average of 14.2 in/yr of surface runoff, or about 31 percent of the normal annual precipitation. About 69 percent of precipitation is lost as evapotranspiration, deep groundwater recharge, and consumptive use. This quantity of surface runoff is typical for this area of Virginia. For example, average surface runoff in the Pamunkey River basin (station 01673000) to the north and the Blackwater River basin (02049500) to the south is 12.6 and 14.1 in/yr, respectively.

The mean monthly discharge of the Chickahominy River near Providence Forge for 1943-87 is shown in figure 5. Typically, the greatest monthly flows occur in March, averaging 470 ft³/s. Flows of this magnitude reflect the end of the recharge season (maximum ground-water storage) and a low evapotranspiration rate. Lowest monthly flows com-

monly occur in September, averaging 115 ft³/s. Flows of this magnitude reflect minimum ground-water storage and rapid rates of evapotranspiration at the end of summer.

Low-flow characteristics of streams are usually defined in terms of the lowest mean flow expected for a given duration and a specified recurrence interval. For example, the 7-day, 10-year (7Q₁₀) low flow is the discharge at the 10-year recurrence interval read from a frequency curve of annual values of the lowest average discharge for 7 consecutive days (Cushing and others, 1973). The 7Q₁₀ is a commonly cited criterion in Federal and state laws that regulate streamflow withdrawals and wastewater discharges. The 7Q₁₀ low-flow value for the Chickahominy River near Providence Forge is 3.7 ft³/s. Low-flow characteristics based on other durations and recurrence intervals are given in table 3. These flow characteristics are important when assessing the reliability of the Chickahominy River as a water supply during extended dry periods.

Table 3.--Summary flow characteristics for Chickahominy River near Providence Forge (02042500), 1943-87

	Discharge, in cubic feet per second		
	1943-87	1943-67	1968-87
Mean daily	264	258	272
Median daily	173	165	181
Maximum instantaneous	7,710	7,710	5,490
Minimum instantaneous	.7	3.4	.7
Low flow ¹			
1-day, 2-year	12	16	7.1
1-day, 10-year	3	6.9	1.6
7-day, 2-year	14	19	8.5
7-day, 10-year	3.7	8	2.1
30-day, 2-year	26	33	17
30-day, 10-year	6.9	13	4.1
120-day, 2-year	72	78	66
120-day, 10-year	24	25	23

¹ See text for explanation of low-flow statistics.

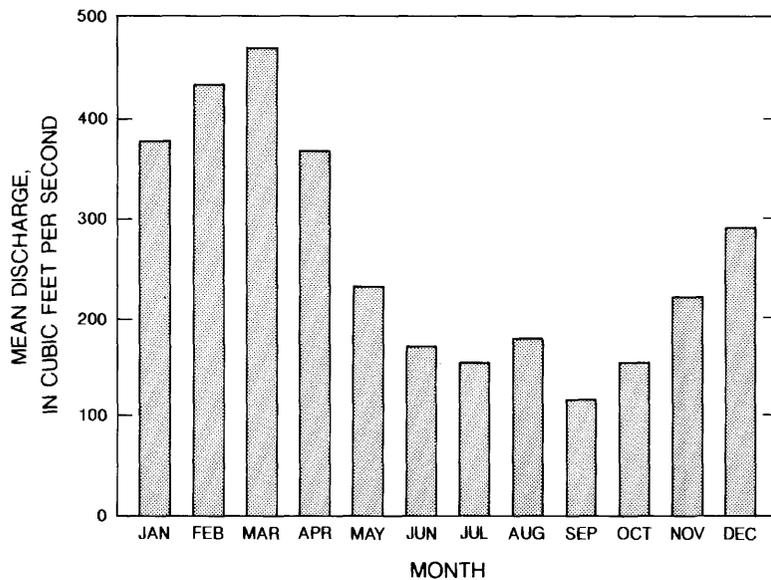


Figure 5.-- Mean monthly discharge of Chickahominy River near Providence Forge (02042500), 1943-87

Flows during 1983-86

During the 1983-86 study period, average daily discharge at Chickahominy River near Providence Forge (sampling site 020402500) was near the long-term average (1943-87); however, three distinct periods of above-normal flow and two distinct periods of below-normal flow occurred during 1983-86 (table 4). During the first 3 months of the study (August through October 1983), flows were very low. In August and September, flows averaged about 6 ft³/s, which is more than 90 percent below normal for these months. From November 1983 through June 1984, flows were well above normal because of intense rainfall. Flows averaged more than 1,000 ft³/s in March and April of 1984. This wet period was followed by 13 months of below-normal flow, from July 1984 through July 1985, with extreme dry conditions during September and October 1984, and in May and July 1985. During these 4 months, flows averaged 80 to 85 percent below normal. After the second dry period (May and July 1985), came 5 months (August through December 1985) of above-normal flows, which were followed by a 10-month period (January through October 1986) of below-normal flows. Flows were also very low in June and July 1986 (table 4).

Long-term trends

Discharge of the Chickahominy River during dry periods has decreased at the Providence Forge gaging station since 1940. For instance, the 7-day, 2-year (7Q₂) low flow for 1943-67 and 1968-87 were 19 and 8.5 ft³/s, respectively (table 3). Decreases in other low-flow characteristics for 1943-87 are also evident. These decreases do not appear to be related to long-term climatic trends because similar patterns have not been noted at nearby gaging stations.

The Kendall test for trends (Hirsch and others, 1982) on 7-day low-flow values for 1943-87 shows a 16 ft³/s reduction in discharge. This trend, which is statistically significant ($p < 0.05$), indicates a loss of about 10 Mgal/d from the Newport News raw-water supply during dry periods. Because the capacity of a water supply is based on low-flow characteristics, the decreasing low flows in the Chickahominy River need to be considered when calculating the capacity of the Newport News water supply.

Several factors could contribute to the observed decrease in low-flow values. The water-table aquifer, which is the primary source of water for sustainment of streamflows during dry periods,

Table 4.--Mean monthly discharge of Chickahominy River near Providence Forge (02042500) for 1983-86, and the percentage above or below the long-term (1943-87) mean-monthly discharge

	1983		1984		1985		1986	
	Mean discharge, in cubic feet per second	Percentage above (+) or below (-) long-term mean	Mean discharge, in cubic feet per second	Percentage above (+) or below (-) long-term mean	Mean discharge, in cubic feet per second	Percentage above (+) or below (-) long-term mean	Mean discharge, in cubic feet per second	Percentage above (+) or below (-) long-term mean
January	251	-34	621	+64	240	-37	242	-36
February	482	+11	575	+33	418	-3	472	+9
March	644	+37	1,025	+114	151	-68	280	-40
April	866	+136	1,152	+213	108	-71	154	-58
May	286	+24	299	+30	34.9	-85	125	-46
June	292	+71	353	+106	107	-37	32.6	-81
July	62.8	-60	78.3	-50	30.1	-81	16.3	-90
August	5.8	-97	165	-7	454	+156	129	-27
September	5.8	-95	22.7	-80	157	+38	28.8	-75
October	56	-63	18.8	-88	402	+164	27	-82
November	241	+10	72.9	-67	768	+249	79	-64
December	409	+41	203	-30	370	+27	448	+54
Total	298 (16.3 inches)	+13	379 (20.8 inches)	+44	269 (14.7 inches)	+2	168 (9.2 inches)	-36

could be receiving less recharge. Urbanization of the headwaters could possibly increase stormwater runoff and decrease infiltration of precipitation. Long-term pumpage of water from confined aquifers in the Coastal Plain could also possibly decrease flows in the Chickahominy River during dry periods (Laczniak and Meng, 1988). The pumpage could possibly capture ground water that would normally discharge to the river, or it could capture river water in certain reaches near outcrops or subcrops of the pumped aquifers. It is possible that the decrease in low flows can also be partly explained by the closing of several small wastewater-treatment plants between 1969 and 1977 that discharged into the Chickahominy River. These plants, which treated water that originated from the James River, could have significantly supplemented flows during dry periods.

Hydraulic retention time

Flow conditions affect the hydraulic retention time of Chickahominy Reservoir. (Hydraulic retention time is the time that is required for outflows to displace one reservoir volume.) The mean hydraulic retention time of Chickahominy Reservoir, 9 days, corresponds to a mean flow near Providence Forge of 264 ft³/s; however, during dry periods, such as the 120-day, 2-year low flow (table 3), the hydraulic retention time is 33 days. During extremely dry periods, such as the 120-day, 10-year low flow, the hydraulic retention time is 100 days. In contrast, the hydraulic retention time of Chickahominy Reservoir is less than 1 day when the average streamflow at Providence Forge exceeds 2,400 ft³/s. An average flow of 1,000 ft³/s at Providence Forge, which is common, results in a hydraulic retention time of 2 to 3 days.

Diascund Creek Reservoir

Diascund Creek Reservoir receives runoff from three main tributaries, Beaverdam Creek, Wahrani Swamp, and Diascund Creek, the largest (fig. 3). If the surface runoff carried by these major tributaries (and several smaller ones) equals the runoff of Chickahominy River near Providence Forge (14.2 in/yr), then Diascund Creek Reservoir receives about 10.8 billion gallons of surface runoff during a typical year. On the basis of this assumption, the mean hydraulic retention time of Diascund Creek Reservoir is about 130 days; however, hydraulic retention time varies considerably during the year. If the seasonal distribution of runoff to Diascund Creek is similar to that of the Chickahominy River, then the hydraulic retention time of Diascund Creek Reservoir averages about 210 days from June through November and 90 days from December through May. These estimates of hydraulic retention time are based on the assumption that direct reservoir losses and gains from evaporation and precipitation offset each other.

From August 1983 through March 1984, the water level in Diascund Creek Reservoir was kept 11 ft below full pool to allow for spillway repairs. The pool elevation was held constant by matching pumpage out of the reservoir to input from precipitation and tributary inflow. The volume of water in Diascund Creek Reservoir was 770 Mgal (table 2) during drawdown, which is about 20 percent of the full-pool volume. Because of the decreased volume, the hydraulic retention time of Diascund Creek Reservoir was very short, averaging only 17 days for the drawdown period. From August through November 1983, however, the hydraulic retention time was longer, an average of 90 days, because of below-normal flows.

Little Creek Reservoir

Information on inflow and outflow quantities of Little Creek Reservoir is more detailed than that for the other two reservoirs studied. Monthly hydrologic budgets are listed in table 5 for 1982, when the reservoir was being filled, through 1987. Data outside the study period are included in table 5 to provide insight into the response of Little Creek Reservoir to different hydrologic conditions. Monthly figures for pumpage into Little Creek Reservoir from Chickahominy and Diascund Creek Reservoirs were obtained from the city of Newport News. Data on the quantity of precipitation falling

on Little Creek Reservoir were obtained from a rain gage at Diascund Creek Reservoir (6 mi northwest of Little Creek Reservoir) that was read daily by personnel from the city of Newport News. Evaporation from Little Creek Reservoir was calculated as 70 percent of the pan evaporation (Linsley and others, 1975) that was recorded at the Kerr Reservoir, 90 mi southwest of Little Creek Reservoir. Leakage through the overflow structure of Little Creek Reservoir is constant at 11 Mgal per month. Change in reservoir storage was determined from reservoir water levels (recorded by the city of Newport News) and the reservoir depth-volume curve shown in figure 6. Figure 6 was constructed by integrating a depth-area curve developed for Little Creek Reservoir from preinundation topographic maps (scale 1:24,000), as described by Wetzel (1975).

Terrestrial inflows in table 5 include surface-water inflows, which are small because of the small drainage basin, and ground-water inflows or losses. The terrestrial-inflow term can be negative when reservoir water recharges the ground-water system. The terrestrial-inflow term is calculated by difference from the following equation:

$$Q_t = Q_o + E + L + S - Q_c - Q_d - P, \quad (1)$$

where:

Q_t is combined ground-and surface-water inflows,

Q_o is pumpage out of Little Creek Reservoir,

E is evaporation from Little Creek Reservoir,

L is leakage through the overflow structure,

S is change in reservoir storage,

Q_c is pumpage into Little Creek Reservoir from Chickahominy Reservoir,

Q_d is pumpage into Little Creek Reservoir from Diascund Creek Reservoir, and

P is direct precipitation on Little Creek Reservoir.

Because the major components of equation 1, namely, precipitation and pumpage into and out of the reservoir, can be measured accurately, the calculated value for terrestrial inflow can be determined with considerable accuracy.

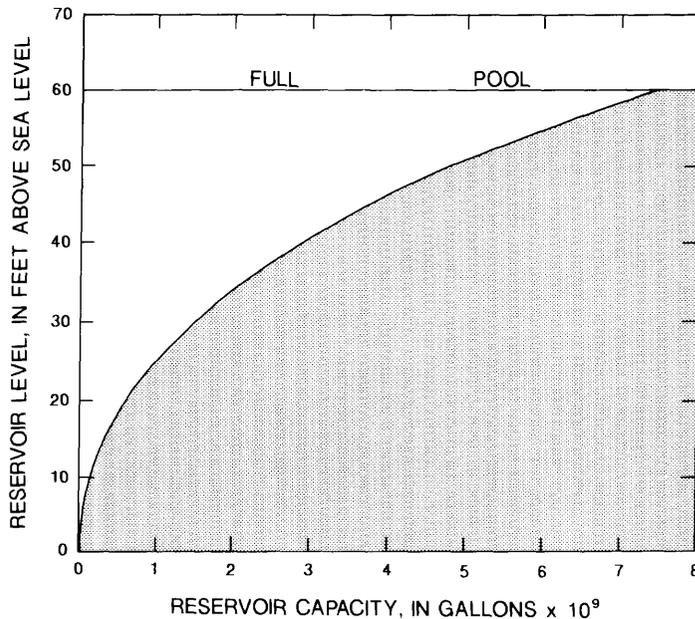


Figure 6.--Depth-volume curve for Little Creek Reservoir.

Terrestrial inflow to Little Creek Reservoir averaged about 830 Mgal/yr from July 1982 through June 1987, which equates to an inflow of 15.8 in/yr for the 3.0 mi² terrestrial drainage basin. This 5-year period was used in the calculation because the net change in reservoir storage was approximately zero, and thereby simplified the estimate. Runoff carried by the Chickahominy River during this period averaged 15.9 in/yr, which is remarkably similar to the estimate of terrestrial inflow to Little Creek Reservoir. The fact that equation 1 balances with a reasonable value for terrestrial inflow indicates that the equation includes important water-budget terms. Presumably, the loss of water from Little Creek Reservoir to a regional ground-water flow system is negligible.

Variations in terrestrial inflows to Little Creek Reservoir follow predictable and consistent patterns, further indicating that this term in equation 1 accurately reflects physical processes in the reservoir. For example, terrestrial inflows are greatest during months of intense precipitation and little evapotranspiration. Much of the variability in terrestrial inflows, however, is related to changing reservoir water levels. As reservoir water levels declined in 1983, 1984, and 1987, terrestrial inflows

were relatively high; as reservoir water levels increased in 1982, 1985, and 1986, terrestrial inflows were relatively low (table 5). This correlation indicates a good hydraulic connection between Little Creek Reservoir and local ground water stored in the surrounding aquifer. Thus, water probably moves in and out of local ground-water storage within a few months after reservoir water levels change. Changes in ground-water storage are quantified in table 6 for selected periods of reservoir fluctuation. As the reservoir water level fluctuates between full pool (60 ft above sea level) and 50 ft, about 400 to 500 Mgal of water enter and leave local ground-water storage. The change in ground-water storage in table 6 is estimated by subtracting monthly terrestrial inflows given in table 5 from the 5-year average terrestrial inflow, estimated at 68 Mgal per month. Positive change-in-storage values in table 6 indicate ground-water recharge; negative change-in-storage values indicate ground-water discharge to Little Creek Reservoir.

The rapid movement of water between Little Creek Reservoir and the surrounding water-table aquifer as the reservoir water level fluctuates has two implications for reservoir management. First, it points out that "usable storage" in Little Creek Res-

Table 5.—Monthly hydrologic budgets for Little Creek Reservoir, 1982-87

[Inflow: Chick, pumped from Chickahominy River; Diascund, pumped from Diascund Creek Reservoir; Precip, precipitation falling on Little Creek Reservoir; Terrest, ground-water and surface-water inputs from the terrestrial part of the Little Creek Reservoir drainage area (calculated by difference); Outflow: Pumpage, pumped out of Little Creek Reservoir; Evapor, evaporation from Little Creek Reservoir; Leakage, leakage through overflow structure; All figures represent millions of gallons per year.]

	Inflow				Outflow			Change in storage
	Chick	Diascund	Precip	Terrest	Pumpage	Evapor	Leakage	
1982								
Jan	758	0	85	19	0	0	11	851
Feb	504	0	99	-4	0	0	11	588
Mar	249	740	98	9	0	0	11	1,085
Apr	0	397	71	-23	0	91	11	343
May	0	29	84	30	0	112	11	20
June	590	297	134	-14	0	104	11	892
July	330	0	144	154	637	110	11	-130
Aug	209	0	145	85	276	101	11	51
Sept	262	365	36	81	783	85	11	-135
Oct	0	991	36	90	1,281	46	11	-221
Nov	0	899	66	49	892	0	11	111
Dec	0	29	82	-11	0	0	11	89
Total	2,902	3,747	1,080	465	3,869	649	132	3,544
1983								
Jan	0	52	46	-33	0	0	11	54
Feb	0	14	83	43	0	0	11	129
Mar	0	0	124	109	0	64	11	158
Apr	0	0	155	-10	0	113	11	21
May	0	0	75	69	0	115	11	18
June	0	93	81	72	270	136	11	-171
July	260	661	28	159	1,665	136	11	-704
Aug	1,144	0	121	172	1,630	129	11	-333
Sept	398	78	90	180	1,489	93	11	-847
Oct	133	434	77	259	1,344	64	11	-516
Nov	918	379	151	110	597	0	11	950
Dec	206	854	102	144	788	0	11	507
Total	3,059	2,565	1,133	1,274	7,783	850	132	-734
1984								
Jan	258	686	121	80	440	0	11	694
Feb	88	64	96	6	0	0	11	243
Mar	59	0	218	-71	0	0	11	195
Apr	0	0	132	-53	0	77	11	-9
May	286	27	117	83	513	114	11	-125
June	671	0	81	179	895	131	11	-106
July	0	0	121	178	0	115	11	173
Aug	381	118	72	164	1,476	94	11	-846
Sept	1,042	79	37	85	1,081	80	11	71
Oct	1,210	0	12	86	1,085	51	11	161
Nov	981	0	77	-80	730	0	11	237
Dec	0	0	45	43	291	0	11	-214
Total	4,976	974	1,129	700	6,511	662	132	474

Table 5.—Monthly hydrologic budgets for Little Creek Reservoir, 1982-87—Continued

[Inflow: Chick, pumped from Chickahominy River; Diascund, pumped from Diascund Creek Reservoir; Precip, precipitation falling on Little Creek Reservoir; Terrest, ground-water and surface-water inputs from the terrestrial part of the Little Creek Reservoir drainage area (calculated by difference); Outflow: Pumpage, pumped out of Little Creek Reservoir; Evapor, evaporation from Little Creek Reservoir; Leakage, leakage through overflow structure; All figures represent millions of gallons per year.]

	Inflow				Outflow			Change in storage
	Chick	Diascund	Precip	Terrest	Pumpage	Evapor	Leakage	
1985								
Jan	0	0	107	10	27	0	11	79
Feb	59	0	89	49	0	0	11	186
Mar	213	0	52	57	0	0	11	311
Apr	0	0	14	88	0	115	11	-24
May	0	0	58	37	0	118	11	-34
June	0	0	109	21	0	128	11	-9
July	0	0	112	147	280	128	11	-160
Aug	0	0	195	15	0	109	11	90
Sept	23	88	204	99	687	73	11	-357
Oct	154	90	149	77	0	51	11	408
Nov	0	0	187	-45	0	0	11	131
Dec	0	0	24	-13	0	0	11	0
Total	449	178	1,300	542	994	722	132	621
1986								
Jan	0	0	83	-57	0	0	11	15
Feb	0	0	77	63	306	0	11	-177
Mar	0	0	46	55	141	0	11	-51
Apr	167	0	40	164	673	86	11	-399
May	894	210	71	93	1,168	104	11	-15
June	341	475	11	175	1,465	132	11	-606
July	0	206	168	187	1,394	134	11	-978
Aug	6	0	286	83	740	95	11	-471
Sept	314	0	18	101	239	76	11	107
Oct	0	243	72	-29	0	58	11	217
Nov	480	30	50	4	0	0	11	553
Dec	612	150	126	11	219	0	11	669
Total	2,814	1,314	1,048	850	6,345	685	132	-1,136
1987								
Jan	1,132	0	164	-43	243	0	11	999
Feb	0	0	54	3	0	0	11	46
Mar	0	0	58	14	0	0	11	61
Apr	0	0	152	112	219	65	11	-31
May	0	0	188	83	323	98	11	-161
June	98	0	110	94	0	130	11	161
July	112	308	78	91	836	129	11	-387
Aug	0	0	102	152	1,153	118	11	-1,028
Sept	115	65	78	201	855	76	11	-483
Oct	0	901	47	105	1,012	55	11	-25
Nov	345	473	81	208	1,197	0	11	-101
Dec	1,291	0	66	-29	880	0	11	437
Total	3,093	1,747	1,178	991	6,718	671	132	-512

Table 6.--Estimated change of ground-water storage in the aquifer surrounding Little Creek Reservoir resulting from reservoir water-level fluctuations

Period of water-level change in Little Creek Reservoir	Beginning reservoir water level, in feet above sea level	Ending reservoir water level, in feet above sea level	Change in ground-water storage, in millions of gallons ¹
7/1/83 - 10/31/83	59.6	50.7	-494
2/1/86 - 9/30/86	60.1	50.6	-369
10/1/86 - 4/30/87	50.6	59.9	411
7/1/87 - 11/30/87	59.9	52.6	-412

¹ Positive values indicate recharge; negative values indicate discharge.

See text for a complete explanation.

ervoir could be 5 to 10 percent greater than the bathymetry indicates. Second, it points out the need to protect ground-water resources surrounding Little Creek Reservoir if reservoir water quality is to be protected. The quality of Little Creek Reservoir would be most vulnerable to adverse effects of drawdown during periods of thermal stratification when ground-water inflows would concentrate in the epilimnion.

The mean annual water budget (1983-86) for Little Creek Reservoir is listed in table 7. Inflows to the reservoir were dominated by pumpage from Chickahominy Reservoir (46 percent), with smaller amounts coming from Diascund Creek Reservoir (21 percent), direct precipitation (19 percent), and terrestrial inflows (14 percent). Outflows from Little Creek Reservoir were dominated by pumpage to terminal reservoirs (86 percent), with smaller amounts lost as evaporation (12 percent) and leakage through the overflow structure (2 percent). These are average figures, however; inflows and outflows differed considerably on a monthly and an annual basis during the study (table 5). For example, pumpage into Little Creek Reservoir totaled 627 Mgal in 1985, considerably less than the average annual pumpage of 5,234 Mgal in 1983, 1984, and 1986. Even during years when large quantities of water were pumped into Little Creek Reservoir, pumpage was zero for many months. In addition, pumpage into Little Creek Reservoir differed considerably by source during the study (1983-86). For

example, inflows from Diascund Creek Reservoir varied from a high of 32 percent in 1983, when Diascund Creek Reservoir was pumped extensively to allow for spillway repairs, to a low of 7 percent in 1985, when Little Creek Reservoir was largely unused.

The amount of water pumped into Little Creek Reservoir from outside sources depends on the reservoir's intended function and strongly affects the hydraulic retention time of the reservoir. When Little Creek Reservoir functions solely as emergency storage, the hydraulic retention time is long because water from Chickahominy and Diascund Creek Reservoirs is delivered only to replenish reserves. When inflows from these reservoirs are zero, the retention time averages 6 years. In 1985, the hydraulic retention time averaged 4 years; therefore, only 25 percent of the water in Little Creek Reservoir was replaced. When large amounts of water from Chickahominy and Diascund Creek Reservoirs are pumped through Little Creek Reservoir to terminal reservoirs, the hydraulic retention time decreases. With this raw-water routing scheme, Little Creek Reservoir functions as a sink or "settling basin" for various constituents and thereby improves the quality of water delivered to the terminal reservoirs. The hydraulic retention time of Little Creek Reservoir decreases to 7 months when outside inflows average 30 Mgal/d. During the study, combined inflows

Table 7.—Mean annual hydrologic budget for Little Creek Reservoir for the study period 1983-86

[Inflow: Chick, pumped from Chickahominy River; Diascund, pumped from Diascund Creek Reservoir; Precip, precipitation falling on Little Creek Reservoir; Terrest, ground-water and surface-water inputs from the terrestrial part of the Little Creek Reservoir drainage area (calculated by difference). Outflow: Pumpage, pumped out of Little Creek Reservoir; Evapor, evaporation from Little Creek Reservoir; Leakage, leakage through overflow structure; —, no data. All figures represent millions of gallons per year.]

	Inflow				Outflow			Change in storage
	Chick	Diascund	Precip	Terrest	Pumpage	Evapor	Leakage	
Annual	2,825	1,258	1,153	841	5,408	730	132	-193
Percent of total	46	21	19	14	86	12	2	--

from Chickahominy and Diascund Creek Reservoirs averaged 11.2 Mgal/d, which resulted in a 16-month hydraulic retention time.

METHODS OF STUDY

Samples from Chickahominy Reservoir (02042720) were collected at the Newport News raw-water intake when the pump station at the reservoir was operating. The intake is in the middle of the reservoir at a depth of about 12 ft below the water surface (fig. 3). During periods when the pump station was not operating, grab samples were obtained off a pier that extends about 50 ft into the water near the pump station. Water quality at the raw-water intake and the pier are similar, as indicated by the data shown in table 8. During a 3-year period, synoptic measurements of specific conductance, water temperature, and chlorophyll-*a* were nearly identical at the two sampling locations.

Two stations were selected to characterize water quality in Diascund Creek Reservoir. Sampling site 02042734 characterizes water quality upstream from the causeway, and sampling site 02042746 characterizes water quality near the Newport News raw-water intake (fig. 3). Samples were collected from each site at depths of 3, 10, and 18 ft below the water surface. During the first 8 months of the study, samples could not be collected from the 18-ft depth because the reservoir was drawn down about 11 ft to allow for repairs to the spillway. The water level in Diascund Creek Reservoir was near full-pool elevation by May 1984.

Early in the study, characterization of the quantity and quality of water in Little Creek Reservoir was emphasized, which included assessment of the vertical and spatial variability of the water quality. Samples were collected from three to five depths at each of the five sites shown in figure 3. Sampling site 0204275430 represents water-quality conditions at the Newport News raw-water intake, and site 0204275470 represents conditions in the main reservoir body. The three remaining sites were selected to characterize water quality in reservoir arms with differing amounts of nearshore development. Sites 0204275440 and 0204275490 are in the arms with the most and least amount of nearshore development, respectively, and site 0204275420 is in an arm with an intermediate amount of development. Sites were located over original stream channels to maximize sampling depths.

After 17 months of sampling from the five original sites on Little Creek Reservoir, sufficient information was gained to characterize the spatial variability of water quality. Sampling sites 0204275420, 0204275440, and 0204275490 were discontinued to avoid redundancy in data and to be consistent with sampling activities at Chickahominy and Diascund Creek Reservoirs.

Sampling Procedures

Thirty-nine trips were made during the 39-month study to sample the water quality at the eight sites. Frequency of sample collection varied during the study as data needs changed. Samples were collected every 2 to 4 weeks during the first 17

Table 8.--Summary of chlorophyll-a concentrations, specific conductance, and water temperatures on the same days at the raw-water intake and a nearby pier in Chickahominy Reservoir, 1983-86

	Chlorophyll-a, in micrograms per liter (n = 25)		Specific conductance, in microsiemens per centimeter at 25 Celsius (n = 29)		Water temperature, in degrees Celsius (n = 29)	
	Mean	Median	Mean	Median	Mean	Median
Raw-water intake	15	13	109	100	20.0	23.0
Pier	17	15	110	101	21.0	23.5

months to determine the temporal variability of water quality and the optimal sampling frequency for the remainder of the study. During the remainder of the study, sample collection was adjusted to every 4 to 8 weeks; the most frequent sampling occurred during periods of reservoir stratification (spring and summer).

Samples from Diascund Creek and Little Creek Reservoirs were collected from a boat by use of a peristaltic pump connected to a 60-ft length of weighted vinyl tubing. This apparatus allowed water at each site to be drawn directly from the depths shown in table 9. Sampling progressed from surface to bottom at each site, which follows the general pattern of decreasing water quality with reservoir depth. This sampling order minimized the chances of cross-contamination of samples. The tubing was allowed to flush several times at each sampling depth before bottles were filled to further minimize cross-contamination of samples. Bottles for unfiltered water samples were filled directly. Filtered samples were obtained by attaching an in-line 102-mm filter press to the peristaltic pump and pumping water through a 0.45- μ m membrane filter. This procedure prevents air from coming in contact with sample water before filtration; exclusion of air from the sample is critical for distinguishing between total and dissolved concentrations of certain constituents (particularly iron and manganese) in a sample obtained from a reducing environment. All sample bottles were field rinsed before being filled.

As noted earlier, grab samples were obtained from Chickahominy Reservoir upstream of Walkers Dam, either from the raw-water intake when the

pump station was operating or from the pier when the station was not operating. These samples were filtered onsite as needed.

All samples for bacterial analysis were collected manually in sterilized bottles from the water surface. This technique prevents the sample from coming in contact with unsterilized equipment. Samples for bacteria analysis were kept chilled until processing (within 2 hours of collection).

Onsite measurements of specific conductance, pH, temperature, and concentration of DO (dissolved oxygen) were made during each sampling trip. Specific conductance and pH were measured immediately for samples obtained from the sampling sites and depths shown in table 9. Measurements of water temperature and DO concentration were made at the site with a DO meter. At each reservoir sampling site, measurements were made at about 5-ft depth intervals to define the vertical profiles of temperature and DO. During periods of thermal stratification, however, additional measurements were made (every 1 to 2 ft) in the metalimnion, which typically has steep temperature and DO gradients. The metalimnion is the transition zone between the warm, well-mixed surface water (epilimnion) and the cooler, isolated deep water (hypolimnion).

Field instruments were calibrated before each sampling trip. Specific-conductance meter readings were checked against two standards of known conductance that bracketed anticipated onsite values to calculate the appropriate cell constant. The pH meter was calibrated to 7.0 and 4.0 standards. The

Table 9.--Summary of data-collection activities

[FLD-- ph, specific conductance, dissolved oxygen, and water temperature; NUT--selected species of phosphorus and nitrogen; CHL--chlorophyll-*a*; BAC-- fecal coliform and fecal streptococci bacteria (8/83-12/84); FE/MN--iron and manganese; ION--all major ions, plus silica and color before 1/85; only alkalinity, chloride, and silica thereafter; TOC--total organic carbon; TRACE--cadmium, copper, lead, mercury, and zinc]

Station name	Sampling depth, in feet below water surface	Period of record	Measurement or analysis
LITTLE CREEK RESERVOIR			
0204275420	3	8/83 - 12/84	FLD, NUT, CHL
0204275420	10	do.	FLD, NUT, CHL
0204275420	20	do.	FLD, NUT, CHL
0204275430	3	8/83 - 10/86	FLD, NUT, CHL, BAC, FE/MN
0204275430	10	do.	FLD, NUT, CHL, FE/MN, ION, TOC
0204275430	20	do.	FLD, NUT, CHL, FE/MN
0204275430	30	do.	FLD, NUT, CHL, FE/MN
0204275430	40	do.	FLD, NUT, CHL, FE/MN, ION, TOC
0204275440	3	8/83 - 12/84	FLD, NUT, CHL, BAC
0204275440	10	do.	FLD, NUT, CHL
0204275440	18	do.	FLD, NUT, CHL
0204275470	3	8/83 - 10/86	FLD, NUT, CHL
0204275470	10	do.	FLD, NUT, CHL
0204275470	20	do.	FLD, NUT, CHL
0204275470	35	do.	FLD, NUT, CHL
0204275470	50	do.	FLD, NUT, CHL
0204275490	3	8/83 - 12/84	FLD, NUT, CHL
0204275490	10	do.	FLD, NUT, CHL
0204275490	20	do.	FLD, NUT, CHL
DIASCUND CREEK RESERVOIR			
02042734	3	8/83 - 10/86	FLD, NUT, CHL
02042734	10	do.	FLD, NUT, CHL
02042734	18	4/84 - 10/86	FLD, NUT, CHL
02042746	3	8/83 - 10/86	FLD, NUT, CHL, BAC, FE/MN, ION ^(b) , TOC
02042746	10	do.	FLD, NUT, CHL, FE/MN, ION, TOC
02042746	18	4/86 - 10/86	FLD, NUT, CHL, FE/MN, ION, TOC
CHICKAHOMINY RESERVOIR			
02042720	^a	8/83- 10/86	FLD, NUT, CHL, BAC, FE/MN, ION, TOC, TRACE

^a When the pump station was operating, the sample was obtained from the Newport News raw-water intake, which withdrew water from about 12 feet below the water surface; when the pump station was not operating, a surface sample was obtained from the pump-station pier.

DO meter was both air-calibrated and checked against a zero DO solution. The thermistor of the DO meter was periodically compared to a calibrated laboratory-grade thermometer. Meter calibrations were checked periodically during sampling trips to correct for instrument drift.

Sample Preparation and Analysis

All samples for laboratory analysis were prepared and preserved onsite, as shown in table 10. Chlorophyll-*a* concentrations and bacteria counts were analyzed in the Richmond, Va., office of the U.S. Geological Survey. The remaining samples were iced and shipped the same day for analysis in the U.S. Geological Survey Central Laboratory in Denver, Colo. The type of measurement or analysis by sampling site, sampling depth, and period of record is given in table 9. At all sampling sites and depths, onsite measurements were made, and samples were analyzed for nutrients and chlorophyll-*a*. Nutrient analyses included dissolved nitrite plus nitrate, dissolved ammonia, total ammonia plus organic nitrogen, total phosphorus, and dissolved orthophosphorus. At sampling sites 0204275430 on Little Creek Reservoir, 02042746 on Diascund Creek Reservoir, and 02042720 on Chickahominy Reservoir, additional samples were analyzed for major ions, silica, total organic carbon, total and dissolved iron and manganese, and bacteria to further characterize water quality. Selected trace metals (cadmium, copper, lead, mercury, and zinc) were analyzed for samples collected from Chickahominy

Reservoir because its headwaters contain considerable industrial and urban development associated with Richmond. Methods for the chemical and biological analysis of samples are listed in table 11.

WATER QUALITY OF RESERVOIRS

The quality of water in the three largest reservoirs that supply raw water to the Newport News is discussed separately in the following sections. In each section, the emphasis is on describing the water quality of the reservoir, patterns of spatial and temporal variability, and the processes or factors that could affect observed water quality. The primary raw-water sources, Chickahominy and Diascund Creek Reservoirs, are discussed first. Little Creek Reservoir is discussed last because its quality is substantially affected by inflows from Chickahominy and Diascund Creek Reservoirs.

Chickahominy Reservoir

The quality of water in Chickahominy Reservoir is affected by many physical factors and chemical and biological processes. Some of the most important physical factors include climate, season, pumping from the reservoir, tidal conditions, water temperature, light conditions, and flow reversals at Walkers Dam. Important chemical processes include mineral weathering of drainage-basin rocks and soils, sorption and desorption of materials from reservoir bottom sediments, and transformation of constituents in changing chemical environ-

Table 10.--Treatment and preservation of water-quality samples

[<, less than]

Constituent(s)	Treatment and preservation ¹
Nutrients:	
Dissolved	Filtered, mercuric chloride, chilled
Total	Mercuric chloride, chilled
Major cations	Filtered, nitric acid to pH <2
Major anions and silica	Filtered
Alkalinity	No treatment
Color	Chilled
Trace metals, except mercury:	
Dissolved	Filtered, nitric acid to pH <2
Total	Nitric acid to pH <2
Mercury	Glass bottle, filtered, nitric acid to pH <2, potassium dichromate added
Total organic carbon	Glass bottle, chilled

¹ Filtered samples are passed through a 0.45 micrometer membrane filter.

Table 11.—Methods of water analysis

Nutrients ¹	
Total and dissolved phosphorus	Colorimetric, potassium persulfate wet digestion, phosphomolybdate method
Orthophosphorus	Colorimetric, phosphomolybdate method
Nitrate plus nitrate	Colorimetric, cadmium reduction, hydrazine reduction-diazotization method
Ammonia	Colorimetric, indophenol method
Ammonia plus organic nitrogen	Colorimetric, block digester-salicylate-hypochlorite method
Major cations, iron, and manganese ¹	
Dissolved	Direct atomic absorption spectrometric
Total recoverable	Atomic absorption spectrometric, hydrochloric acid wet digestion
Major anions and silica ¹	
Chloride	Colorimetric, ferric thiocyanate method
Sulfate	Turbidimetric, barium sulfate method
Fluoride	Electrometric, ion-selective electrode
Alkalinity	Electrometric, titration to pH 4.5 endpoint
Silica	Colorimetric, molybdate blue method
Trace metals, except mercury ¹	Atomic absorption spectrometric, chelation and extraction
Mercury ¹	Cold-vapor flameless atomic absorption spectrometric
Color ¹	Electrometric, visual comparison
Total organic carbon ²	Carbon analyzer, potassium persulfate wet digestion
Chlorophyll- <i>a</i> ³	Fluorometric, 90 percent acetone extraction
Bacteria ⁴	
Fecal coliform	Direct count, membrane filter, M-FC agar
Fecal streptococci	Direct count, membrane filter, KF agar

¹ Fishman and Friedman, 1985.

² Wershaw and others, 1987.

³ Strickland and Parsons, 1972.

⁴ Greeson and others, 1977.

ments. Biological processes affecting water quality include primary and secondary productivity in the reservoir and microbiological activity. All these factors and processes affect the flow of energy and the sources and sinks of constituents in the Chickahominy River drainage basin and reservoir.

Specific Conductance

Specific conductance is a measure of the ability of water to conduct an electrical charge. Consequently, specific conductance is closely related to the concentration of dissolved solids and individual ions in the water (Hem, 1985). The variability of specific conductance in Chickahominy Reservoir (fig. 7) is principally related to (1) flow reversals of salty water at Walkers Dam, (2) runoff of road salt, (3) seasonal variation of weathering rates of rocks and soils, and (4) flow conditions.

The variability of specific conductance associated with the season of the year and flow conditions is subtle in Chickahominy Reservoir and is largely obscured by the effects of flow reversals at Walkers Dam and applications of road salt. Specific conductance tends to be elevated during base-flow periods, however, because reservoir water is not diluted by surface-water inflows. For example, specific conductance was increased during base flow in July 1985, averaging about 120 $\mu\text{S}/\text{cm}$, but decreased sharply in August 1985 to 61 $\mu\text{S}/\text{cm}$ after an intense storm (fig. 7). Irrespective of flow conditions, specific conductance also tended to increase during warm months. Rates of carbonic-acid weathering of basin rocks and soils increased with temperature and resulted in increased concentrations of charge-carrying weathering products (ions) in the reservoir.

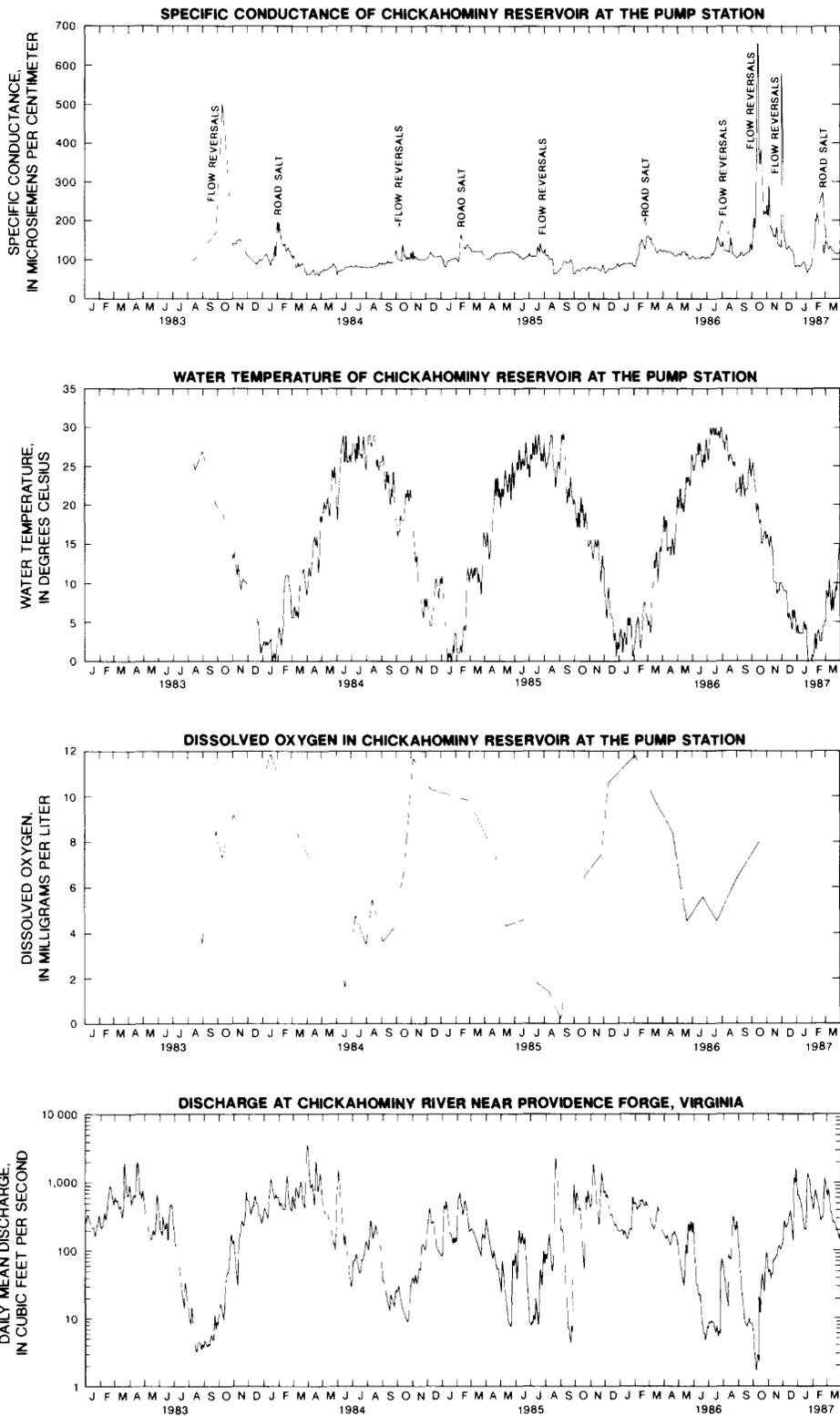


Figure 7.--Specific conductance, water temperature, and dissolved-oxygen concentration in water from Chickahominy River (020402500), and daily mean discharge at Chickahominy River near Providence Forge, 1983-87.

High specific conductances observed periodically during the winter months in Chickahominy Reservoir (fig. 7) result from runoff of road salt applied in and around the city of Richmond to remove snow and ice. Peak values in excess of $150\ \mu\text{S}/\text{cm}$ are common. The lag time between road-salt application and the arrival of the specific-conductance peak at the raw-water intake ranged from 13 to 23 days for the six largest snow or ice storms during the study (table 12). This lag time is inversely correlated ($r^2 = 0.83$) with the log of mean flow at the Providence Forge gaging station during the lag period (fig. 8). Figure 8 could be useful for estimating time of travel for peak concentrations of conservative (nonreactive) contaminants originating in Richmond to pass the raw-water intake in Chickahominy Reservoir; however, caution must be exercised when extending the relation in figure 8 much beyond the available data.

High specific conductances and chloride concentrations have occurred periodically during winter months (January through March) for many years in the Chickahominy River. Peak values, measured by the U.S. Geological Survey from 1969 through 1986 at the Providence Forge streamflow-gaging station (02042500) (fig. 2) and recorded in computer files, can all be related to snow or ice in the basin and application of road salt. The magnitude of specific-conductance and chloride-concentration peaks have increased with time at the Providence Forge station. A Kendall time-series trend test (Hirsch and others, 1982) indicates that specific conductance during the winter months increased $4.4\ \mu\text{S}/\text{cm}$ per year during this period ($p < 0.01$), which equates to an annual increase of 4.2 percent since 1969. Winter chloride concentrations increased $1.25\ \text{mg}/\text{L}$ per year ($p < 0.01$), which equates to an annual increase of 7.8 percent since 1969.

The winter increases in specific conductance and chloride concentrations in Chickahominy River result from an increased use of road salt in the basin during 1969-86. The construction of many miles of interstate highway in the basin during this period, and the rapid suburban growth in the headwaters of the Chickahominy River, undoubtedly increased the area deiced after snow and ice storms, and thus the quantity of road salt applied. More importantly, however, road-salt use in Virginia has increased markedly over the last few decades. Data from the Salt Institute in Alexandria, Va. (table 13), indicates an annual increase of 8.2 percent since the 1964-65 winter, which agrees well with the 7.8 percent

annual increase in winter chloride concentrations measured in Chickahominy River near Providence Forge for 1969-86.

Flow reversals of salty water at Walkers Dam produced the prominent specific-conductance peaks at the raw-water intake in October 1983, October 1986, and December 1986, each exceeding $500\ \mu\text{S}/\text{cm}$ (fig. 7). Flow reversals resulted in several smaller specific-conductance peaks in September and October 1984, July 1985, and July and August 1986. During flow reversals, salty water downstream from the dam moves through the locks or overtops the spillway, mixing with reservoir water. On three occasions between July 1983 and December 1986, the city of Newport News stopped withdrawing water from Chickahominy Reservoir because of high chloride concentrations (greater than $50\ \text{mg}/\text{L}$). Although such flow reversals are infrequent, they can be troublesome for Newport News because they typically occur during extended dry periods (fig. 7) when demand for drinking water is greatest and supplies from other surface-water sources are at a minimum.

Specific-conductance peaks in Chickahominy Reservoir associated with flow reversals at Walkers Dam are relatively infrequent because two conditions must be met simultaneously for their occurrence. First, salty water must be present downstream from Walkers Dam. Flow reversals at other times do not appreciably affect reservoir quality because of the similarity of water upstream and downstream from the dam. Second, for salty water to move upstream through the locks, water levels below the dam must exceed the reservoir water level. (The locks open automatically when the water level downstream from the dam exceeds the reservoir water level.) During dry periods lasting several months, a wedge of dense salty water from Chesapeake Bay can migrate upstream under the fresher waters of the James and Chickahominy Rivers (Virginia Water Control Board, 1987a, b). A specific conductance greater than $10,000\ \mu\text{S}/\text{cm}$ was measured at the mouth of Chickahominy River (24 mi downstream from Walkers Dam) in the dry summer and fall of 1986. During dry periods, the specific conductance of surface samples (depth 1 ft) collected 2,000 ft downstream from Walkers Dam are frequently greater than $500\ \mu\text{S}/\text{cm}$ (Virginia Water Control Board computer files). Although subsurface samples (greater than 1 ft depth) have not been collected

Table 12.--Lag time between road-salt application in Chickahominy River basin and peak specific conductance at the Chickahominy Reservoir raw-water intake

[$\mu\text{S}/\text{cm}$, microsiemens per centimeter at 25 degrees Celsius]

Dates of road-salt application in and around Richmond	Lag time of specific conductance peak, in days	Maximum specific conductance, in $\mu\text{S}/\text{cm}$	Mean discharge at Providence Forge gaging station during the lag period, in cubic feet per second
Jan 18-19, 1984	15	197	614
Jan 17-20, 1985	23	162	331
Jan 27-29, 1986	19	150	477
Feb 14-15, 1986	16	160	456
Jan 22-26, 1987	18	220	535
Feb 16-17, 1987	13	147	672

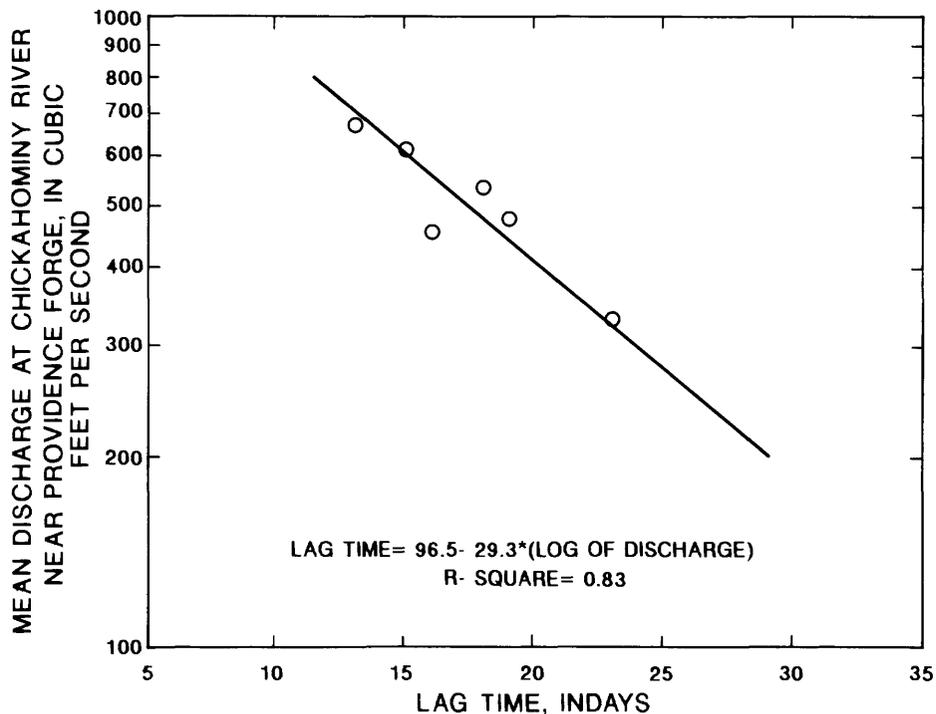


Figure 8.--Relation between flow conditions at Chickahominy River near Providence Forge (02042500) and lag time for specific-conductance peaks at Chickahominy Reservoir intake (02042720) that result from road-salt application in Richmond.

Table 13.--Road-salt use in Virginia, 1964-87

[In thousands of tons]

Winter period ¹	Sodium chloride	Calcium chloride	Total
1964-65	30	19	49
1966-67	64	22	86
1969-70	56	22	78
1973-74	63	6	69
1978-79	144	5	149
1981-82	178	2	180
1982-83	95	1	96
1986-87	250	0	250

¹ Values are not available for all winter periods. Information was obtained from the Salt Institute, Alexandria, Va.

for analysis, this wedge of salty water undoubtedly reaches Walkers Dam and enters Chickahominy Reservoir through the locks during flow reversals.

When flows at the Providence Forge gage average less than 90 ft³/s for 3 months, a wedge of salty water commonly reaches Walkers Dam. Each time these flow conditions were met during the study, at least one flow reversal brought in a significant quantity of salty (high specific conductance) water to the reservoir (table 14). In contrast, when flows averaged greater than 90 ft³/s for 3 months, no specific-conductance peaks associated with flow reversals were noted in Chickahominy Reservoir; the absence of peaks indicates the absence of salty water downstream from the dam at these times.

The water level downstream from Walkers Dam is affected by tides, weather (wind and barometric pressure), and flow conditions. The water level in the reservoir is affected by precipitation, surface-water and ground-water inflows, evaporation, and pumpage from the reservoir by the city of Newport News. Flow reversals occur most frequently during the summer and fall months when the reservoir water level is at a minimum because of low flows, high rates of evaporation, and maximum withdrawals to meet demands for drinking water. When the reservoir water level is low, flow reversals at Walkers Dam are relatively common at the maximum tide of a lunar cycle or during storms accompanied by strong easterly winds or low atmospheric pressure that cause mounding of water downstream from the dam.

Table 14.--Specific-conductance peaks at the Chickahominy Reservoir raw-water intake associated with flow reversals at Walkers Dam, 1983-86

[μS/cm, microsiemens per centimeter at 25 degrees Celsius]

Date of specific conductance peak	Specific conductance peak, in μS/cm	Mean discharge at Providence Forge gaging station for 3 months before specific-conductance peak, in cubic feet per second
11/10/83	500	11
10/15/84	138	82
07/24/85	158	55
07/25/86	158	64
08/19/86	153	55
10/13/86	653	58
12/02/86	580	44

The interplay of factors affecting water levels upstream and downstream from Walkers Dam during the last 3 months of 1986 can be seen in figure 9. This period was chosen because several reversals that affected the quality of Chickahominy Reservoir occurred within a short time. During the last 3 weeks of September 1986, flows at the Providence Forge streamflow-gaging station were low, averaging 10 ft³/s, and pumpage from the reservoir averaged 27 Mgal/d (or 42 ft³/s). These conditions contributed to a steady decline in the reservoir water level, which was as low as 2.3 ft above sea level in

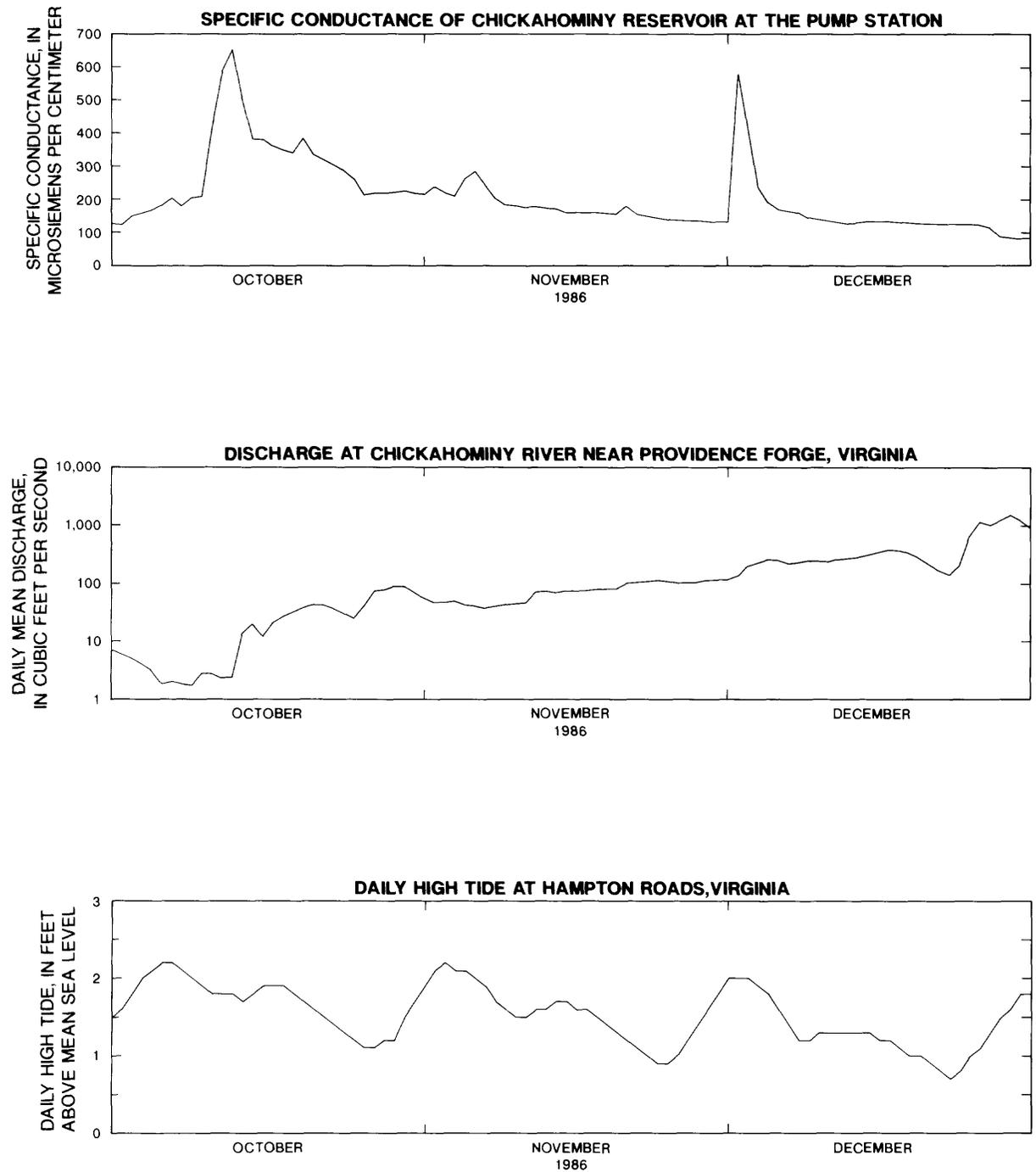


Figure 9.--Relations among high specific-conductance values at Chickahominy Reservoir intake resulting from flow reversals at Walkers Dam and streamflow conditions, tidal fluctuations, wind speed, daily pumpage, and reservoir water levels, October through December 1986.

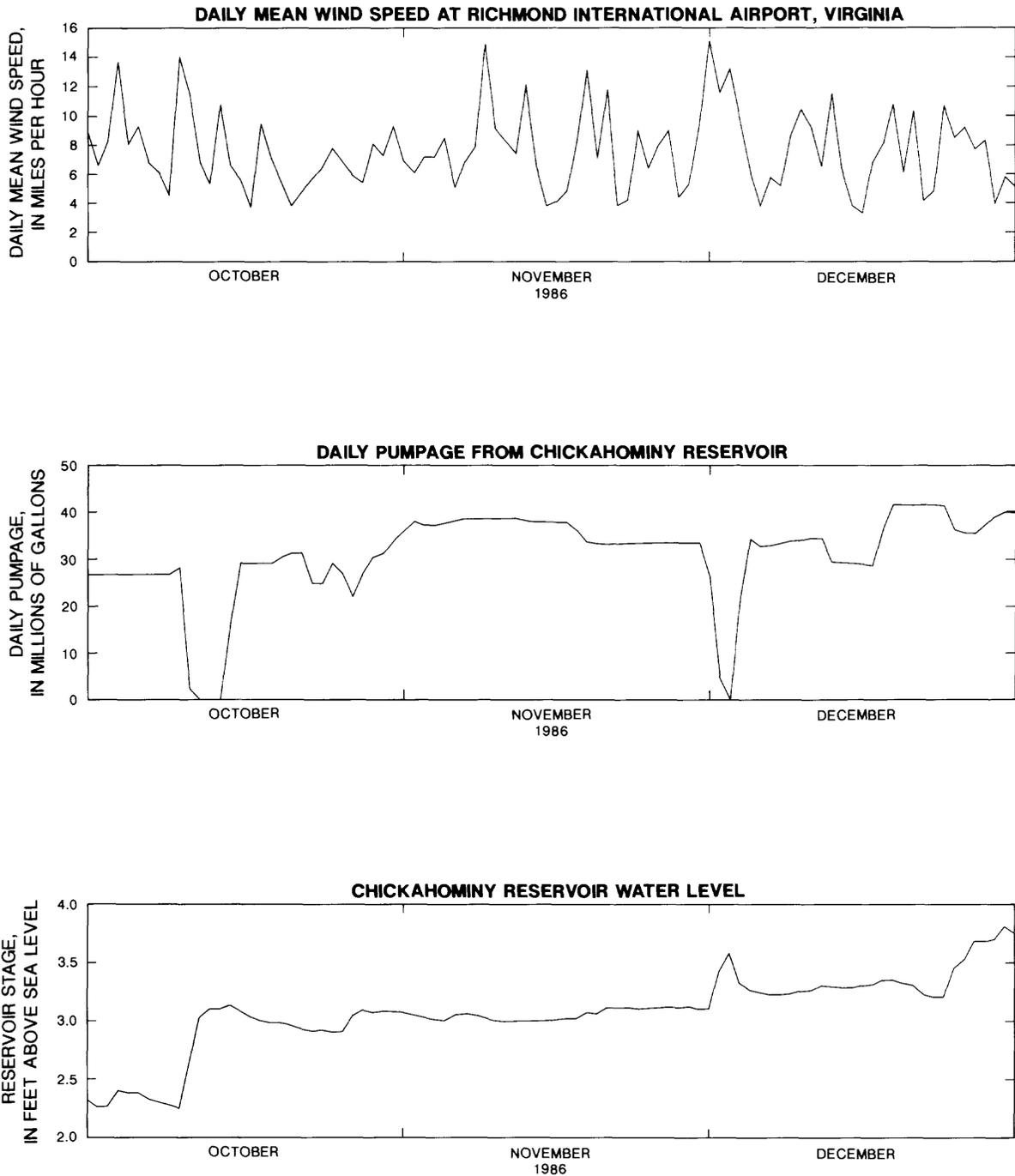


Figure 9.—Relations among high specific-conductance values at Chickahominy Reservoir intake resulting from flow reversals at Walkers Dam and streamflow conditions, tidal fluctuations, wind speed, daily pumpage, and reservoir water levels, October through December 1986.—Continued

early October 1986. Specific conductance remained constant at 120 $\mu\text{S}/\text{cm}$ as the water level declined; however, during early October 1986, multiple flow reversals occurred at Walkers Dam. These flow reversals raised the reservoir water level by 0.9 ft, and brought in sufficient salty water to increase specific conductance at the raw-water intake to greater than 600 $\mu\text{S}/\text{cm}$ (fig. 8). Flow reversals occurred on these days because (1) the reservoir water level was low, (2) high tide was near a monthly maximum (National Oceanic and Atmospheric Administration, 1985), and (3) winds were strong (National Oceanic and Atmospheric Administration, 1986) (fig. 9). Winds at the Richmond International Airport averaged about 12 mi/h during October 9-10, 1983, and corresponded to the largest flow reversals at the dam. Specific conductance of water in Chickahominy Reservoir remained high for several weeks after these flow reversals because flushing from river inflows was minimal; discharge at the Providence Forge station averaged 27 ft^3/s in October 1986, or 82 percent below the long-term mean for October (table 4).

In early November 1986, a smaller flow reversal occurred, during the maximum high tide for the month (fig. 9). Specific conductance at the raw-water intake increased from 210 to 285 $\mu\text{S}/\text{cm}$. The early November flow reversal was smaller than reversals in October 1986 because the reservoir water level was higher in November and winds were not as strong. The November flow reversal raised the reservoir water level by less than 0.1 ft.

Extremely strong winds in early December (fig. 9), together with the maximum high tide for the month, caused another large flow reversal in early December. The reservoir water level rose 0.5 ft, and specific conductance at the raw-water intake increased from 130 to 580 $\mu\text{S}/\text{cm}$. Wind speed at Richmond International Airport averaged 13 mi/h for the first 3 days of December and could have contributed the most to the magnitude of the flow reversal. In contrast to flow reversals in October 1986, specific conductance measured at the raw-water intake decreased rapidly after the December flow reversal (fig. 9) because of higher flows. Chickahominy River flows in early December averaged 240 ft^3/s at the Providence Forge station and flushed the lower part of Chickahominy Reservoir in about 5 days.

The potential for salty water entering Chickahominy Reservoir by way of flow reversals at Walkers Dam can be easily monitored. During dry periods, when the average flow at Providence Forge is less than 100 ft^3/s for 1 to 2 months, specific conductance could be measured weekly on the downstream side of Walkers Dam near the channel bottom. These measurements, made with a submersible conductivity probe, would provide early detection of a saltwater wedge on the downstream side of Walkers Dam. When salty water reaches the dam, withdrawals from Chickahominy Reservoir could be adjusted to maintain the reservoir water level at least 3.0 ft above sea level, thereby decreasing the frequency and magnitude of flow reversals at Walkers Dam.

pH and Water Temperature

The pH at the raw-water intake of Chickahominy Reservoir varied little during the study and generally ranged from 6.0 to 7.0 (fig. 10, table 15). The median pH was 6.6. Variability in pH was not associated with time of year or flow conditions. Values during dry summer periods were slightly higher than those during wet winter periods, but the differences were not statistically significant.

Water temperature in Chickahominy Reservoir at the pump station follows a consistent seasonal trend (fig. 7). Superimposed on the seasonal trend is the comparatively weak, short-term variability associated with daily changes in weather. The maximum annual water temperatures measured during the study ranged from 29 to 30 $^{\circ}\text{C}$ and typically occurred in July and August. Annual minimum water temperatures ranged from 0 to 1 $^{\circ}\text{C}$, occurring December through February (fig. 7).

Dissolved Oxygen

DO concentrations at the raw-water intake follow a consistent seasonal trend (fig. 7). During the cool months (October through March), DO concentrations were relatively high, ranging from 6.5 to 11.9 mg/L and averaging 9.1 mg/L (table 16). The percentage of DO saturation averaged 83 percent during these months. Relatively high concentrations of DO were found during the cool months because (1) water in the reservoir is mixed well and is in contact with oxygen in the atmosphere, (2) the rate of oxygen consumption by decomposition and respiration in the reservoir is low because of cool

Table 15.--Concentrations of selected dissolved constituents and physical properties of water from Chickahominy Reservoir (02042720), 1983-86

[Concentration in milligrams per liter, unless otherwise indicated; <, less than; --, no data]

Date	pH	Calcium	Magnesium	Sodium	Potassium	Alkalinity, as CaCO ₃	Sulfate, as SO ₄	Chloride	Fluoride	Silica, as SiO ₂	Hardness, as CaCO ₃	Solids
08-04-83	6.9	7.9	1.8	7.2	0.7	27	6.6	11	0.1	6.7	27	59
08-16-83	6.7	7.6	2.0	11	1.1	22	9.5	13	<.1	.9	27	58
08-31-83	6.8	7.9	2.3	12	1.4	25	10	18	.2	4.3	29	71
09-28-83	6.5	9	3.1	29	2.0	23	9.1	44	.2	4.3	35	110
10-11-83	7.0	12	7.4	75	3.7	22	24	130	.2	3.5	60	270
11-03-83	6.9	9.0	2.4	14	1.8	14	22	21	.1	7.1	32	86
11-30-83	5.8	7.4	1.8	7.8	3.3	8	25	14	.2	7.7	26	73
01-18-84	--	4.6	1.1	7.2	1.8	8	15	13	<.1	4.7	16	53
03-14-84	6.3	4.4	1.1	8.9	1.4	6	10	13	<.1	2.3	16	45
04-18-84	6.6	4.0	1.0	4.6	1.4	10	6.7	7.0	<.1	1.9	14	33
05-08-84	6.5	5.4	1.3	5.4	1.5	15	11	7.8	.1	2.0	19	45
05-29-84	6.8	6.3	1.5	6.2	1.1	20	3.9	10	.1	3.7	22	46
06-19-84	6.3	6.1	1.4	5.8	1.2	17	3.9	8.1	.1	5.7	21	44
07-10-84	7.0	6.9	1.6	5.9	1.0	24	3.5	7.7	<.1	4.8	24	46
08-01-84	6.5	7.0	1.5	5.5	.8	21	4.8	7.3	<.1	5.9	24	46
08-14-84	6.9	7.1	1.4	5.9	.6	21	5.5	4.6	<.1	8.0	23	46
09-04-84	6.8	7.7	1.6	6.8	.5	26	2.7	9.4	.1	9.8	26	55
09-26-84	6.7	8.0	1.7	8	.7	25	3.7	8.3	<.1	7.1	27	53
10-17-84	6.9	7.2	2.3	12	1.6	25	6.4	17	<.1	3.5	27	65
11-06-84	6.5	7.6	2.1	11	1.4	23	7.3	14	<.1	4.4	28	62
12-11-84	6.7	6.5	1.8	8.5	3.9	7	16	17	<.1	8.4	24	67
02-28-85	6.6	6.0	1.6	15	2.0	8	13	25	.1	1.9	22	70
04-23-85	6.9	7.8	1.7	11	1.8	21	5.5	16	<.1	1.6	26	58
05-14-85	6.1	8.9	2.0	9.9	2.0	27	4.3	17	<.1	1.0	30	62
06-20-85	6.4	7.8	1.7	9.1	1.2	21	8.5	11	<.1	6.1	26	59
07-17-85	6.6	8.5	1.8	11	1.0	24	6.4	14	<.1	6.9	29	65
08-12-85	6.8	7.9	1.6	9.5	.6	24	5.4	11	.1	10	26	61
09-04-85	6.0	6.5	1.4	5.6	1.5	16	7.0	8.6	.1	8.0	22	50
10-23-85	6.3	5.1	1.2	6.7	1.4	9	9.1	10	<.1	7.6	18	47
11-26-85	6.2	4.9	1.3	5.2	3.3	12	5.2	10	<.1	8.0	18	46
12-12-85	6.3	4.7	1.2	5.7	2.7	9	11	9.8	<.1	7.5	17	49
02-04-86	6.7	5.5	1.4	7.7	2.2	9	11	11	<.1	5.3	20	50
03-19-86	6.6	6.2	1.5	12	1.8	11	11	22	.1	.7	22	62
04-22-86	6.8	7.3	1.7	10	1.7	22	11	14	.1	1.8	25	61
05-21-86	6.6	7.7	1.7	9.7	1.5	25	8.8	10	.1	1.6	26	56
06-24-86	6.8	7.5	1.7	9.9	1.1	22	10	12	<.1	4.7	26	60
07-22-86	6.3	7.6	2.2	15	1.6	23	9.3	22	.1	2.5	28	75
09-02-86	6.2	7.8	1.5	9.6	.7	18	12	12	.1	12	26	67
10-17-86	6.7	11	5.6	46	2.5	22	20	80	.3	5.0	51	180
Mean	6.5	7.1	1.9	12	1.6	18	9.6	18	<.1	5.1	26	67
Maximum	7.0	12	7.4	75	3.9	27	25	130	.3	12	60	270
Upper quartile	6.8	7.8	1.9	11	1.9	24	11	17	.1	7.3	27	66
Median	6.6	7.4	1.7	9.1	1.5	21	9.1	12	<.1	4.8	26	59
Lower quartile	6.3	6.2	1.4	6.4	1.1	12	5.5	9.9	<.1	2.4	22	48
¹ Drinking-water regulations	6.5-8.5(a)	--	--	270(b)	--	--	250(a)	250(a)	4.0(c)	--	100(b)	500(a)

¹ Drinking-water regulations are as follows:

(a) Secondary maximum contaminant level (U.S. Environmental Protection Agency, 1986b).

(b) Nonnational drinking-water standard or criterion established. Hardness concentrations greater than 100 mg/L are objectionable for domestic purposes (Hem, 1985), and sodium concentrations greater than 270 mg/L are not recommended for humans on salt-restricted diets (U.S. Environmental Protection Agency, 1986c).

(c) Maximum contaminant level (U.S. Environmental Protection Agency, 1986a).

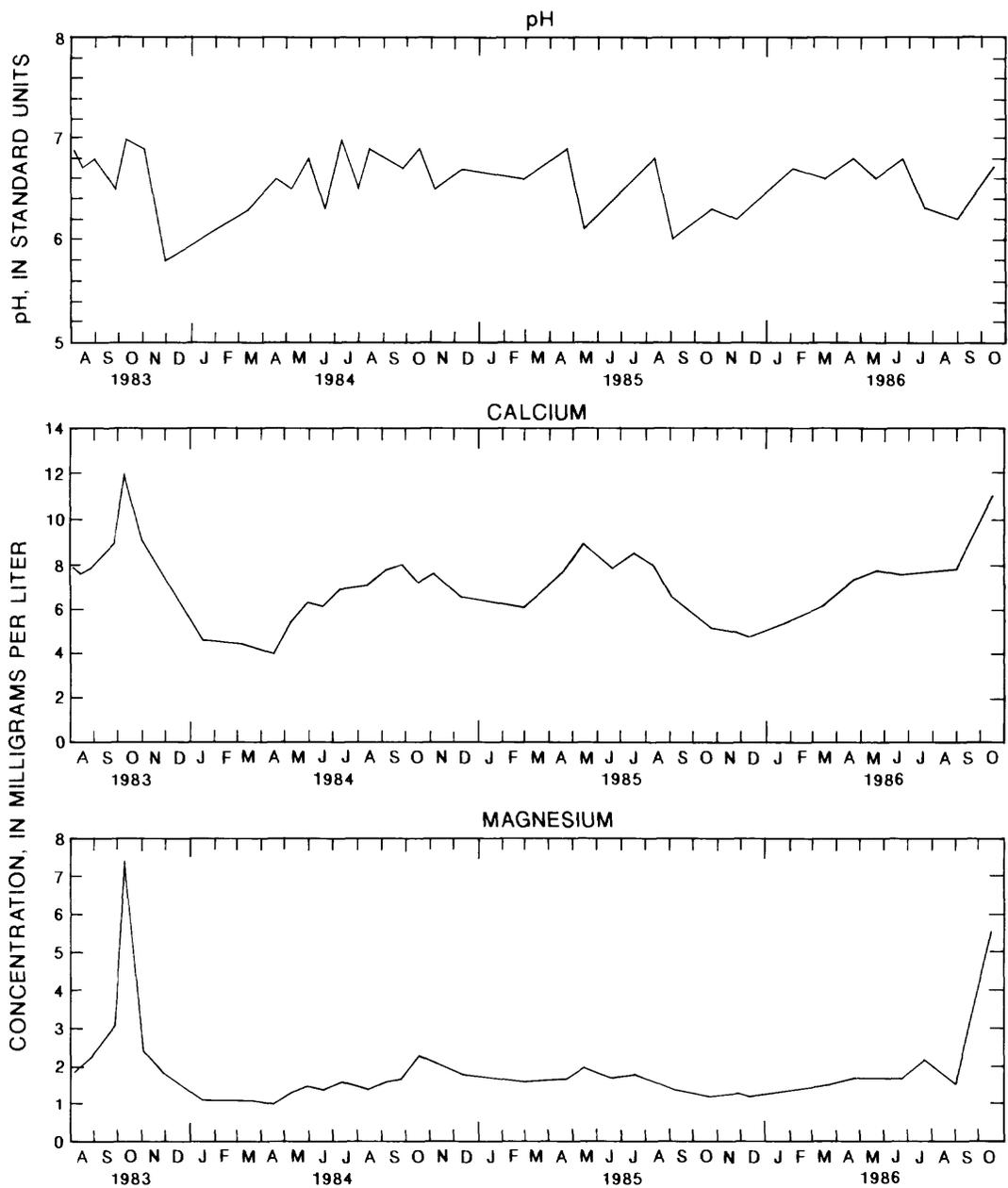


Figure 10.--pH and major dissolved-cation concentrations in water from Chickahominy Reservoir intake, 1983-86.

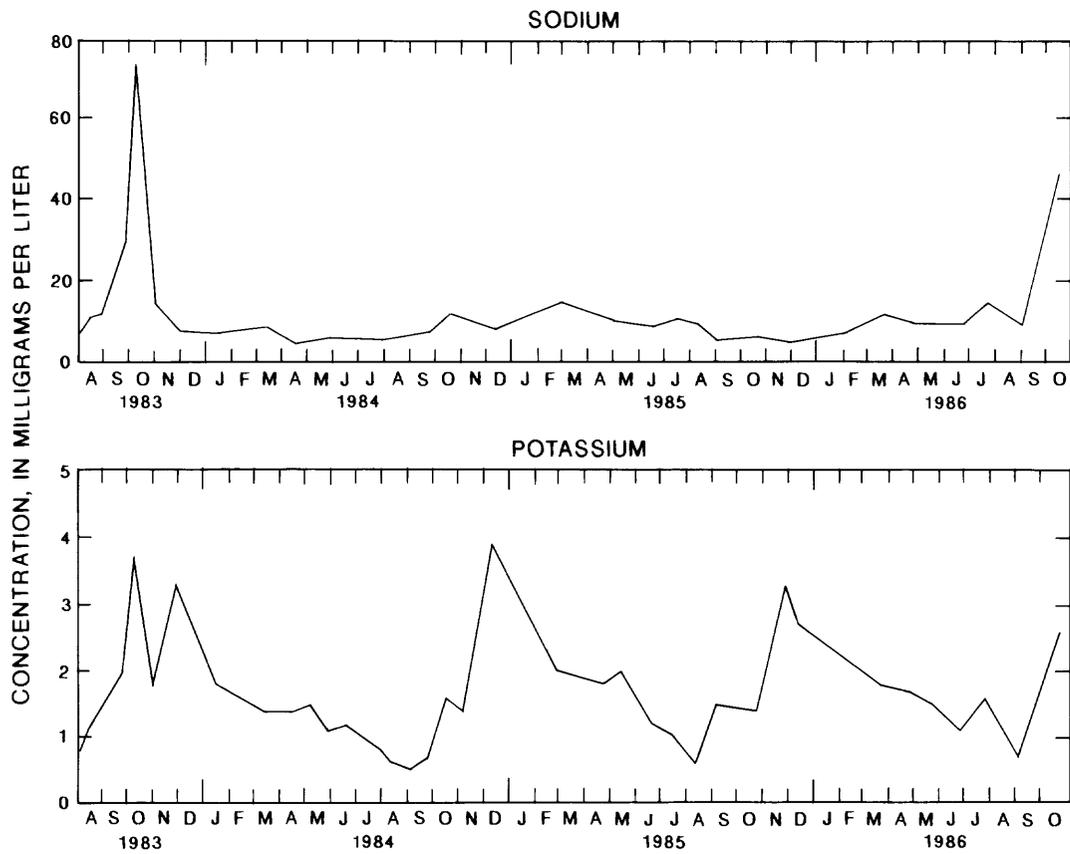


Figure 10.--pH and major dissolved-cation concentrations in water from Chickahominy Reservoir intake, 1983-86.--Continued

Table 16.--Physical and biological measurements of Chickahominy Reservoir (02042720), 1983-86

[--, no data available; <, less than]

Date	Mean flow at Providence Forge, in cubic feet per second	Specific conductance, in microsiemens per centimeter	Water temperature, in degrees Celsius	Color, in platinum-cobalt units	Dissolved oxygen, in milligrams per liter	Dissolved oxygen, in percent of saturation	Coliform, fecal, in colonies per 100 milliliters	Streptococci, fecal, in colonies per 100 milliliters
08-04-83	8.2	93	28.5	--	8.1	--	--	--
08-16-83	3.2	100	24.5	--	5.8	69	5	77
08-31-83	4.1	125	27.0	--	3.5	44	8	94
09-28-83	8.9	211	20.0	--	8.5	94	<1	2,000
10-11-83	9.1	500	19.0	--	7.3	79	5	1,800
11-03-83	113	140	14.0	--	9.2	90	<1	370
11-30-83	543	112	10.0	--	8.1	71	170	350
01-18-84	920	83	3.0	31	11.9	88	7	360
03-14-84	713	80	6.0	40	8.3	66	<1	<1
04-18-84	1,880	58	16.5	75	6.8	71	96	67
05-08-84	352	72	20.0	90	5.7	65	13	80
05-29-84	204	85	23.0	75	4.5	54	26	71
06-19-84	92	72	27.0	70	1.6	20	--	--
07-10-84	75	77	25.5	40	4.8	59	2	11
08-01-84	112	80	24.5	7	3.5	42	8	350
08-14-84	163	80	28.0	18	5.5	71	130	67
09-04-84	34	90	27.0	12	3.6	46	10	58
09-26-84	18	93	23.0	15	4.2	49	2	35
10-17-84	11	125	18.0	15	6.6	70	8	40
11-06-84	40	120	17.0	11	11.7	122	20	110
12-11-84	260	112	5.0	20	10.3	81	23	48
02-28-85	220	150	10.0	10	9.8	86	--	--
04-23-85	59	115	22.0	30	7.5	86	--	--
05-14-85	13	110	23.5	30	4.3	51	--	--
06-20-85	154	100	26.0	40	4.6	57	--	--
07-17-85	8.2	120	27.5	35	1.8	23	--	--
08-12-85	72	100	25.0	40	1.4	17	--	--
09-04-85	203	74	25.0	100	2	2	--	--
10-23-85	433	72	17.5	50	6.5	68	--	--
11-26-85	1,430	70	10.0	100	7.4	66	--	--
12-12-85	329	75	8.5	60	10.6	92	--	--
02-04-86	426	83	4.0	50	11.8	90	--	--
03-19-86	417	118	14.5	40	9.7	97	--	--
04-22-86	181	106	16.0	--	8.4	86	--	--
05-21-86	181	110	20.0	30	4.5	50	--	--
06-24-86	5.3	105	24.0	40	5.6	67	--	--
07-22-86	5.4	145	28.0	20	4.5	58	--	--
09-02-86	124	103	21.5	60	6.4	73	--	--
10-17-86	21	350	17.0	45	8.0	83	--	--
Mean	252	118	19.0	39	6.5	66	28	320
Maximum	1,880	500	28.5	100	11.9	122	170	2,000
Upper quartile	294	119	25.0	50	8.4	86	22	350
Median	113	100	20.0	40	6.5	68	8	77
Lower quartile	16	80	15.0	19	4.5	51	4	53
¹ Drinking-water regulations	--	--	--	15(a)	--	--	<1 (b)	--

¹ Drinking-water regulations are as follows:

- (a) Secondary maximum contaminant level (U.S. Environmental Protection Agency 1986b).
- (b) Maximum contaminant level (U.S. Environmental Protection Agency, 1986a).

water temperatures, and (3) the solubility of oxygen increases as water temperature decreases. In contrast, during the warm months (April through September), DO concentrations at sampling site 02042720 were relatively low, ranging from 0.2 to 8.5 mg/L and averaging 4.8 mg/L. Percentage of saturation of DO averaged 55 percent during these months. These lower concentrations result from greater biological demand for oxygen and lower solubility of oxygen in warm water. On four occasions, measured concentrations of DO were less than 2 mg/L (table 16). These low concentrations probably resulted from withdrawal of water from an oxygen-depleted hypolimnion. The raw-water intake, which is about 12 ft below the water surface of Chickahominy Reservoir, could be in the hypolimnion during warm, calm, base-flow periods.

Major Inorganic Constituents

Major inorganic constituent concentrations in water from Chickahominy Reservoir are summarized in table 15. Although not discussed in detail, all of the major inorganic constituents are included here for completeness. These constituents include major cations (calcium, magnesium, sodium, and potassium), major anions (sulfate, chloride, and fluoride), alkalinity, and silica. Selected trace constituents (including iron, manganese, cadmium, copper, lead, mercury, and zinc), nutrients (phosphorus and nitrogen), chlorophyll-*a*, and total organic carbon are discussed in subsequent sections.

As discussed earlier, the major processes or factors affecting specific conductance in Chickahominy Reservoir include flow reversals at Walkers Dam, runoff of road salt, natural weathering of rocks and soils in the basin, and flow conditions. These same processes or factors largely affect the concentrations of major inorganic constituents in the reservoir. For example, the highest concentrations of dissolved calcium, magnesium, sodium, sulfate, and chloride measured in Chickahominy Reservoir (table 15) occurred after large flow reversals of salty water in September and October 1983 and October 1986 (figs. 10 and 11). This is expected because these are the five major constituents in seawater (Hem, 1985).

The small increases in dissolved-sodium and chloride concentrations during several winter months (figs. 10 and 11) result from runoff of road salt. Because of the time span between sample collections, particularly in the winter months, the effect from each application of road salt is not visible in

these figures; however, results of sample collections in late February 1985 and mid-March 1986 show some residual effects of road-salt application. Typically, water in Chickahominy Reservoir changes from a calcium sodium bicarbonate chloride type, to a predominantly sodium chloride type several weeks after a major application of road salt in the drainage basin.

The season of the year also affects concentrations of major constituents in Chickahominy Reservoir. Dissolved-calcium and alkalinity concentrations (figs. 10 and 11) follow similar seasonal fluctuations that are related to temperature, rates of biological activity, and flow conditions in the basin. Warm, base-flow periods, typical of the summer months, yield water with the highest concentrations of dissolved calcium and alkalinity because (1) stream water is not diluted by surface runoff; (2) the long mean residence time of water in the basin during base flow increases rock-water reaction time; (3) warm temperatures increase the rates of carbonic acid weathering reactions that yield these ions; and (4) warm temperatures also increase rates of decomposition in the soil, which increase the partial pressure of carbon dioxide in the soil and the concentration of carbonic acid in ground water. Carbonic acid is the principal reactant, yielding calcium and bicarbonate (alkalinity) during weathering reactions. In contrast, cool periods of median-to-high flows, typical of the winter months, yield water with the lowest concentrations of these ions. Concentrations are lower because of dilution by surface runoff, shorter rock-water reaction times, and slower rates of chemical weathering because of cooler temperatures and a lower partial pressure of carbon dioxide in soil.

Seasonal fluctuations of dissolved-magnesium concentrations are similar to fluctuations of calcium and alkalinity concentrations. The source of magnesium in Chickahominy Reservoir is primarily carbonic acid weathering of rocks and soils in the basin. Concentrations of dissolved magnesium tend to be highest in the late summer and lowest in the winter (fig. 10); concentrations averaged 1.9 mg/L during the study. The seasonal pattern for dissolved magnesium is partly obscured by periodic inflows of salty water.

Potassium in Chickahominy Reservoir comes primarily from mineral weathering of potassium-bearing rocks in the basin; however, the variability of observed concentrations in water are largely the

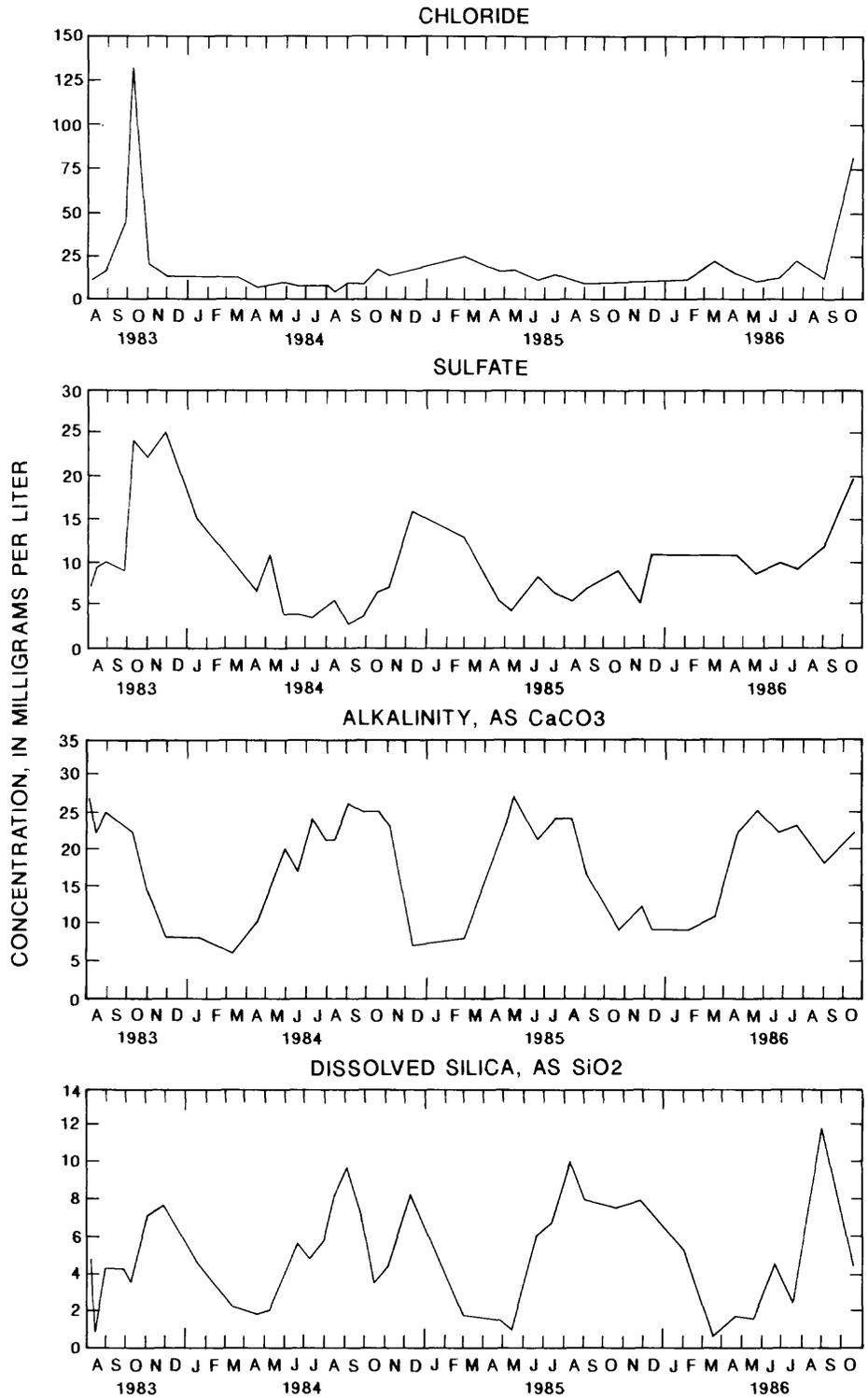
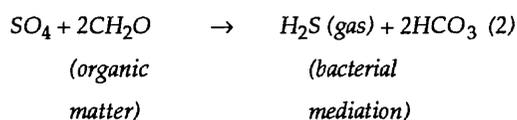


Figure 11.--Concentrations of major dissolved-anion and silica in water from Chickahominy Reservoir intake, 1983-86.

result of potassium cycling by terrestrial plants. Concentrations are elevated in the fall and winter as potassium in decomposing leaf litter reaches the streams. Lower dissolved-potassium concentrations in the spring and summer probably reflect a depleted leaf-litter source and uptake of potassium from soil and ground water by terrestrial plants during the growing season.

Dissolved-sulfate concentrations in water from Chickahominy Reservoir are inversely related to concentrations of alkalinity and, to a lesser extent, concentrations of dissolved calcium and magnesium. Dissolved-sulfate concentrations are lowest in the spring and summer months because biologically mediated sulfate-reduction reactions dominate the sulfur cycle at this time. Concentrations decrease in soils, ground water, and surface water as sulfate is taken up by vegetation, biologically reduced to organic-sulfur forms, and used in tissue, principally leaf matter (Swank and Fitzgerald, 1984). Decreases also occur as sulfate is biologically reduced during the decomposition of organic matter in anaerobic environments, such as in reservoir bottom sediments and wetland soils. This reaction results in a loss of sulfur (as hydrogen sulfide gas) from the drainage basin, as shown in the following reaction (Herlihy and others, 1987):



Both of these biological reactions consume hydrogen ions, thereby generating alkalinity. The reactions partly explain the inverse relation between dissolved-sulfate and alkalinity concentrations in water from Chickahominy Reservoir during the spring and summer months. For example, 2 moles of alkalinity are generated for each mole of sulfate lost as hydrogen sulfide gas in reaction 2.

Dissolved-sulfate concentrations in water from Chickahominy Reservoir are relatively high in the fall and winter months. Sulfate is leached from leaf litter as the organic-sulfur forms are oxidized. Oxidation of organic sulfur also yields hydrogen ions and thereby reduces alkalinity of soils and runoff. For each mole of organic sulfur oxidized in leaf litter, 1 mole of hydrogen ions is generated. This reaction partly explains the same inverse relation between dissolved-sulfate and alkalinity concentrations in water from Chickahominy Reservoir during the fall and winter months (fig. 11).

Concentrations of dissolved silica in water from Chickahominy Reservoir follow a seasonal pattern similar to those of alkalinity and dissolved calcium and magnesium. Maximum dissolved-silica concentrations occur in the summer and fall (fig. 11), corresponding to warm temperatures and low flows. These conditions favor high dissolved-silica concentrations because (1) carbonic-acid weathering of silicates and aluminosilicates in the basin increases with temperature (Hem, 1985), (2) dissolution of diatom frustules (cell walls) buried in reservoir sediments increases with temperature, and (3) average reaction time between reservoir water and basin rocks and soils is greatest during periods of low flow. Maximum dissolved-silica concentrations in water from Chickahominy Reservoir for 1983-86 ranged from 7.7 to 12 mg/L.

Concentrations of dissolved silica tend to be lowest in the spring, averaging about 2 mg/L, because of dilution by surface runoff and slow rates of carbonic acid weathering of silicon-bearing minerals in the basin. Superimposed on this seasonal pattern, however, is the biological uptake of silica by diatoms in Chickahominy Reservoir. During periods of very low flow in the summer and fall months (and perhaps to a lesser extent in the spring), uptake of dissolved silica by planktonic and attached diatoms in Chickahominy Reservoir results in periodically low concentrations. The low concentrations observed in August 1983 (0.9 mg/L), October 1984 (3.5 mg/L), and July 1986 (2.5 mg/L), are at least partly the result of diatom uptake. Significant decreases associated with diatom uptake occur during periods of low flow when external sources of dissolved silica are negligible and diatom populations flourish.

Iron, Manganese, and Selected Trace Elements

"Trace element" is a term used to describe many constituents typically found in trace quantities in water. The discussion of trace elements in this section is limited to iron, manganese, cadmium, copper, lead, mercury, and zinc. Median concentrations of cadmium, copper, lead, mercury, and zinc in water from the raw-water intake of Chickahominy Reservoir were typically less than one-tenth the maximum contaminant levels (MCL's) and secondary maximum contaminant levels (SMCL's) for drinking water established by the U.S. Environmental Protection Agency (USEPA) (1986a, b) (table 17). Moreover, the maximum concentrations of these five metals were less than the USEPA MCL's and

Table 17.--Concentrations of iron, manganese, and selected trace elements in water from Chickahominy Reservoir (02042720), 1983-86

[µg/L, micrograms per liter; mg/L, milligrams per liter; --, no data available; <, less than]

Date	Cadmium, dissolved, in µg/L	Copper, dissolved, in µg/L	Iron, total, mg/L	Iron, dissolved, in mg/L	Lead, dissolved, in µg/L	Manganese, total, in mg/L	Manganese, dissolved, in mg/L	Mercury, dissolved, in µg/L	Zinc, dissolved, in µg/L
08-04-83	<1	60	--	0.12	2	--	0.003	<0.1	250
08-16-83	<1	<10	--	.10	5	--	.021	<.1	<3
08-31-83	<1	<10	--	.12	1	--	.11	<.1	<3
09-28-83	1	7	--	.09	1	--	.013	<.1	<3
10-11-83	<1	<1	--	.06	1	--	.082	.1	<3
11-03-83	<1	2	--	20	2	--	.061	<.1	9
11-30-83	<1	2	--	23	4	--	.24	<.1	34
01-18-84	1	6	--	21	2	--	.064	.3	19
03-14-84	<1	<1	--	.24	<1	--	.035	<.1	18
04-18-84	1	1	--	.52	1	--	.043	<.1	14
05-08-84	<1	2	--	.95	3	--	.038	.2	9
05-29-84	2	1	--	.97	2	--	.087	<.1	8
06-19-84	2	<1	--	.77	4	--	.19	.1	13
07-10-84	<1	<1	--	.28	1	--	.022	.1	5
08-01-84	<1	<1	--	.24	<1	--	.027	.2	<3
08-14-84	<1	1	--	.32	5	--	.019	.3	4
09-04-84	<1	<1	--	.30	6	--	.14	<.1	4
09-26-84	1	1	--	.11	2	--	.032	<.1	<3
10-17-84	<1	<1	--	.04	3	--	.001	<.1	<3
11-06-84	<1	<1	--	.08	1	--	.006	.3	11
12-11-84	<1	<1	--	.24	2	--	.054	.8	17
02-28-85	<1	1	.47	.14	<1	.05	.033	.2	15
04-23-85	<1	1	.79	.35	1	.07	.008	.1	10
05-14-85	<1	1	.61	.14	3	.15	.026	.2	24
06-20-85	<1	1	.82	.36	2	.29	.21	<.1	<3
07-17-85	<1	2	1.1	.56	2	.70	.69	<.1	<3
08-12-85	<1	2	1.4	.45	<1	.23	.078	.1	6
09-04-85	<1	2	2.6	.86	<1	.41	.34	.1	7
10-23-85	2	1	1.0	.41	<1	.05	.033	.1	11
11-26-85	<1	1	2.5	.83	6	.05	.038	<.1	21
12-12-85	<1	1	1.4	.60	<1	.04	.030	<.1	8
02-04-86	<1	<1	.77	.28	2	.04	.040	<.1	13
03-19-86	<1	<1	.71	.26	2	.07	.044	<.1	8
04-22-86	1	4	1.4	.50	<1	.05	.008	<.1	16
05-21-86	<1	2	.80	.21	1	.13	.027	<.1	13
06-24-86	<1	2	1.6	.25	<5	.31	.036	<.1	25
07-22-86	<1	3	1.4	.69	<5	.53	.48	.1	5
09-02-86	<1	<1	1.5	.73	<5	.11	.14	.1	4
10-17-86	<1	1	.32	.05	<5	.09	.003	<.1	14
Mean	<1	3	1.18	.36	2	.19	.091	.1	16
Maximum	2	60	2.60	.97	6	.70	.69	.8	250
Upper quartile	<1	2	1.40	.51	2.5	.29	.084	.1	14
Median	<1	1	1.05	.26	2	.10	.038	<.1	9
Lower quartile	<1	<1	.77	.14	1	.05	.024	<.1	4
¹ Drinking-water regulations	10(a)	1,000(b)	--	.30(b)	50(a)	--	.050(b)	2(a)	5,000(b)

¹ Drinking-water regulations are as follows:

(a) Maximum contaminant level (U.S. Environmental Protection Agency, 1986a).

(b) Secondary maximum contaminant level (U.S. Environmental Protection Agency, 1986b).

SMCL's. In contrast, concentrations of iron and manganese commonly exceeded the USEPA SMCL's (0.3 and 0.05 mg/L, respectively) (table 17).

The median concentration of dissolved iron was 0.26 mg/L, and concentrations as high as 0.97 mg/L were measured. Concentrations equaled or exceeded the 0.30 mg/L SMCL in 44 percent of the analyses. The median concentration of dissolved manganese was 0.038 mg/L, and concentrations as high as 0.7 mg/L were measured. Concentrations equaled or exceeded the 0.05 mg/L SMCL in 38 percent of the analyses. Although iron and manganese in water do not pose a human health risk, these elevated concentrations are troublesome for water suppliers. Raw water with elevated concentrations must be treated to minimize taste and odor problems in finished water and to reduce staining of plumbing fixtures and laundered clothes.

About one-third of the iron in water from Chickahominy Reservoir is dissolved, and two-thirds is in particulate form. Total-iron concentrations in excess of 2.0 mg/L were measured on two different occasions (fig. 12). About two-thirds of the manganese in water from Chickahominy Reservoir is dissolved, and one-third is in a particulate form. Total manganese concentrations in excess of 0.5 mg/L were measured (fig. 12).

The distinction between dissolved and total iron at the Chickahominy Reservoir raw-water intake could be arbitrary. Black and Christman (1963) report that colored material in water is almost exclusively in a particulate form, made up of single or polymerized organic molecules. Depending on the size of these polymers, some of the colored material passes through a 0.45- μ m filter (operationally defined as dissolved), and some is retained on the filter (operationally defined as particulate). Assuming that iron is complexed with organic molecules, the distinction between dissolved and particulate iron could simply be a function of the size of the complexing organic molecule. In Chickahominy Reservoir, about one-third of the organic iron complexes pass through a 0.45- μ m filter; two-thirds of the complexes are retained on the filter. Irrespective of the total-iron concentration, the percentage of iron passing through a 0.45- μ m filter remained constant during the study, as illustrated in figure 12.

Water-quality samples, collected by the U.S. Geological Survey from 1970 through 1986 from Chickahominy River near Providence Forge, indi-

cate that most of the dissolved iron at the raw-water intake comes from external sources in the drainage basin. (As discussed earlier, the Providence Forge station is 11 mi upstream from the raw-water intake and it gages streamflow from 83 percent of the reservoir's drainage basin.) Other dissolved-iron sources, such as internal loading of ferrous iron from anaerobic reservoir sediments, do not appear to be as large. The mean concentration of dissolved iron in water from the Providence Forge station is 0.49 mg/L (table 18), which is similar to the mean concentration of 0.36 mg/L at the raw-water intake for 1983-86 (table 17). The seasonal variability of dissolved iron at both sites also is similar; elevated concentrations typically occur between April and September.

Internal sources of iron periodically contribute to the elevated dissolved-iron concentrations at the raw-water intake during periods of thermal stratification. Ferrous ions released from anaerobic bottom sediments could result in elevated dissolved-iron concentrations in the hypolimnion of Chickahominy Reservoir. The solubility of iron (and manganese as well) is much greater in reducing environments than in oxidizing environments. Dissolved iron in the hypolimnion can reach the raw-water intake through several processes. During reservoir overturn, strong winds, or periods of high flow, part or all of the hypolimnion is mixed with the epilimnion, thereby increasing iron concentrations at the raw-water intake. In addition, the raw-water intake could be directly entraining anaerobic hypolimnetic water as the thermocline level rises during warm, calm weather. Direct entrainment of anaerobic hypolimnetic water probably occurred on September 4, 1985, as evidenced not only by a DO concentration of 0.2 mg/L at the raw-water intake (table 16) but also by some of the highest concentrations of total iron (2.6 mg/L) and dissolved iron (0.86 mg/L) measured during the study (table 17).

Concentrations of dissolved iron in the oxygenated water of Chickahominy Reservoir are at least two orders of magnitude greater than that predicted from inorganic equilibrium considerations. In the absence of iron-complex formation, Stumm and Morgan (1981) estimate that dissolved-iron concentrations would not exceed 0.001 mg/L in water with a pH range of 6.0 to 7.0, which is typical of Chickahominy Reservoir (table 18). Concentrations greater than expected probably result from the formation of organic-iron complexes with humic and fulvic substances in the water, which would prevent the pre-

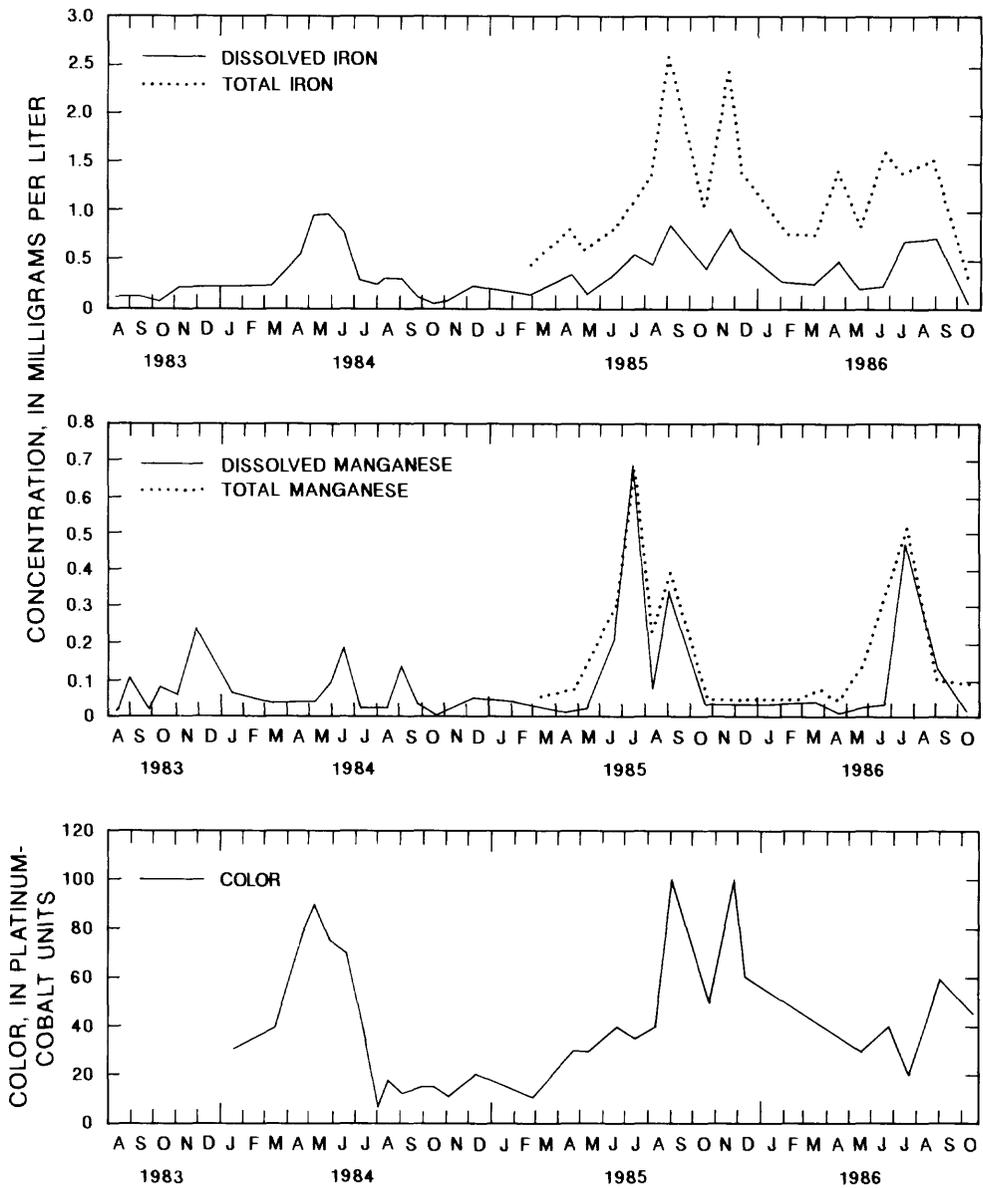


Figure 12.--Concentrations of iron and manganese and color intensity of water from Chickahominy Reservoir intake, 1983-86.

cipitation of iron (Sholkovitz and Copland, 1982). Humic and fulvic substances are refractory, colored, polyelectrolytic organic acids produced during the microbiological degradation of plant material (Thurman, 1985). Humic and fulvic substances probably constitute most of total organic carbon measured in water from Chickahominy Reservoir, which averaged 9.5 mg/L (table 19). The presence of these substances in Chickahominy Reservoir imparts a brownish-yellow color to the water. These substances originate in the Chickahominy River drainage basin, as evidenced by similar average color intensities at the Providence Forge station (45 platinum-cobalt units; table 18) and the Chickahominy Reservoir raw-water intake (39 platinum-cobalt units; table 16).

Table 18.--Mean monthly dissolved-iron concentrations and color intensity of water from Chickahominy River near Providence Forge (02042500), 1970-86

[Pt-Co, Platinum-Cobalt units]			
Month	Dissolved iron, in milligrams per liter	Color intensity, in Pt-Co units	Number of observations
January	0.28	34	11
February	.19	24	15
March	.30	41	11
April	.77	65	14
May	.90	75	11
June	.83	67	11
July	.54	41	10
August	.62	47	9
September	.36	29	9
October	.40	36	13
November	.39	34	12
December	.35	44	12
Mean	.49	45	--

As noted by Stumm and Morgan (1981), colored waters (that is, water with high concentrations of humic and fulvic substances) commonly contain elevated concentrations of dissolved iron. This relation holds for Chickahominy Reservoir. A strong positive correlation ($r^2 = 0.63$) between dissolved-

iron concentration and color intensity (fig. 12) probably indicates the presence of organic-iron complexes in Chickahominy Reservoir.

The seasonal variability of iron in Chickahominy River and Chickahominy Reservoir is probably related to the seasonal variability of humic and fulvic substances leached from the Chickahominy River basin. The increased color intensity of Chickahominy River near Providence Forge, which typically begins in April and continues into August (table 18), is probably related to increased rates of microbial decomposition during these warm months. The wetlands along much of the length of Chickahominy River are probably a major source of humic and fulvic substances coloring the water (Hem, 1985).

Flow conditions appear to be an important factor affecting iron concentrations and color intensity in Chickahominy Reservoir as well. The first major rain after the weather begins to warm (typically after March) yields runoff with the highest annual color intensity and concentration of iron. Apparently, the first major rain leaches humic and fulvic substances, along with complexed iron, that accumulate in the basin during the fall and winter months. Subsequent rainfall yields runoff with less color intensity as the amount of leachable humic and fulvic substances in the basin decreases. Typically, color intensity in Chickahominy Reservoir peaks in April or May when high flows and warm temperatures occur together; however, the timing of the peak color intensity in Chickahominy River and Chickahominy Reservoir can occur much later during dry years. In 1985, the peak color intensity and highest iron concentrations in Chickahominy Reservoir did not occur until late summer (fig. 12) because of extremely dry conditions in the spring and summer. From April through mid-August 1985, flows averaged 73 ft³/s in the Chickahominy River near Providence Forge, or about 32 percent of the long-term average for these months. Moreover, not a single storm during this period produced flows in excess of the median annual flow. Consequently, the heavy rainfall in mid-August 1985 provided the first major leaching and transport of organic matter in the basin since the previous winter. Similarly, dry conditions in the spring and summer of 1986 delayed the color intensity and dissolved-iron peaks in Chickahominy Reservoir until late summer.

The color intensity of water is an important factor in the study of Chickahominy Reservoir because it provides an indirect measurement of the concentration of humic and fulvic substances in the water. Beyond acting as complexers for keeping iron in solution or suspension, humic and fulvic substances present another problem to the city of Newport News. These substances react with chlorine added during the disinfection process, producing trihalomethanes (THM), primarily chloroform, in finished water (Rook, 1974). The MCL for THM in drinking water is 0.1 mg/L (U.S. Environmental Protection Agency, 1986a). This MCL applies only to community water systems serving more than 10,000 individuals. Meeting this MCL has periodically presented a problem for the city of Newport News, presumably because of elevated levels of THM precursors (humic and fulvic substances and perhaps other organic matter) in the raw-water supplies.

Reducing or eliminating withdrawals from Chickahominy Reservoir when color intensity is high is one option that could be used to alleviate some problems associated with treating water containing high concentrations of THM precursors and iron. Color intensity, which can be monitored easily onsite (without the delay or expense of a laboratory analysis), would be a type of real-time data used to adjust raw-water withdrawals. For example, the Chickahominy Reservoir pump station would be idle about 2 months per year if a practice was instituted of not withdrawing water with a color intensity exceeding 60 Pt-Co units. Such a practice would not create a raw-water shortage for the city of Newport News under most circumstances because (1) other water sources with lower color intensity are available, and (2) color intensity tends to be highest in Chickahominy Reservoir during high-flow periods when supplies in all reservoirs are plentiful.

Manganese and iron are redox-sensitive elements, and their concentrations in reservoirs often follow similar patterns. As shown in figure 12, however, peak manganese concentrations in water from the raw-water intake of Chickahominy Reservoir do not necessarily correspond to peak iron concentrations. Statistically, concentrations of dissolved iron and manganese are not significantly correlated ($p > 0.05$). The lack of correlation is attributable to the different sources of the two elements in the reservoir. Most of the iron in water at the raw-water intake comes from the drainage basin, complexed with humic and fulvic substances. In contrast, man-

ganese does not form complexes with humic and fulvic substances (Stumm and Morgan, 1981; Hem, 1985), and thus it is not transported to Chickahominy Reservoir as readily as iron. Reservoir bottom sediments are the primary source of manganese responsible for the periodically elevated concentrations in water at the raw-water intake (fig. 12). As anaerobic conditions develop in Chickahominy Reservoir after thermal stratification, soluble manganese ions are released from bottom sediments and accumulate in the hypolimnion. Manganese in the hypolimnion can reach the raw-water intake through a number of pathways. Hypolimnetic water is withdrawn directly when the thermocline level rises above the raw-water intake during warm, calm weather. Elevated manganese concentrations could also reach the intake when hypolimnetic water mixes with the upper waters during periods of reservoir overturn, strong wind, and high flow. In addition, diffusion of manganese from the hypolimnion to the upper waters could also occur, but it is questionable whether this pathway contributes significantly to the peak concentrations shown in figure 12.

Direct withdrawal of anaerobic hypolimnetic waters probably accounts for most of the elevated manganese concentrations analyzed in water from the raw-water intake. Six of the seven largest dissolved-manganese concentrations shown in figure 12 were associated with DO concentrations of 4.5 mg/L or less. These low DO concentrations indicate that part or all of the 5-ft long intake screen could have been in the hypolimnion. Direct withdrawal of hypolimnetic water undoubtedly occurred in the summer of 1985 (July, August, and September), as evidenced by DO concentrations of less than 2 mg/L in water from the raw-water intake (table 16), and the highest concentrations of total manganese (0.70 mg/L) and dissolved manganese (0.69 mg/L) analyzed during the study, in July 1985 (table 17). The dissolved-manganese concentration in late November 1983, which was not associated with low DO concentrations, probably reflects the recent overturn of the reservoir or mixing of hypolimnetic water into the epilimnion as a result of high flows (fig. 12).

Unlike iron, reduced manganese from an anaerobic hypolimnion can remain soluble for days or weeks when exposed to DO in the epilimnion (Stumm and Morgan, 1981). The rate of manganese oxidation is pH-dependent, being slower at lower pH values. This slow rate of oxidation explains why

the peak manganese concentrations in water from Chickahominy Reservoir are dominated by dissolved (reduced) species (fig. 12).

Nutrients and Chlorophyll-a

Nutrient concentrations observed in water from Chickahominy Reservoir are listed in table 19, and their total concentrations are shown in figure 13. Total-nitrogen concentrations in water from Chickahominy Reservoir ranged from 0.4 to 1.7 mg/L and averaged 0.8 mg/L (table 19). Total-nitrogen concentrations are obtained by summing concentrations of total ammonia plus organic nitrogen and dissolved nitrite plus nitrate nitrogen. Organic nitrogen comprises about 90 percent of the total nitrogen, with the remainder present as dissolved ammonia, and nitrite plus nitrate (table 19). Dissolved-ammonia concentrations averaged 0.04 mg/L (as nitrogen), and the concentration of nitrite plus nitrate averaged less than 0.10 mg/L (as nitrogen).

Total-phosphorus concentrations ranged from 0.02 to 0.10 mg/L and averaged 0.049 mg/L in water from Chickahominy Reservoir. Forty-five percent (0.022 mg/L) of the total phosphorus is present in various dissolved forms; 20 percent (0.01 mg/L) of the total phosphorus is present as dissolved orthophosphorus (table 19). On the basis of total-phosphorus concentration, Chickahominy Reservoir lies in the middle-to-upper eutrophic range, as defined by Carlson (1977). A Carlson Trophic State Index of 60 for this reservoir indicates that nutrients (particularly phosphorus) are in adequate supply for algae to reach nuisance levels (Moore and Thornton, 1988). High concentrations of algae in raw water can cause treatment problems by clogging filters, contributing THM precursors, and imparting objectionable tastes and odors to finished water.

Mean annual nutrient concentrations for 1983-86 ranged from 0.7 to 0.9 mg/L for total nitrogen, and 0.042 to 0.056 mg/L for total phosphorus (table 20). Concentrations from June through September did not differ appreciably from yearly averages. These narrow concentration ranges indicate that the wetlands in the basin can provide either a physical and (or) chemical "buffering" mechanism for nutrients. During high flows, nutrient concentrations are undoubtedly reduced as particulate matter in runoff settles in low-velocity wetland areas. Wetlands can also affect nutrient concentra-

tions through processes of biological uptake, sorption, and chemical exchange as water moves through and over wetland sediments (Martin, 1988).

The ratio (by weight) of total nitrogen to total phosphorus (N:P) provides an indication of which nutrient potentially limits algal growth in an aquatic system. When the N:P ratio exceeds 7, algae are potentially phosphorus limited; when the ratio decreases to less than 7, algae are potentially nitrogen limited. Nitrogen limitation rarely occurs in freshwater, however. As supplies of ammonia, nitrate, and nitrite are depleted, blue-green algae capable of "fixing" atmospheric nitrogen become dominant. The influx of atmospheric-derived nitrogen prevents the N:P ratio from becoming much less than 7. Consequently, phosphorus-enriched reservoirs in which the N:P ratio is small tend to support a large biomass of nitrogen-fixing blue-green algae. Blue-green algae cause the most severe algal problems during water treatment. Therefore, it is useful for water suppliers to track the N:P ratio of their raw-water supply because it could signal a shift in the algal population from preferred species to nuisance blue-green algae. If potential problems are identified early, water suppliers can adjust treatment processes or initiate action to decrease phosphorus loads to the reservoir.

Algae in the Chickahominy Reservoir are potentially phosphorus limited, as evidenced by a mean annual N:P ratio of 16 (table 20). This ratio ranged from 6 in November 1985 to 55 in October 1984, although it typically was between 10 and 30 (fig. 13). A phosphorus limitation indicates that a growing algal population will deplete reserves of phosphorus before depleting reserves of nitrogen. Therefore, the phosphorus concentration in Chickahominy Reservoir sets a theoretical upper limit for algal biomass, which would increase if more phosphorus is added to the system. Moreover, if phosphorus is added without nitrogen, the N:P ratio in Chickahominy Reservoir would decrease, and the algal population could shift to nuisance nitrogen-fixing blue-green algae.

A mean annual N:P ratio of 16 does not necessarily imply that algae in Chickahominy Reservoir are phosphorus limited. Other factors also affect algal growth rate and biomass. In fact, concentrations of total phosphorus and chlorophyll-*a* are not significantly correlated ($p > 0.05$) in Chickahominy Reservoir (fig. 13). Chlorophyll-*a* concentration is used as a surrogate for algal biomass because of its

Table 19.--Concentrations of nutrients, total organic carbon, and chlorophyll-*a* in water from Chickahominy Reservoir (02042720), 1983-86

[Concentrations in milligrams per liter, except for chlorophyll-*a*, which is in micrograms per liter, <, less than; --, no data available]

Date	Nitrite, plus nitrate, dissolved, as N	Ammonia, dissolved, as N	Nitrogen, total, as N	Ammonia plus organic, total, as N	Phosphorus, total, as P	Phosphorus, dissolved, as P	Phosphorus, ortho dissolved, as P	Organic carbon, total, as C	Chlorophyll- <i>a</i>
08-04-83	<.1	.01	--	--	.05	--	<.01	7.8	44
08-16-83	<.1	.01	.6	.6	.05	--	.01	8.5	59
08-31-83	<.1	<.01	.4	.4	.05	--	<.01	9.8	64
09-28-83	<.1	.03	.6	.6	.04	--	<.01	8.0	28
10-11-83	<.1	<.01	.5	.5	.05	--	<.01	8.6	15
11-03-83	<.1	.15	1.2	1.2	--	--	<.01	6.8	23
11-30-83	<.1	<.01	.5	.5	.05	--	<.01	8.5	3.1
01-18-84	<.1	.01	.4	.4	.03	0.02	.02	6.4	1.9
03-14-84	<.1	.02	.6	.6	.02	<.01	<.01	8.1	.6
04-18-84	<.1	.02	1.1	1.1	.05	.01	.01	8.9	2.6
05-08-84	<.1	.05	.8	.8	.06	.03	<.01	10	1
05-29-84	<.1	.05	1.7	1.7	.10	.06	<.01	10	3.3
06-19-84	<.1	.10	1.4	1.4	.07	.03	.02	12	3.8
07-10-84	<.1	<.01	1.1	1.1	.04	.03	<.01	9.9	16
08-01-84	<.1	.09	.4	.4	.03	<.01	<.01	7.6	6
08-14-84	<.1	.03	.9	.9	.05	<.01	.02	8.4	27
09-04-84	<.1	<.01	.6	.6	.02	--	<.01	7.7	15
09-26-84	<.1	.07	.6	.6	.03	.01	<.01	7.5	26
10-17-84	<.1	<.01	1.1	1.1	.02	<.01	<.01	8.4	7.1
11-06-84	<.1	<.01	.7	.7	.04	.03	<.01	7.2	29
12-11-84	<.1	.03	.5	.5	.03	.02	.02	7.4	1.4
02-28-85	--	--	--	.9	.029	.017	--	7.4	2.2
04-23-85	.18	<.01	1.1	.9	.063	.018	.014	8.5	9.5
05-14-85	.02	.05	.5	.5	.037	.011	.009	8.9	13
06-20-85	.03	.03	.6	.6	.049	.030	.008	10	19
07-17-85	.05	.08	.6	.6	.051	.020	.010	12	23
08-12-85	.03	.10	.5	.5	.054	.034	.020	9.2	12
09-04-85	.02	.10	.7	.7	.062	.034	.016	20	6.4
10-23-85	.03	<.01	.7	.7	.054	.024	.004	11	5.6
11-26-85	.02	.07	.6	.6	.103	.049	.035	16	.9
12-12-85	.04	.05	.6	.6	.062	.035	.023	9.9	1
02-04-86	.12	.03	.6	.5	.042	.015	.007	5.9	3.1
03-19-86	<.01	<.01	.5	.5	.036	.013	.003	9.2	7.6
04-22-86	<.01	.02	.7	.7	.055	.026	.013	12	6.2
05-21-86	.01	.06	.8	.8	.049	.017	.009	10	13
06-24-86	<.01	.01	.9	.9	.051	.011	.005	9.6	14
07-22-86	.03	.04	1.0	1.0	.067	.012	.004	10	48
09-02-86	.03	.02	.8	.8	.071	.037	.021	15	16
10-17-86	<.01	<.01	1.0	1.0	.032	<.01	<.001	9.8	29
Mean	<.1	.037	.8	.8	.049	.022	.010	9.5	16
Maximum	.18	.150	1.7	1.7	.103	.060	.035	20	64
Upper quartile	<.1	.050	.9	.9	.055	.030	.014	10	23
Median	<.1	.025	.6	.6	.050	.020	<.01	8.9	12
Lower quartile	<.1	<.01	.5	.5	.036	.011	<.01	7.9	3.2
¹ Drinking-water regulation	10	--	--	--	--	--	--	--	--

¹ Maximum contaminant level (U.S. Environmental Protection Agency, 1986a).

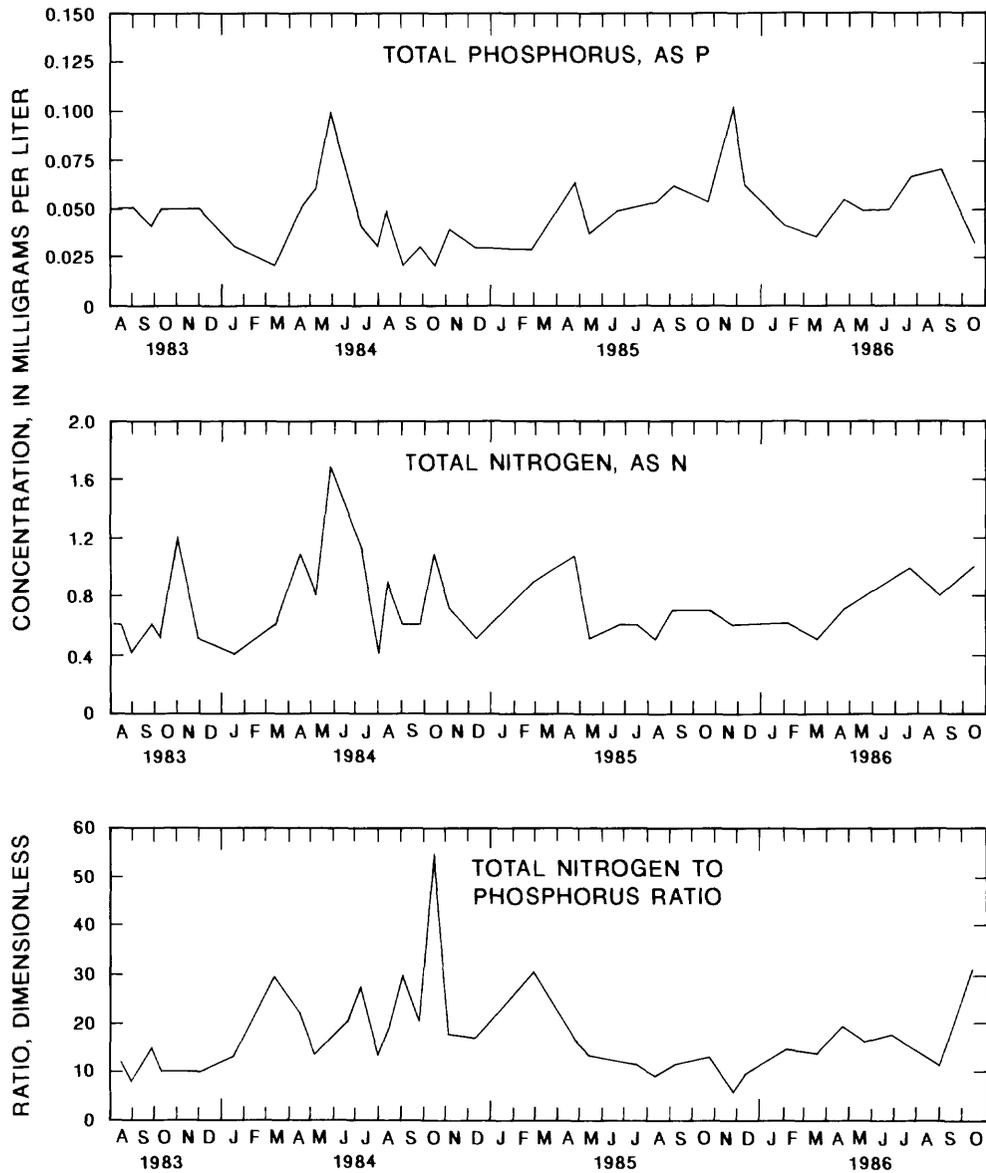


Figure 13.--Concentrations of total nitrogen and total phosphorus, and ratio of total nitrogen to total phosphorus, at the Chickahominy Reservoir intake, 1983-86.

Table 20.—Mean concentrations of nutrients and chlorophyll-*a*, in water from Chickahominy Reservoir (020242720), and mean discharge of Chickahominy River near Providence Forge (02042500), 1983-86

[mg/L, milligrams per liter; µg/L, micrograms per liter; ft³/s, cubic feet per second]

	1983		1984		1985		1986		Mean	
	Annual ¹	Aug-Sept.	Annual	June-Sept.	Annual	June-Sept.	Annual ²	June-Sept.	Annual	June-Sept.
Total nitrogen, mg/L as N	0.7	0.5	0.9	0.9	0.7	0.6	0.8	0.9	0.8	0.8
Total phosphorus, mg/L as P	.048	.048	.042	.040	.056	.054	.050	.063	.049	.048
N/P ratio	14	11	21	22	12	12	16	15	15	16
Chlorophyll- <i>a</i> , µg/L	34	49	10	16	9	15	17	26	16	25
Discharge, ft ³ /s	144	5.8	379	154	269	188	148	52	261	113

¹ Includes August through December, 1983.

² Includes January through October, 1986.

ease of measurement. Chlorophyll-*a* is present in all algae and is commonly in direct proportion to algal biomass (Moore and Thornton, 1988; Ambrose and others, 1988). Total-phosphorus concentrations in water from Chickahominy Reservoir remained relatively constant, whereas chlorophyll-*a* concentrations differed markedly, ranging from 0.6 to 64 µg/L and averaging 16 µg/L (table 19). Total-phosphorus concentration sets the upper limit for chlorophyll-*a* concentration (algal biomass) when conditions for growth are optimal in Chickahominy Reservoir, but other factors—including hydraulic retention time of the reservoir, light intensity, water temperature, and zooplankton grazing rates—account for much of the chlorophyll-*a* variability apparent in figure 14.

Hydraulic retention time is the factor that affects most of the chlorophyll-*a* variability in Chickahominy Reservoir. When hydraulic retention time is short, the rate of algal washout can equal or exceed net algal growth rates and can thereby keep biomass low regardless of nutrient levels. When the hydraulic retention time of the reservoir is long, the rate of algal washout is insignificant. Thus, algal growth and assimilation of nutrients are possible.

Moore and Thornton (1988) indicate that algal biomass is severely limited when the hydraulic retention time of a reservoir is less than 1 to 2 weeks. The hydraulic retention time of Chickahominy Reservoir averages 9 days, indicating that algal biomass (and chlorophyll-*a* concentration) is frequently affected by inflow rates.

Chlorophyll-*a* concentration at the Newport News raw-water intake and flow conditions at the Providence Forge streamflow-gaging station are inversely related in Chickahominy Reservoir, as illustrated in figure 14. High flows between August and October 1985 kept chlorophyll-*a* concentrations low, averaging only 8 µg/L, less than one-third of the summer average. The hydraulic retention time of Chickahominy Reservoir during this period was 7 days. In contrast, during the same months in 1983, chlorophyll-*a* concentration averaged 42 µg/L (table 19). Flows were low during this period, and the hydraulic retention time was 100 days. This difference in chlorophyll-*a* concentrations cannot be explained by differing total-phosphorus concentrations. In fact, total-phosphorus concentrations averaged 0.05 mg/L for August through October in both

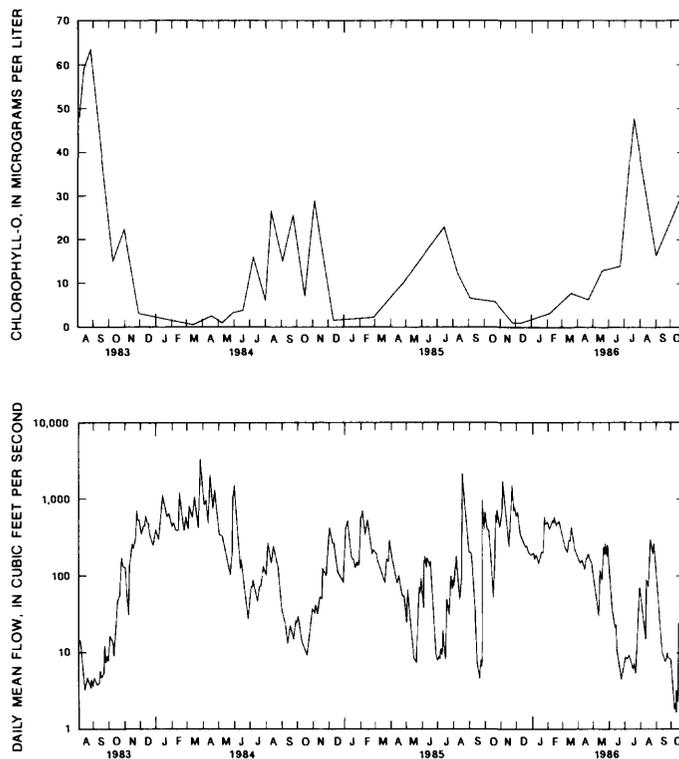


Figure 14.—Relation between chlorophyll-*a* concentration in water from Chickahominy River intake and daily mean discharge of Chickahominy River near Providence Forge, 1983-86.

1983 and 1985. Indeed, most of the rapid decreases in chlorophyll-*a* concentrations shown in figure 14 are preceded by periods of high flow, such as in November 1983 and 1984, and August 1985 and 1986.

A regression model was developed to further explore the physical and chemical processes affecting chlorophyll-*a* concentrations in water from Chickahominy Reservoir. The model terms are (1) reservoir inflow rate, which is related to hydraulic retention time; (2) time of year, which is related to water temperature and light intensity; and (3) total-phosphorus concentration, which is the potentially limiting nutrient for algal growth. The form of the "inflow-rate" term was determined by finding the best fit (largest coefficient of determination) when the natural log of the chlorophyll-*a* concentration, $\ln(\text{CHL})$, is regressed against the following functions of discharge:

Q_{10} and Q_{20} linear;

$\ln(Q_{10})$ and $\ln(Q_{20})$ logarithmic;

$1/Q_{10}$ and $1/Q_{20}$ inverse; and

$1/(1 + a \times Q_{10})$ and $1/(1 + a \times Q_{20})$ hyperbolic;

where:

Q_{10} is mean discharge at Providence Forge gaging station 10 days before sampling;

Q_{20} is mean discharge at Providence Forge gaging station 20 days before sampling; and

a is the hyperbolic coefficient, which was varied from 0.001 to 0.05.

Chlorophyll-*a* concentrations were log-transformed before regression analysis to stabilize the variance of residuals. Instantaneous discharge at the time of sampling was not used in these functions because chlorophyll-*a* concentration is related to the rate of algal washout in a reservoir over a period of days or weeks before sampling.

The following hyperbolic function of discharge at the Providence Forge gaging station best explains the variability in chlorophyll-*a* concentrations:

$$HYPQ_{20} = 1/(1 + 0.004 X Q_{20}). \quad (3)$$

When mean flows are relatively high (greater than 300 ft³/s for 20 days at Providence Forge), this hyperbolic function is low and relatively insensitive to changing flow rates. Thus, the function simulates the effect of hydraulic retention time on the algal population. Algae in Chickahominy Reservoir are washed out in about 8 days at a flow rate of 300 ft³/s. This 8-day period is probably too short for the development of a large algal population. When these flow conditions were met, chlorophyll-*a* concentrations averaged 3 µg/L in Chickahominy Reservoir and never exceeded 6 µg/L. As average flows become less than 300 ft³/s, chlorophyll-*a* concentrations determined from equation 3 increase fairly rapidly, simulating improving flow conditions for algal growth as Chickahominy Reservoir shifts from a riverine to a lake system. Chlorophyll-*a* concentrations averaged 21 µg/L when flows at the Providence Forge gaging station averaged less than 300 ft³/s for 20 days before sampling.

The "time-of-year" terms considered in the regression model are the following sine and cosine functions (in radians) of time:

$$STIME = \text{sine} (DTIME X 2\pi) \quad (4)$$

$$CTIME = \text{cosine} (DTIME X 2\pi), \quad (5)$$

where DTIME is a decimal representation of sampling date. For example, August 4, 1983, is expressed as 1983.589. These time functions approximate seasonal variation of water temperature and light intensity, both of which affect chlorophyll-*a* concentration. Total-phosphorus concentration (TP) is included in the regression model because it is the potentially limiting nutrient. TP is log-transformed before regression analysis.

All independent variables considered in the model except ln(TP) explain a statistically significant ($p < 0.05$) part of the variance in ln(CHL). The resulting regression equation,

$$\ln (CHL) = 0.0370 - 0.3668 (STIME) - 0.5039 (CTIME) + 3.0220 (HYPQ_{20}), \quad (6)$$

has a coefficient of determination (r^2) of 0.79 and a mean-square error of 0.3529. Thus, equation 6 explains 79 percent of the variability in ln(CHL). That HYPQ₂₀ explains 71 percent of the variability emphasizes the importance of hydraulic retention time in controlling algal biomass in Chickahominy Reservoir. Scatter plots of residuals and estimated values of the dependent variable and plots of residuals and independent variables show no patterns or indication of bias.

After exponentiation of equation 6 and correction for the bias that this transformation produces (Ferguson, 1986), the regression model for mean chlorophyll-*a* concentration in Chickahominy Reservoir becomes:

$$CHL = \exp [(0.2134 - 0.3668(STIME) - 0.5039(CTIME) + 3.0220(HYPQ_{20})] \quad (7)$$

A comparison of the model's simulated values to measured values shows how well chlorophyll-*a* concentrations at the raw-water intake can be estimated, given flow conditions at the Providence Forge gaging station and time of year (fig. 15). The magnitude and timing of most chlorophyll-*a* peaks are accurately simulated by the model. In addition, the duration of depressed concentrations of chlorophyll-*a* during median and high flows are accurately simulated.

The regression model given by equation 7 successfully simulates chlorophyll-*a* concentration without including an independent variable for total-phosphorus concentration (the potentially limiting nutrient). This independent variable is unnecessary because the concentration of total phosphorus was relatively constant during the period of simulation, and therefore, does not explain much of the chlorophyll-*a* variability. As discussed earlier, however, total-phosphorus concentration probably sets the upper limit for chlorophyll-*a* concentrations in

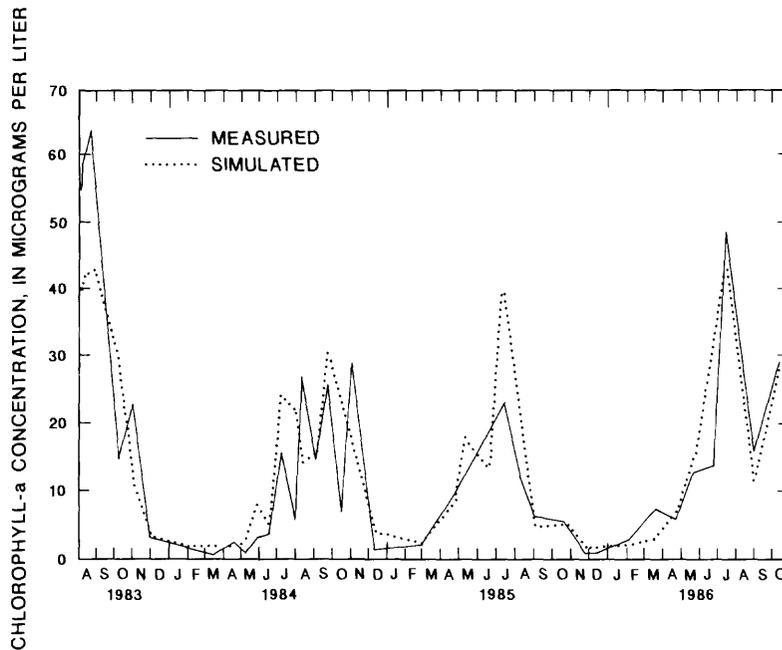


Figure 15.—Measured and simulated chlorophyll-*a* concentrations in water from Chickahominy Reservoir intake, 1983-86.

Chickahominy Reservoir and probably affects algal growth rates as well. The effect of phosphorus on chlorophyll-*a* concentrations are implicit in the constant and coefficients of equation 7. Therefore, equation 7 may be invalid if the mean total-phosphorus concentration in Chickahominy Reservoir changed because the constant and coefficients may change.

Diascund Creek Reservoir

The quality of water in Diascund Creek Reservoir, as does that at Chickahominy Reservoir, depends on a number of factors and processes, including internal cycling of constituents, natural weathering reactions in the drainage basin, season, and climate. However, Diascund Creek Reservoir differs substantially from Chickahominy Reservoir in three important ways. First, Diascund Creek Reservoir is deeper than Chickahominy Reservoir, has a well-defined hypolimnion in the summer months, and has a much longer hydraulic retention time. Consequently, Diascund Creek Reservoir water quality is mainly affected by internal reservoir processes. Second, unlike Chickahominy Reservoir, Diascund Creek Reservoir is largely unaffected by

drainage-basin development. In fact, the lack of drainage-basin development indicates that tributary inflows to Diascund Creek Reservoir may closely resemble background or ambient quality for this area. Third, the reservoir level fluctuates more in Diascund Creek Reservoir because of differing operational constraints. Reservoir level affects internal processes in Diascund Creek Reservoir and thus affects water quality.

The purpose of this section is to describe the quality of Diascund Creek Reservoir, and how it varies spatially, seasonally, and with reservoir level. Although the quality of water in tributaries of Diascund Creek Reservoir was only determined for the last 13 months of this study, the information is discussed briefly because it provides insight into reservoir water quality.

Specific Conductance

The variation of specific conductance in Diascund Creek Reservoir is affected primarily by the quality of tributary inflows and the cycling of various constituents between an anaerobic hypolimnion and associated bottom sediments. Tributary inflows

affect the temporal and spatial distribution of specific conductance in the upper 5 to 10 ft of water. Because of its long hydraulic retention time, the specific conductance of the upper 5 to 10 ft of water changes slowly in Diascund Creek Reservoir; however, annual differences in specific conductance are evident in figure 16. These differences appear to be inversely related to flow conditions. (It is assumed that unit flows in the Diascund Creek Reservoir drainage basin are similar to unit flows recorded for the Chickahominy River basin (table 4).) The year of the largest amount of runoff (1984) was the year of the lowest mean specific conductance, whereas the year of the least amount of runoff (1986) was the year of the highest specific conductance (fig. 16). As noted by Hem (1985, p. 181-185), specific conductance is commonly related inversely to streamflow.

Time-depth graphs, as shown in figure 16 and elsewhere in the report, are constructed from measurements made on many occasions and at multiple depths at a single station. These graphs illustrate how the vertical profile of a constituent or measurement (such as specific conductance) differs in a lake or reservoir over a period of time. For example, figure 16 shows that specific conductance in Diascund Creek Reservoir was constant with depth in January 1986, averaging 70 $\mu\text{S}/\text{cm}$. In contrast, specific conductance in August 1986 averaged 90 $\mu\text{S}/\text{cm}$ near the water surface but increased markedly with depth, exceeding 200 $\mu\text{S}/\text{cm}$ near the sediment-water surface.

The upper 10 ft of water at the two Diascund Creek Reservoir sampling sites differ slightly in specific conductance. Specific conductance in water from the upper 10 ft at sampling sites 02042734 and 02042746 averaged 72 and 80 $\mu\text{S}/\text{cm}$, respectively (table 21). Specific conductance at sampling site 02042746 is greater because it is on the side of the causeway that receives inflows from Beaverdam Creek (fig. 3). The flow-weighted specific conductance of Beaverdam Creek inflows averaged 149 $\mu\text{S}/\text{cm}$ for October 1985 through October 1986, compared with 62 and 82 $\mu\text{S}/\text{cm}$ for Diascund Creek and Wahrani Swamp inflows, respectively (table 22). Because the causeway impedes horizontal mixing in Diascund Creek Reservoir, waters to the west of the causeway (sampling site 02042734) are less affected by inflows from Beaverdam Creek and, consequently, have lower specific conductances.

Inflows from Beaverdam Creek have the highest specific conductance of the reservoir tributaries for two reasons. First, runoff of road salt has the largest effect on Beaverdam Creek, as evidenced by the highest flow-weighted chloride concentration of the three tributaries (table 22). Second, Beaverdam Creek apparently drains an area of the Yorktown Formation containing the most calcareous marine deposits (shell beds) in the reservoir drainage basin. These deposits account for flow-weighted alkalinity concentrations nearly twice as high as for the other tributaries (table 22). Calcareous marine deposits yield calcium and bicarbonate (alkalinity) to the

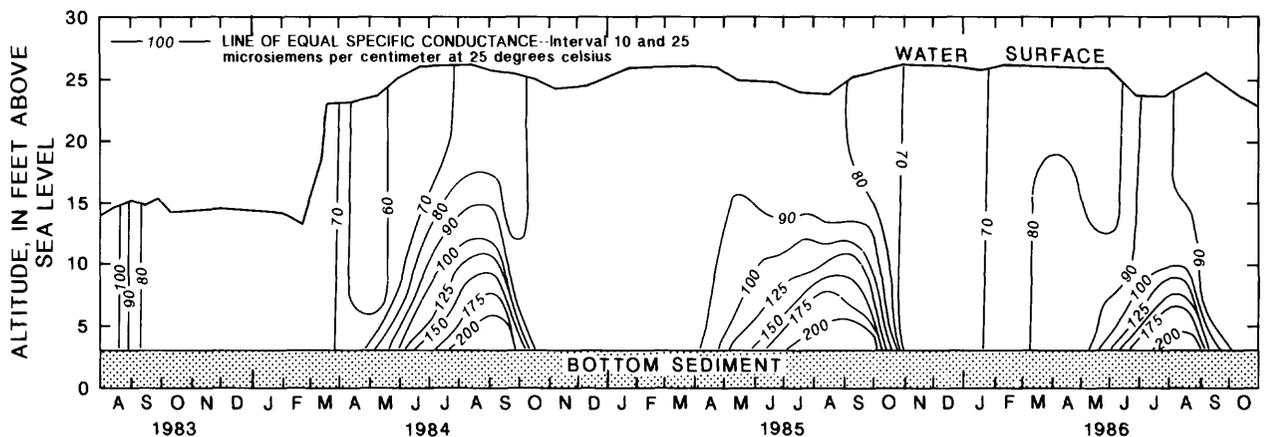


Figure 16.--Specific conductance in water from Diascund Creek Reservoir, 1983-86.

Table 21.—Average seasonal specific conductance, pH, water temperature, and dissolved-oxygen concentration, in water from sampling sites 02042734 and 02042746 in Diascund Creek Reservoir, 1983-86

[Sampling sites shown in figure 3]

Depth, in feet	Water temperature, in degrees Celsius		Dissolved oxygen, in milligrams per liter		Dissolved oxygen, in percent of saturation		Specific conductance, microsiemens per centimeter		pH	
	02042734	02042746	02042734	02042746	02042734	02042746	02042734	02042746	02042734	02042746
January-March										
3	8.5	8.5	11.5	11.2	98	94	65	79	6.7	6.7
10	8.0	8.0	11.1	10.9	93	92	62	78	6.7	6.8
18	7.5	7.5	11.0	11.0	91	92	69	78	6.7	6.8
April-June										
3	22.5	22.5	8.1	8.1	94	94	72	73	6.7	6.7
10	21.0	21.0	6.0	7.3	67	83	81	85	6.7	6.7
18	15.0	15.5	1.8	2.0	17	19	82	87	6.5	6.5
July-September										
3	26.0	26.0	7.0	7.1	87	88	76	81	6.8	7.0
10	25.0	25.5	3.8	4.1	45	49	82	88	6.6	6.8
18	20.0	20.5	0.0	0.0	0	0	121	145	6.5	6.6
October-December										
3	14.5	14.5	8.3	8.4	80	81	71	77	6.7	6.8
10	14.5	14.5	7.6	7.9	73	76	72	75	6.7	6.8
18	13.5	14.0	8.0	7.4	75	70	71	77	6.7	6.9
Annual Average										
3	18.0	18.0	8.7	8.7	90	89	71	78	6.7	6.8
10	17.0	17.0	7.1	7.5	70	75	74	82	6.7	6.8
18	14.0	14.5	5.2	5.1	46	45	86	97	6.6	6.7

water upon carbonic acid weathering. Because of the heterogeneous distribution of gravel, sand, silt, clay, and shell beds in the Yorktown Formation (Mixon and others, 1989), it is common to find significantly different specific conductances and concentrations of calcium and alkalinity in streams draining adjacent basins.

The most prominent feature in figure 16 is the steady increase in specific conductance of water near the bottom of the reservoir during summer stratification. This increase begins in late April or early May, corresponding to oxygen depletion at the sediment-water interface. By July of each year, specific conductance near the sediment-water interface exceeded 200 $\mu\text{S}/\text{cm}$, except in 1983 when the reservoir water level was drawn down. Release of bicar-

bonate (alkalinity), ammonia, and reduced species of iron and manganese from anaerobic bottom sediments accounts for the increase in specific conductance. The processes affecting the release of these ions are discussed in the following paragraphs.

Ions released from the anaerobic bottom sediments move in the hypolimnion through diffusion, dispersion, and advection; however, vertical transport of released ions is slow. The steep vertical gradient of specific conductance in the hypolimnion, with the highest values occurring near the source of the released ions (fig. 16), indicates that vertical transport is probably dominated by diffusion and perhaps by vertical dispersion associated with horizontal currents in the hypolimnion.

Table 22.--Average flow-weighted values of selected constituents and measurements in water from three major tributaries to Diascund Creek Reservoir, October 1985 through October 1986

[$\mu\text{S}/\text{cm}$, microsiemens per centimeter at 25 degrees Celsius; mg/L, milligrams per liter]

Constituent or property	Flow-weighted values		
	Diascund Creek (02042726)	Beaverdam Creek (02042736)	Wahrani Swamp (02042742)
Specific conductance, $\mu\text{S}/\text{cm}$	62	149	82
pH	6.5	6.5	6.3
Alkalinity, mg/L as CaCO_3	19	36	21
Chloride, dissolved, mg/L	5.9	19	7.6
Silica, dissolved, mg/L as SiO_2	6.2	6.5	7.4
Nitrite plus nitrate, dissolved, mg/L as N	.019	.02	.085
Ammonia, dissolved, mg/L as N	.03	.03	.03
Ammonia plus organic nitrogen, total, mg/L as N	.5	.6	.8
Nitrogen, total, mg/L	.5	.6	.9
Phosphorus, total, mg/L as P	.103	.096	.098
Phosphorus, ortho dissolved, mg/L as P	.024	.025	.029
Iron, total, mg/L	2.2	2.8	3.3
Iron, dissolved, mg/L	.67	.75	.97
Manganese, total, mg/L	.063	.066	.10
Manganese, dissolved, mg/L	.054	.061	.070
Organic carbon, total, mg/L	9.6	8.6	13

Epilimnion water quality is largely unaffected by the release of ions from anaerobic bottom sediments. As has been previously discussed, vertical transport of these ions is slow in the hypolimnion, providing only a small source to the epilimnion. Moreover, ions that are vertically transported to the epilimnion rapidly precipitate in the aerobic environment and settle out of the water column. At fall overturn, a similar process occurs, but on a much larger scale. As hypolimnetic water is vertically mixed and exposed to oxygen, many of these ions rapidly precipitate and are redeposited to the bottom sediments. Consequently, entraining water from the hypolimnion during fall overturn has little effect on the specific conductance of reservoir water during September or October (fig. 16). This process of sediment release, followed by redeposition, is referred to as "internal cycling."

pH and Water Temperature

The pH of water from Diascund Creek Reservoir typically ranges from 6.5 to 7.0 (table 21) and is relatively constant during the year. From October through March, when Diascund Creek Reservoir is isothermal, the pH is constant with depth, averaging 6.7 to 6.9. Between April and September, however, the hypolimnion pH is 0.2 to 0.3 units lower than the epilimnion pH. Decomposition of organic matter in the hypolimnion, which results in an accumulation of carbon dioxide (a weak acid), accounts for the lower pH values. The hypolimnion pH does not drop below about 6.5 in Diascund Creek Reservoir because decomposition of organic matter in an anaerobic hypolimnion consumes hydrogen ions, offsetting the acidity contributed by carbon dioxide.

Occasionally, pH is greater than 7.0 in the upper 10 ft of water in Diascund Creek Reservoir when photosynthetic uptake of carbon dioxide exceeds the transfer of atmospheric carbon dioxide across the air-water interface. Uptake of carbon dioxide by algae temporarily decreases the acidity of water, resulting in an increase in pH (Stumm and Morgan, 1981).

Diascund Creek Reservoir is monomictic, undergoing one primary period of thermal stratification (thermal-density layering) during the warmer months (fig. 16). Stratification typically begins in early April and ends in September or October. Annual minimum water temperatures in Diascund Creek Reservoir typically occur in January and range from 0 to 4 °C. Annual maximum temperatures typically occur in July or August and periodically exceed 28 °C. During some winters, when an ice cover forms (January 1984 and 1985), Diascund Creek Reservoir becomes weakly stratified (fig. 17). During these short periods of winter stratification, warmer, more dense water (4 °C) is present near the reservoir bottom, and colder, less dense water (0 °C) is present near the ice cover. Because winter stratification does not occur every year and is generally very slight and shortlived, this phenomenon probably has little effect on the quality of water in Diascund Creek Reservoir.

The timing and degree of summer stratification depends primarily on climatic conditions. Air temperature, insolation, and wind speed and direction are the most significant factors. The progression of summer stratification at sampling sites 02042734 and 02042746 in 1985 is illustrated in figure 18. Isothermal conditions existed in Diascund Creek Reservoir from November 1984 through March 1985, as indicated by the uniform (vertical isotherms) temperature profiles for February 28, 1985, in figure 18. During isothermal conditions, the water column is thoroughly mixed by wind- and convection-generated currents. As air temperatures warm in the spring and insolation intensifies, water at the surface of the reservoir progressively warms. Thermal stratification begins when winds are not adequate to mix the warmer, less-dense surface layer with the cooler bottom water. By April 23, 1985, thermal stratification was pronounced at both stations, with epilimnion and hypolimnion temperatures at about 24.5 and 14 °C, respectively (fig. 18). The thin (5 ft thick), but well-defined epilimnion in April is typical of early stratification established during a warm, calm period. During the summer, the epilimnion is

about 10 to 15 ft thick, as typified by the September 4, 1985, profile in figure 18. Hypolimnion temperatures increase steadily during summer stratification, and typically reach 20 to 25 °C by September (fig. 17). Heat flow from the epilimnion accounts for most of this temperature rise. Reservoir destratification does not occur until September or October, when cooling of surface water and vertical mixing generated by wind and thermal convection have reduced temperature differences between surface and bottom waters. The reservoir is isothermal after complete mixing, as illustrated by the nearly vertical temperature profiles on October 24, 1985, at both sampling sites (fig. 18). (The process of destratification is referred to as fall overturn.) Except during short periods of ice cover, isothermal conditions persist in Diascund Creek Reservoir until spring.

Water temperatures are similar at the two Diascund Creek Reservoir sampling sites (table 21). The progression of stratification at the two sampling sites in 1985 is also similar (fig. 18). Apparently, factors that affect heating, cooling, and mixing of water are generally uniform throughout Diascund Creek Reservoir; however, seasonal water temperatures and the timing of stratification and fall overturn varied among years. For example, epilimnion water temperatures in the summer of 1985 averaged 28.5 °C, compared with 24.5 °C in 1986; these temperature differences are the result of climatic differences in the 2 years. The cooler epilimnion water temperatures in 1986 contributed to a fall overturn that occurred 1 month earlier than in 1985 (fig. 17).

Reservoir levels in Diascund Creek Reservoir affects thickness of the hypolimnion. In 1983, a well-defined hypolimnion never developed because wind-generated mixing kept the water column (maximum depth 13 ft) nearly isothermal, as illustrated by the vertical temperature isotherms in figure 17. For a short period in August 1983, however, Diascund Creek Reservoir was weakly stratified, with the bottom waters 1 to 2 °C cooler than the upper waters.

Dissolved Oxygen

The distribution of DO concentrations in water from Diascund Creek Reservoir is affected by mixing patterns and thermal stratification. During isothermal periods, when vertical mixing keeps the water in contact with atmospheric oxygen, water in Diascund Creek Reservoir is nearly saturated with DO, and concentrations are relatively constant with

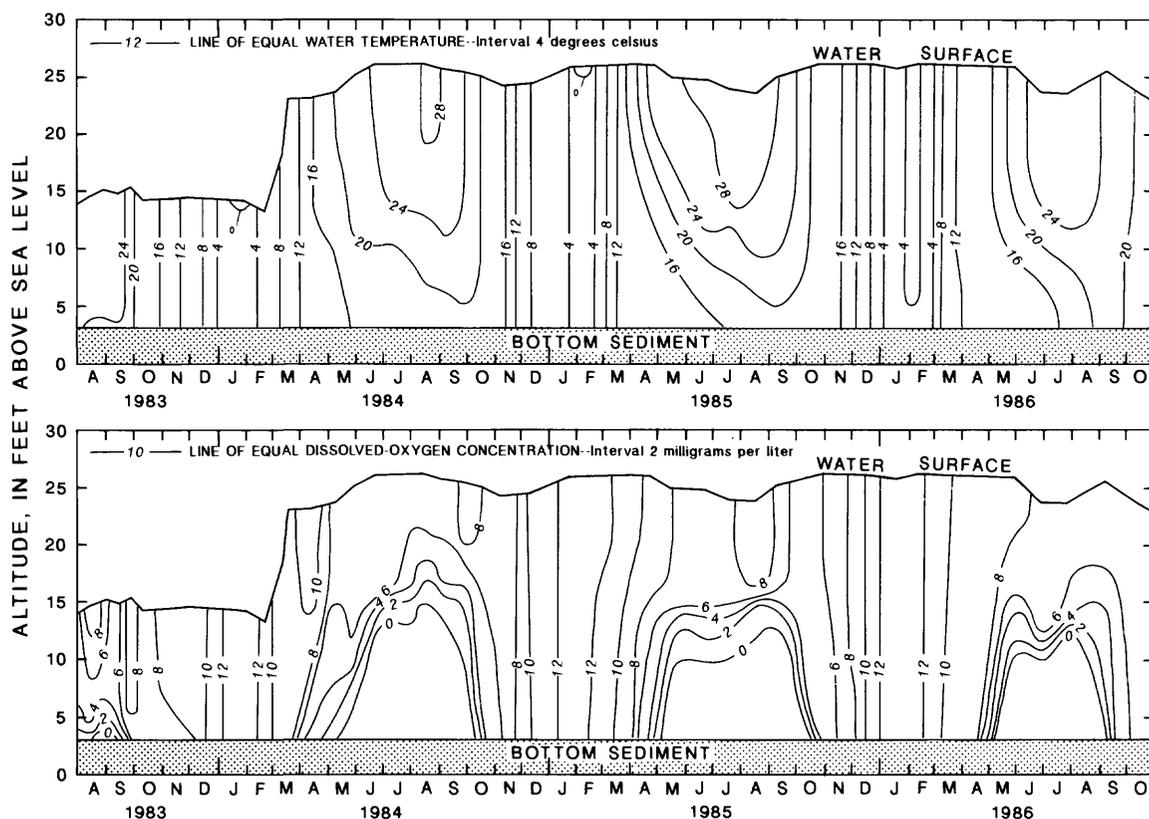


Figure 17.--Water temperature and dissolved-oxygen concentrations in water from Diascund Creek Reservoir, 1983-86.

depth, as depicted by the February 28, 1985, profiles in figure 18. This mixing pattern is illustrated by the nearly vertical lines of equal DO concentrations in figure 17 for the fall and winter months. During isothermal periods and in the epilimnion during summer stratification, DO concentrations are largely affected by water temperature. DO concentrations tend to be highest in winter months and lowest in summer months because the solubility of oxygen in water is inversely related to water temperature. Concentrations in the epilimnion typically range from 10 to 14 mg/L in the winter and from 6.5 to 8.5 mg/L in the summer.

Although the mixed layers of water in Diascund Creek Reservoir tend to be nearly saturated with DO, bacterial decomposition of organic matter, along with plant and animal respiration, can consume large quantities of DO; however, consumption of DO is largely offset by renewal mechanisms, including photosynthetic generation of oxygen and transfer of oxygen across the air-water interface. DO concentrations can differ considerably throughout a given day, from day to day, and at different depths,

because (1) exchange of oxygen across the air-water interface is slow, and (2) vertical mixing of DO is not instantaneous. For example, DO occasionally supersaturates the water in Diascund Creek Reservoir when photosynthetic generation of oxygen exceeds the rate of oxygen transfer across the air-water interface. DO concentrations exceeding 120 percent saturation have been measured. In contrast, water in the lower epilimnion can be undersaturated during the same time period. The summer DO concentration at a depth of 10 ft averaged 4.0 mg/L at both sampling sites in Diascund Creek Reservoir, which equals 50 percent of saturation (table 21). Poor vertical mixing of the epilimnion, which occurs during hot, calm weather, can create conditions where oxygen demand in the lower epilimnion temporarily exceeds reaeration rates. Oxygen demand in the lower epilimnion comes from (1) decomposition of settling organic matter, (2) respiration of algae positioned below the euphotic zone, and (3) upward diffusion of reduced constituents from the hypolimnion, particularly iron and manganese.

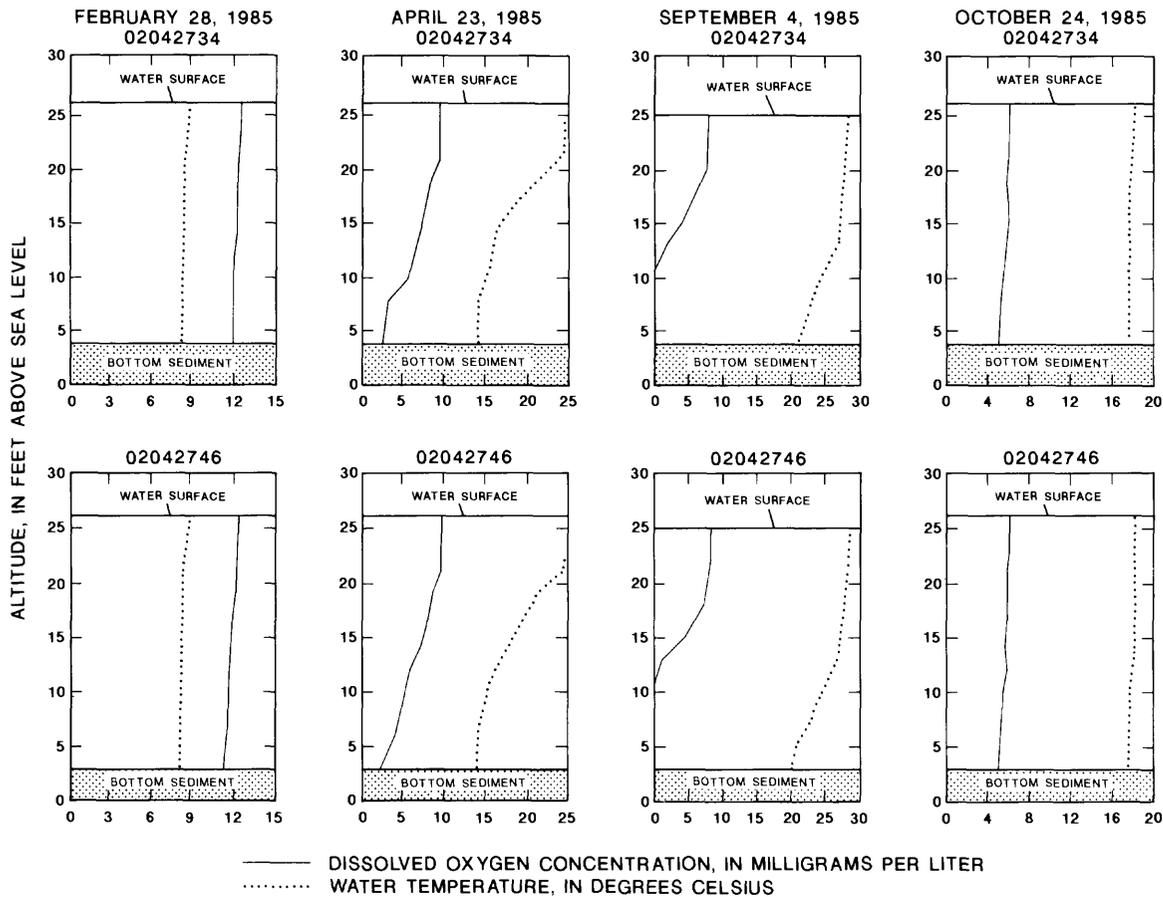


Figure 18.—Water temperature and dissolved-oxygen concentrations in water from sampling sites in Diascund Creek Reservoir on selected dates in 1985.

Severe hypolimnetic oxygen deficits occur during the spring and summer months in Diascund Creek Reservoir because thermal stratification prevents vertical mixing of water. The lack of vertical mixing isolates the hypolimnion from a renewing source of oxygen by eliminating its contact with the air and limiting exposure to light for photosynthetic generation of oxygen. Hypolimnion DO is depleted by decomposition of particulate organic matter settling from the epilimnion and by benthic oxygen demand. Plant and animal respiration is probably of secondary importance. Oxygen is first depleted near the reservoir bottom (see the April 23, 1985, profiles in figure 18); chemical and biological processes in and near the benthos create a strong oxygen demand on the hypolimnion. This demand for oxygen rapidly depletes the supply of DO in the small hypolimnion. At full pool, the hypolimnion averages only 4 ft in thickness and comprises less than 20 percent of the reservoir volume. As illustrated in figure 18, the DO concentration was low (2 mg/L) at the sediment-water interface as early as

April for both sampling sites. The entire hypolimnion is anaerobic by late May and remains anaerobic until fall overturn in September or October (fig. 17). The September 4, 1985, DO profiles in figure 18 illustrate typical late-summer stratification in Diascund Creek Reservoir.

In contrast to conditions in 1984 through 1986, Diascund Creek Reservoir did not develop a distinct hypolimnion in 1983 because of reservoir draw-down. Consequently, oxygen was present throughout the water column during most of the summer. Only during a short period of stratification in late August and early September did anaerobic conditions develop near the sediment-water interface at sampling site 02042746 (fig. 18).

During summer stratification, the altitude of DO concentrations at sampling site 02042746 changed in response to weather conditions and reservoir pumpage. For example, incomplete vertical mixing of the epilimnion during a calm, hot period in August 1984 resulted in low DO concentrations at

shallow depths (fig. 17). Intensive pumping from Diascund Creek Reservoir during June 1985 and 1986, as evidenced by declining reservoir water levels in figure 17, resulted in temporarily elevated DO concentrations in the metalimnion at sampling site 02042746. Because of the position of the raw-water intake, water with low DO concentrations is pumped from the metalimnion and (or) the hypolimnion, and replaced with epilimnetic water with a higher DO concentration. The thinning of the anaerobic hypolimnion in June 1985 and 1986 indicates that water of poor quality is occasionally withdrawn from Diascund Creek Reservoir during summer stratification. The effects of pumping on thermal stratification are evident at sampling site 02042746 because of its proximity (300 ft) to the raw-water intake. Similar effects of pumping were not observed at sampling site 02042734 because of its distance (3,000 ft) from the raw-water intake.

DO concentrations in Diascund Creek Reservoir can be relatively low throughout the water column for several weeks after fall overturn. DO undersaturates the water because of the heavy oxygen demands exerted by organic matter and reduced species of iron, manganese, and nitrogen in previously anaerobic bottom sediments and waters. Because of this large oxygen demand, DO concentrations during the month of October averaged only 6.0 mg/L during the study, less than 60 percent of saturation. Undersaturation is evident at both reservoir stations, as illustrated in figure 18 by DO profiles with similarly low concentrations (averaging 5.7 mg/L) on October 24, 1985. The effects of fall overturn on DO concentrations are also evident in November. By mid-November of 1984 and 1985, DO saturation in Diascund Creek Reservoir was only about 70 percent; DO concentrations averaged about 7 mg/L (fig. 17).

Major Inorganic Constituents

For most of the year, Diascund Creek Reservoir contains a calcium bicarbonate-type water. Calcium and alkalinity concentrations in the upper 3 ft of water in Diascund Creek Reservoir averaged 10 and 22 mg/L, respectively (table 23). The source of these constituents is calcareous marine deposits in the drainage basin, which undergo congruent dissolution when weathered by carbonic acid. The average calcium-to-bicarbonate mole ratio measured in Diascund Creek Reservoir (August 1983 through

December 1984) was 0.55, which is similar to the mole ratio of 0.50 expected from the following dissolution reaction of calcareous marine deposits:



The calcium-to-bicarbonate mole ratio differs seasonally, depending on oxidation-reduction reactions of sulfur in the drainage basin. These reactions also explain the inverse relation between sulfate and alkalinity (bicarbonate) concentrations in water from Diascund Creek Reservoir. When sulfate is being reduced in the basin (primarily through spring and summer vegetative uptake and sulfate reduction in anaerobic soils and sediments), hydrogen ions are consumed; therefore, bicarbonate ion is found in concentrations in excess of what would be expected from dissolution of calcareous deposits. This is seen as a calcium-to-bicarbonate mole ratio of less than 0.50, which occurred on several occasions in the summer of 1984. In contrast, when reduced forms of sulfur are oxidized in the basin (primarily during fall and winter decomposition of leaf litter), hydrogen ions are generated; therefore, the concentration of bicarbonate ions is less than would be expected from the dissolution of marine deposits. This is seen as a calcium-to-bicarbonate mole ratio of greater than 0.50. A ratio exceeding 0.90 was measured in November 1983.

Alkalinity concentrations in the hypolimnion of Diascund Creek Reservoir are affected by ambient reservoir concentrations at the time of stratification and release of bicarbonate ions from reducing bottom sediments. The alkalinity concentration at a depth of 18 ft in Diascund Creek Reservoir was 80 mg/L on September 4, 1984, compared with 27 mg/L in the overlying epilimnion, a threefold difference (table 24). Bicarbonate ion is generated in or near anaerobic bottom sediments when oxidized forms of iron and manganese are biologically reduced during oxidation of organic matter. As discussed earlier, the buildup of bicarbonate ions in the hypolimnion does not significantly affect the alkalinity of the reservoir after fall overturn. Hydrogen ions released from the oxidation of reduced iron and manganese in the hypolimnion quickly neutralize bicarbonate ions generated in the hypolimnion during stratification. This process is discussed in detail in the section, "Iron and Manganese."

Table 23.--Concentrations of selected dissolved constituents at a depth of 3 feet in water from Diascund Creek Reservoir (02042746), 1983-86

[Concentration in milligrams per liter; --, no data available; <, less than]

Date	Calcium	Magnesium	Sodium	Potassium	Alkalinity, as CaCO ₃	Sulfate, as SO ₄	Chloride	Fluoride	Silica, as SiO ₂	Hardness, as CaCO ₃	Solids
08-04-83	12	0.93	3.4	0.9	33	6.2	6.0	0.1	5.6	34	55
08-16-83	12	1.0	4.1	1.2	26	9.3	5.7	<.1	6.9	34	57
08-31-83	11	.97	3.7	1.2	24	7.9	5.7	.1	7.9	31	53
09-14-83	11	.96	3.8	1.2	22	9.2	5.8	.1	9.3	31	55
09-28-83	11	.90	4.0	1.3	21	9.6	5.7	.1	8.2	31	54
10-11-83	10	.91	3.7	1.4	17	12	5.7	<.1	6.3	29	50
11-03-83	11	.96	4.7	1.4	15	17	6.8	<.1	4.2	31	56
11-30-83	11	.92	4.0	1.8	17	17	7.4	<.1	6.2	31	60
03-14-84	9.3	.76	3.9	1.0	18	7.9	6.6	<.1	4.2	26	46
04-18-84	6.0	.66	2.5	1.0	13	6.0	4.3	<.1	2.7	18	31
05-08-84	7.1	.78	2.6	.9	16	6.0	4.7	.1	2.2	21	35
05-31-84	7.5	.76	2.5	.9	18	4.9	4.4	<.1	1.7	22	35
06-20-84	8.4	.88	2.7	1.0	20	3.8	4.6	<.1	2.4	25	37
07-10-84	8.9	.86	2.7	1.0	23	3.8	4.5	<.1	2.5	26	39
08-01-84	9.5	.89	2.6	1.0	25	3.2	4.5	<.1	3.0	27	40
08-14-84	9.2	.94	2.6	1.0	26	2.6	4.5	.1	2.7	27	40
09-04-84	9.8	.89	2.7	1.0	27	2.2	4.6	<.1	3.2	28	41
09-26-84	11	.97	2.9	1.1	29	1.9	4.8	<.1	3.3	31	45
10-17-84	11	.98	2.8	1.1	29	2.0	4.8	<.1	3.1	32	44
11-07-84	11	.96	2.9	1.1	27	3.1	4.9	<.1	3.1	31	44
12-11-84	9.8	.90	3.0	1.0	25	3.1	5.1	<.1	2.9	28	41
02-28-85	--	--	--	--	--	--	--	--	3.8	--	-
04-23-85	--	--	--	--	20	--	7.7	--	2.6	--	-
05-14-85	--	--	--	--	21	--	8.1	--	1.7	--	-
06-20-85	--	--	--	--	24	--	7.4	--	2.6	--	-
07-17-85	--	--	--	--	26	--	7.5	--	3.1	--	-
08-12-85	--	--	--	--	26	--	7.4	--	3.4	--	-
09-04-85	--	--	--	--	25	--	7.5	--	3.4	--	-
10-24-85	--	--	--	--	20	--	5.7	--	3.4	--	-
11-27-85	--	--	--	--	18	--	5.7	--	3.7	--	-
12-18-85	--	--	--	--	17	--	6.1	--	4.2	--	-
02-05-86	--	--	--	--	17	--	6.5	--	4.1	--	-
03-20-86	--	--	--	--	18	--	9.5	--	3.4	--	-
04-24-86	--	--	--	--	24	--	10	--	1.9	--	-
05-28-86	--	--	--	--	22	--	9.3	--	81	--	--
06-25-86	--	--	--	--	26	--	8.1	--	1.2	--	-
07-23-86	--	--	--	--	26	--	8.5	--	1.6	--	--
09-04-86	--	--	--	--	25	--	7.8	--	2.8	--	--
10-16-86	--	--	--	--	27	--	8.1	--	2.6	--	--
Mean	10	.89	3.2	1.1	22	6.6	6.4	<.1	3.6	28	46
Maximum	12	1.0	4.7	1.8	33	17	10	.1	9.3	34	60
Upper quartile	11	.96	3.8	1.2	26	9.2	7.5	.1	4.2	31	54
Median	10	.91	2.9	1.0	24	6.0	5.9	<.1	3.1	29	44
Lower quartile	9.2	.88	2.7	1.0	18	3.1	4.8	<.1	2.6	26	40

Table 24.--Typical late-summer values of selected constituents and measurements in water from the epilimnion and hypolimnion of Diascund Creek Reservoir (02042746), September 4, 1984

[$\mu\text{S/cm}$, microsiemens per centimeter at 25 degrees Celsius; mg/L, milligrams per liter]

Constituent or measurement	Depth, in feet	
	3	18
Specific conductance, $\mu\text{S/cm}$	75	162
Solids, sum of constituents, dissolved, mg/L ¹	41	109
pH	7	6.7
Water temperature, degrees Celsius	26	21
Oxygen, dissolved, mg/L	6.3	0
Calcium, dissolved, mg/L	10	14
Magnesium, dissolved, mg/L	.9	1.1
Sodium, dissolved, mg/L	2.7	2.7
Potassium, dissolved, mg/L	1	1.4
Alkalinity, mg/L as CaCO_3	27	80
Bicarbonate, mg/L	33	98
Sulfate, dissolved, mg/L as SO_4	2.2	3.1
Chloride, dissolved, mg/L	4.6	6.6
Fluoride, dissolved, mg/L	<.1	<.1
Silica, dissolved, mg/L as SiO_2	3.2	7.5
Nitrite plus nitrate, dissolved, mg/L as N	<.1	<.1
Ammonia, dissolved, mg/L as N	<.01	.67
Ammonia plus organic nitrogen, total, mg/L as N	.6	1.3
Nitrogen, total, mg/L	.6	1.3
Phosphorus, total, mg/L as P	.03	.04
Phosphorus, ortho dissolved, mg/L as P	<.01	<.01
Iron, dissolved, mg/L	.50	22
Manganese, dissolved, mg/L	.03	2.4
Organic carbon, total, mg/L	8.5	8.5

¹ Includes dissolved iron and manganese in the calculation.

Sulfate concentration in the upper 3 ft of water in Diascund Creek Reservoir averaged 6.6 mg/L from August 1983 through December 1984 (table 23). Atmospheric deposition is probably the primary source of sulfate in the drainage basin. Sulfate can be cycled through several oxidation states and through microbes and vegetation many times before reaching Diascund Creek Reservoir. Biologically driven oxidation-reduction in the drainage basin and reservoir bottom sediments are most responsible for seasonal variability of sulfate concentrations. Concentrations are commonly at a maximum in the fall and winter months. Sulfate concentrations in water from Diascund Creek Reservoir reached a maximum of 17 mg/L in November 1983, probably as a result of runoff of leaf-litter leachate. The maximum concentrations occurred during reservoir drawdown when inflows were mixed with only one-fifth of the normal reservoir volume in November 1983. At full pool, the seasonal maximum sulfate concentration in Diascund Creek Reservoir would be lower as fall and winter runoff is diluted with a much larger volume of water.

Similar to sulfate, the primary source of magnesium and potassium ions in water from Diascund Creek Reservoir is probably atmospheric deposition. Concentrations of both ions are low, averaging about 1.0 mg/L, and are relatively constant throughout the year (table 23). Both ions could also come from carbonic acid weathering of small amounts of magnesium- and potassium-bearing minerals in the basin.

The primary source of sodium and chloride ions in water from Diascund Creek Reservoir is atmospheric deposition and road-salt application for deicing. Sodium and chloride concentrations in the upper 3 ft of water in Diascund Creek Reservoir averaged 3.2 and 5.3 mg/L, respectively, from August 1983 through December 1984 (table 23). These concentrations equate to a sodium-to-chloride mole ratio of 0.93, which is similar to the mole ratio of 0.85 for seawater (Hem, 1985). This similarity of mole ratios indicates that atmospheric sodium and chloride ions originate as sea-salt aerosols; however, road salt can periodically be an important source of these ions. The maximum chloride concentration (10 mg/L) in water from Diascund Creek Reservoir was measured in April 1986 (table 23), 2 months after a large snowfall in the basin and the application of a substantial amount of road salt in the drainage basin (table 23).

Silica in water from Diascund Creek Reservoir comes primarily from congruent dissolution of quartz in the basin, and to a lesser extent, from incongruent dissolution of aluminum silicates. Flow-weighted concentrations of dissolved silica in the three main reservoir tributaries ranged from 6.2 to 7.4 mg/L from October 1985 through October 1986 (table 22). For this same period of time, the average dissolved silica concentration in the upper 3 ft of reservoir water averaged 2.7 mg/L (table 23), representing an approximate 60-percent loss of imported dissolved silica. Uptake of silica by diatoms, and the subsequent sedimentation of their frustules (cell walls), represents the major sink for dissolved silica. This sink is most dominant in the spring months when diatom populations are typically largest. Minimum concentrations of dissolved silica occurred in May of each year of the study, with concentrations decreasing to less than 2.0 mg/L (fig. 19). As diatoms were replaced by other algae following thermal stratification, however, the dissolved-silica concentrations increased because sources from tributary inflows and dissolution of

diatom frustules in bottom sediments were greater than losses from uptake by the diminished diatom population.

Dissolution of diatom frustules provides an internal source of dissolved silica to water in Diascund Creek Reservoir. Dissolution occurs within bottom sediments and as frustules settle in the water column. During the summer, concentrations of dissolved silica in the upper 3 ft of water are relatively constant (fig. 19), with internal releases and tributary inflows approximately balancing diatom uptake and sedimentation. In the hypolimnion, however, concentrations of dissolved silica increase rapidly in the summer because dissolution processes exceed diatom uptake (as a result of poor light conditions). Dissolved silica in water at a depth of 18 ft reached a concentration of 7.5 mg/L (as SiO_2) by early September 1984; this concentration was considerably greater than in the epilimnion, 3.2 mg/L, on the same date. Silica concentrations at depths of 3 and 18 ft in water from Diascund Creek Reservoir converged during fall overturns and remained approximately equal during isothermal periods (fig. 19).

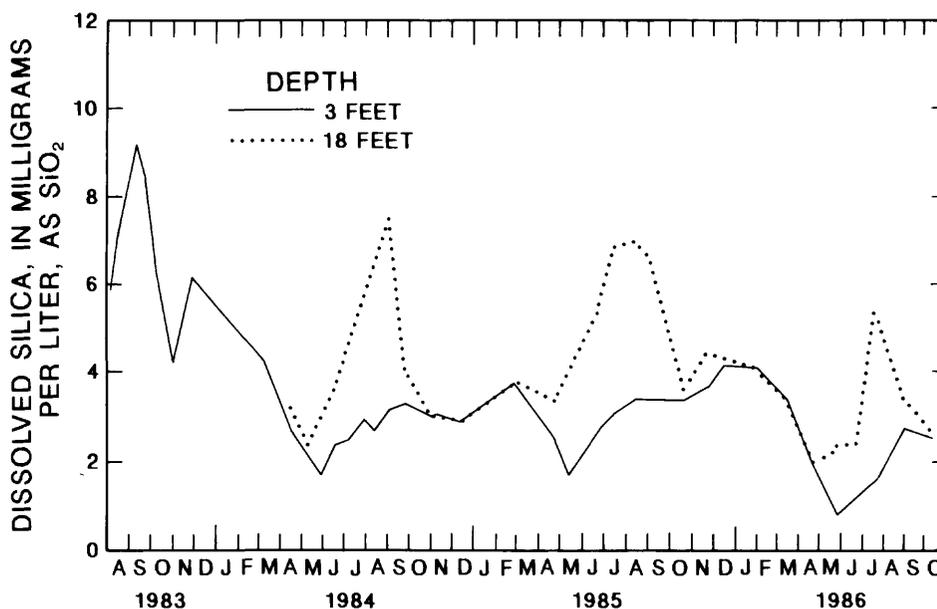


Figure 19.--Dissolved-silica concentrations at depths of 3 and 18 feet in water from Diascund Creek Reservoir, 1983-86.

Concentrations of dissolved silica in the upper 3 ft of water in Diascund Creek Reservoir averaged 6.8 mg/L during 1983, which is more than double the average concentration (2.8 mg/L) during subsequent years (table 23). A different balance between silica sources and sinks during reservoir drawdown resulted in higher concentrations in 1983 compared with 1984-86. Drawdown favors high dissolved-silica concentrations by (1) preventing the development of a hypolimnion, which is a seasonal sink for silica; (2) decreasing the time for diatom uptake by shortening the reservoir hydraulic retention time; (3) concentrating silica released from the sediment in a smaller reservoir volume; (4) increasing the temperature of bottom sediments, thereby increasing silica-dissolution rates; and (5) shifting the algal population away from diatoms because of high nutrient concentrations and (or) shallower water depths.

Flow conditions affect the concentration of several major inorganic constituents in Diascund Creek Reservoir and its tributaries. Because of the reservoir's long hydraulic retention time, changes in tributary quality associated with flow conditions need to persist for weeks or months to produce a measurable effect on reservoir water quality. Concentrations of weathering products in Diascund Creek Reservoir (primarily calcium, alkalinity, and silica) are higher during base-flow periods because reservoir inflows have a longer mean residence time in the basin, and thus, more reaction time with minerals than high-flow periods do. For example, lower-than-average flows during the summers of 1984 and 1986 resulted in October alkalinity concentrations of 29 and 27 mg/L, respectively (table 23). In contrast, high flows for several months prior to October 1985 resulted in an alkalinity concentration of 20 mg/L. High flows in the first 4 months of 1984 produced the lowest concentrations of calcium and alkalinity measured in Diascund Creek Reservoir during the study (table 23).

Iron and Manganese

Sand and clay in the Yorktown Formation contain appreciable quantities of iron and manganese and are undoubtedly the main source of these metals to water in Diascund Creek Reservoir. Iron and manganese enter the reservoir through its tributaries, ground-water inflows, and anaerobic bottom sediments.

Concentrations of total and dissolved iron in tributary inflows were elevated, averaging 2.8 and 0.8 mg/L, respectively, for October 1985 through October 1986 (table 22). For this same period, concentrations of total and dissolved iron in the upper 3 ft of water in Diascund Creek Reservoir averaged 0.59 and 0.27 mg/L, respectively. This discrepancy between inflow and in-lake concentrations of iron indicates that most imported iron (about 65 to 80 percent) is lost to bottom sediments.

Concentrations of dissolved iron in the upper 3 ft of water in Diascund Creek Reservoir result from a balance between sources (primarily external loading from tributaries) and sinks (sorption and settling to bottom sediments). As the external sources of dissolved iron decrease, such as during periods of lower-than-average flow, losses of dissolved iron temporarily exceed input, and the concentration in reservoir water decreases. The dissolved-iron concentration decreases until a new balance between sources and sinks is established. In contrast, as external sources increase, such as during periods of higher-than-average flow, inputs of dissolved iron temporarily exceed losses, and the reservoir's dissolved-iron concentration increases. The concentration increases until a new balance between sources and sinks is established.

The changing balance between sources and sinks of dissolved iron is related to flow conditions. The elevated dissolved-iron concentrations in the epilimnion of Diascund Creek Reservoir between May and September of 1984, averaging 0.8 mg/L at a depth of 3 ft (table 25), resulted from high flows and large external loads of dissolved iron from the drainage basin. Much of the iron that entered during this period was probably in an organic-metal complex and was thereby kept in solution. In contrast, low flows and small external loads of dissolved iron during the same months in 1985 and 1986 resulted in low epilimnion dissolved-iron concentrations in both years, averaging 0.1 mg/L (table 25).

Thermal stratification and the ensuing hypolimnetic DO depletion result in elevated dissolved-iron concentrations near reservoir bottom sediments at sampling site 02042746 that commonly exceed 15 mg/L by June (fig. 20). Concentrations of dissolved iron exceeding 20 mg/L have been measured at a depth of 18 ft in Diascund Creek Reservoir (table 24), and concentrations greater than this could be present in pore water of anaerobic sediments.

Table 25.—Dissolved-iron and manganese concentrations in water from multiple depths below the surface at sampling site 02042746 in Diascund Creek Reservoir, 1983-86

[Concentrations in milligrams per liter; --, no data available]

Date	Dissolved iron			Dissolved manganese		
	3 feet	10 feet	18 feet	3 feet	10 feet	18 feet
08-04-83	0.012	0.24	--	0.002	0.45	--
08-16-83	.10	.17	--	.047	.12	--
08-31-83	.023	.028	--	.002	.14	--
09-14-83	.090	.083	--	.16	.17	--
09-28-83	.027	.025	--	.011	.024	--
10-11-83	.078	.10	--	.020	.023	--
11-03-83	.12	.17	--	.035	.10	--
11-30-83	.40	.51	--	.094	.11	--
03-14-84	.48	.39	--	.030	.031	--
04-18-84	.25	--	.40	.004	--	.036
05-08-84	.59	--	.61	.023	--	.023
05-31-84	1.0	--	.6	.034	--	.49
06-20-84	1.3	--	3.7	.012	--	.88
07-10-84	1.0	--	10.	.007	--	1.3
08-01-84	.65	--	14.	.044	--	1.8
08-14-84	.48	.30	--	.008	.74	--
09-04-84	.50	--	22.	.030	--	2.4
09-26-84	1.2	--	2.5	.052	--	.80
10-17-84	.49	--	.53	.001	--	.059
11-07-84	.61	--	.18	.15	--	.099
12-11-84	.15	--	.13	.004	--	.004
02-28-85	.14	.13	.15	.010	.009	.010
04-23-85	.21	.21	.20	.005	.004	.12
05-14-85	.093	.18	2.0	.028	.084	1.4
06-20-85	.065	.086	8.3	.038	.12	2.2
07-17-85	.040	.073	17.	.004	.11	2.8
08-12-85	.024	.029	16.	.013	.004	2.8
09-04-85	.075	.12	15.	.066	.039	2.3
10-24-85	.33	.34	.36	.014	.012	.033
11-27-85	.38	.40	.64	.004	.004	.025
12-18-85	.58	.63	.44	.006	.011	.004
02-05-86	.56	.44	.54	.012	.012	.012
03-20-86	.26	.26	.27	.001	.001	.002
04-24-86	.16	.18	.24	.031	.033	.031
05-28-86	.14	.18	1.9	.003	.003	.76
06-25-86	.091	.038	2.0	.025	.003	1.6
07-23-86	.14	.072	17.	.050	.11	2.2
09-04-86	.19	.33	.43	.001	.007	.30
10-16-86	.10	.091	.10	.003	.003	.004

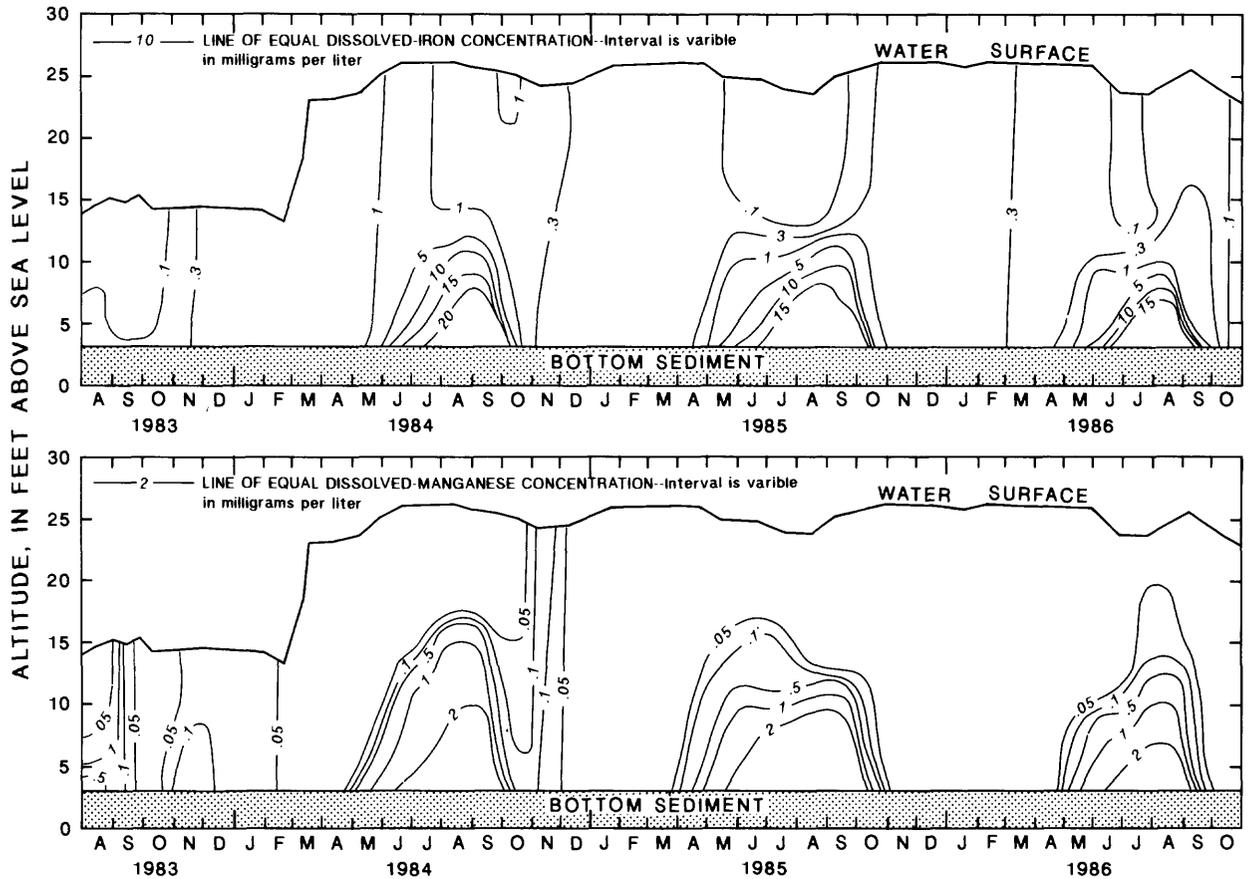
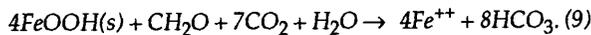


Figure 20.—Dissolved-iron and dissolved-manganese concentrations in water from Diascund Creek Reservoir, 1983-86.

Ferrous ions make up greater than 90 percent of the total iron in the anaerobic hypolimnion. Ferrous ions are released from anaerobic bottom sediments as solid-phase ferric oxyhydroxide is used as an electron acceptor in the oxidation of organic matter. Organic matter driving this reaction comes primarily from settling algal cells and detritus from the epilimnion, but organic matter present in the sediments before stratification could be important as well. Release of ferrous ions begins in early May in Diascund Creek Reservoir and continues through the period of thermal stratification. Each mole of ferrous ions released is accompanied by 2 moles of bicarbonate ions, as shown in the following reaction:



(ferric (organic (bacterial
oxyhydroxide) matter) mediation)

The stoichiometry of reaction 9 closely matches the ratio of ferrous ions to bicarbonate ions (alkalinity) that accumulate in the anaerobic hypolimnion of Diascund Creek Reservoir after accounting for other known sources of these constituents. For example, the corrected mole ratio of ferrous ions to bicarbonate ions on September 4, 1984 (depth of 18 ft in table 24), was 0.48, which is similar to the mole ratio of 0.50 expected from reaction 9. Hypolimnial concentrations of dissolved iron and bicarbonate ions are corrected by (1) subtracting quantities of these constituents present in the hypolimnion before becoming anaerobic, as represented by the sample collected on April 18, 1984; (2) subtracting bicarbonate ions entering with ground water, assuming that the hypolimnial increase in calcium concentrations results from inflow of a calcium bicarbonate-type water; and (3) subtracting bicarbonate ions generated by the reduction of manganese and mineralization of ammonia in anaerobic bottom sediments. Chemical reduction of iron in anaerobic bottom sed-

iments accounted for 98 percent of the dissolved iron and 48 percent of the bicarbonate ions (alkalinity) present in the hypolimnion on September 4, 1984.

The quantity of dissolved iron released from anaerobic bottom sediments is probably controlled by a ferrous-carbonate solid phase, such as siderite, through the following reaction (Stumm and Morgan, 1981):



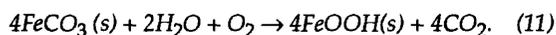
WATEQF, a mineral-solution equilibrium program (Plummer and others, 1976), was used to evaluate the possible effect of siderite on aqueous species of dissolved iron and carbonate in pore water of anaerobic bottom sediments. Pore water was not evaluated directly; however, water collected near the sediment-water interface (at a depth of 18 ft), close to the end of thermal stratification (September 4, 1984), was evaluated because it is probably representative of the chemistry of sediment pore water (table 24).

Water near anaerobic bottom sediments is saturated and perhaps slightly supersaturated with respect to siderite, as evidenced by a log (AP/K) value of 0.48. Log (AP/K) is the logarithm of the ratio of the activity product (AP) to the equilibrium constant (K) for a mineral. The activity product is obtained by multiplying the activities of the chemical species that form the mineral considered. For instance, the activity product for siderite is the product of the activities of ferrous ions and carbonate ions. As suggested by Paces (1972), values of log (AP/K) between -0.5 and 0.5 indicate possible equilibrium with the mineral considered.

Because carbonate ion represents one-half of the activity product of siderite, the activity (concentration) of ferrous ions in the hypolimnion is sensitive to factors affecting carbonate-ion activity. Higher carbonate-ion activities result in lower ferrous-ion activities. Carbonate-ion activity of a solution increases with alkalinity concentration and pH of the solution. Consequently, the alkalinity and pH of water in the hypolimnion at the beginning of thermal stratification could partly control the quantity of ferrous ions released from anaerobic bottom sediments.

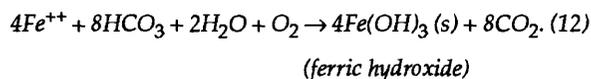
The quantity of siderite formed in bottom sediments is undoubtedly related to the availability of organic matter to drive reaction 9. The pore-water concentration of ferrous ions results from a balance between their formation (reaction 9) and loss through transport to the hypolimnion [diffusion and (or) advection]. A large supply of organic matter to bottom sediments results in the formation of ferrous ions in pore water that exceeds their transport from bottom sediments, thereby allowing pore water to become saturated with respect to siderite. Under these conditions, siderite precipitates and accumulates in bottom sediments. In contrast, when the supply of organic matter is small, transport of ferrous ions from the sediment is adequate to keep pore water undersaturated with respect to siderite. Under these conditions, siderite does not precipitate or accumulate in bottom sediments.

Upon fall overturn and exposure of bottom sediments to DO, some or all of the newly formed ferrous carbonate (siderite) could be oxidized. Oxidation of siderite proceeds as follows:



This reaction could represent a significant benthic chemical oxygen demand that affects oxygen concentrations in Diascund Creek Reservoir for a period of time after fall overturn.

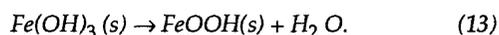
During the reaeration of hypolimnial water at fall overturn, ferrous ions are rapidly oxidized and precipitated, and bicarbonate ions are neutralized. The net reaction is summarized by



For each mole of ferric hydroxide precipitated in this reaction, 2 moles of bicarbonate ions are neutralized by generated hydrogen ions. As noted earlier, overturn of hypolimnial waters enriched with ferrous and bicarbonate ions has little effect on reservoir water quality in the fall and winter because (1) virtually all the ferrous ions are precipitated quickly by this reaction, and (2) ferrous and bicarbonate ions are lost from the water column in the same proportion as they are released from anaerobic bottom sediments.

iments. Consequently, large quantities of iron are internally cycled between bottom sediments and the hypolimnion in Diascund Creek Reservoir, but little of this iron leaves the reservoir. For example, of the 24,000 kg of dissolved iron present in the hypolimnion in early September 1984, 95 percent (23,000 kg) was rapidly lost as ferric hydroxide to bottom sediments soon after fall overturn. Overall, Diascund Creek Reservoir functions as an effective sink for tributary contributions of iron.

Ferric hydroxide is probably converted within the sediment to an amorphous ferric oxyhydroxide complex through the following reaction:



This reaction completes the internal cycle of iron. The newly formed ferric-oxyhydroxide complex in the upper sediment layers is probably the main source of iron released to the anaerobic hypolimnion in the spring and summer months the following year, as depicted in reaction 9.

Overall, the concentration of dissolved iron near the Diascund Creek Reservoir raw-water intake is low, averaging about 0.3 to 0.4 mg/L (table 25). The raw-water intake is set at an altitude of 15 ft above sea level. Although this concentration is near the secondary maximum contaminant level (U.S. Environmental Protection Agency, 1986b), water of this quality is easily treated for domestic uses; however, raw water with an elevated concentration of dissolved iron is periodically withdrawn, thus making water treatment difficult and expensive. During high flows, particularly in the spring months when iron concentrations in tributaries tend to be highest, dissolved-iron concentrations can be as high as 1.0 mg/L near the intake, as they were in the spring and summer of 1984 (fig. 20). Additionally, water containing elevated dissolved-iron concentrations (and manganese as well) is periodically withdrawn during summer months because the raw-water intake is near and sometimes in the upper hypolimnion. The greatest potential for withdrawing hypolimnetic water of poor quality occurs during hot, calm weather when the reservoir is at full-pool elevation.

Manganese reacts similarly to iron in water from Diascund Creek Reservoir. Manganese is relatively insoluble in oxidizing environments and soluble in reducing environments. The difference in

manganese solubility between oxidizing and reducing environments is illustrated in figure 20. The concentration of dissolved manganese exceeds 2 mg/L in the anaerobic hypolimnion of Diascund Creek Reservoir, and approaches 3 mg/L near bottom sediments. Manganous ions released from reducing sediments are the source of this dissolved manganese. The majority (greater than 95 percent) of manganese present in the hypolimnion is dissolved. In contrast, dissolved-manganese concentrations in the aerobic epilimnion of Diascund Creek Reservoir, and throughout the water column during isothermal periods, are generally less than 0.05 mg/L (table 25) and average 0.025 mg/L.

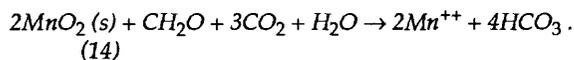
Concentrations of dissolved manganese in water from Diascund Creek Reservoir are an order of magnitude less than dissolved iron in both reducing and oxidizing water (fig. 20). Manganese is present in lower concentrations because (1) tributary inputs of manganese are about an order of magnitude less than those of iron (table 22); and (2) manganese does not readily form organic-metal complexes, which are known to keep dissolved-iron concentrations elevated in oxygenated water (Hem, 1985). In addition, differences in solubility of ferrous and manganous ions in reducing environments and differing solid-phase kinetics of these metals could explain the reduced concentrations of dissolved manganese in the hypolimnion.

Concentrations of manganese in oxygenated water reflect a balance between sources and sinks. Dissolved manganese in oxygenated water of Diascund Creek Reservoir originates mostly from diffusion and advection of manganous ions from the hypolimnion and sediment pore water. The upward movement of manganous ions is probably balanced by settling out of an oxidized manganese precipitate. The retention time of dissolved manganese in oxygenated water of Diascund Creek Reservoir is probably on the order of days or weeks, which reflects the rate of oxidation and precipitation of manganous ions at pH 7.0 (Stumm and Morgan, 1981).

Tributary inflows account for only a small part of the direct source of manganese to oxygenated water in Diascund Creek Reservoir. Without a mechanism to keep manganese in solution (such as the organic-metal complex seen for iron), tributary inputs of manganous ions probably oxidize and settle out soon after entering the reservoir. Even during high-flow periods (such as the first one-half of 1984,

which resulted in an average dissolved-iron concentration of 0.7 mg/L, dissolved-manganese concentrations in oxygenated water remained low, averaging 0.02 mg/L (table 25).

Manganous ions are released from anaerobic bottom sediments as early as April in Diascund Creek Reservoir and continue to be released until fall overturn, following the generalized reaction:



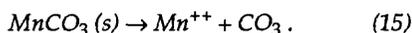
(amorphous (organic matter)

manganese

oxide)

This reaction is generalized because solid-phase manganese oxide present in bottom sediments is not always fully oxidized. The formula for manganese oxide typically ranges from $MnO_{1.3}$ to $MnO_{1.9}$ (Stumm and Morgan, 1981). In reaction 14, solid-phase manganese oxide serves as an electron acceptor during biological oxidation of organic matter. Algal cells and detrital material settling from the epilimnion are the primary source of organic matter driving this reaction. Hydrogen ions consumed during the reaction account for the generation of alkalinity (bicarbonate ions). Each mole of manganous ions released is accompanied by about 2 moles of bicarbonate ions, depending on the average oxidation state of the manganese oxide. This reaction accounts for the accumulation of about 3,000 kg of dissolved manganese in the hypolimnion of Diascund Creek Reservoir by late summer.

The quantity of manganous ions released from anaerobic bottom sediments to the hypolimnion of Diascund Creek Reservoir is probably controlled by a manganous carbonate solid phase (Stumm and Morgan, 1981), such as rhodochrosite, through the following reaction:



WATEQF (Plummer and others, 1976) was used to evaluate the possible effect exerted by rhodochrosite on aqueous species of dissolved manganese and carbonate in pore water of anaerobic bottom sediments. Water collected at a depth of 18 ft on September 4,

1984 (table 24), is assumed to closely reflect the chemistry of anaerobic pore water of Diascund Creek Reservoir.

Water near anaerobic bottom sediments is slightly undersaturated with respect to rhodochrosite, as evidenced by a log (AP/K) value of 0.45; however, as suggested by Paces (1972), log (AP/K) values between -0.5 and 0.5 indicates possible equilibrium with the mineral considered. The slight undersaturation observed could be the result of the use of hypolimnial water (table 24, depth of 18 ft) to approximate the composition of sediment pore water. The log (AP/K) value becomes zero when the concentration of manganous and bicarbonate ions in pore water are 4.0 and 98 mg/L, respectively.

Upon fall overturn and reaeration of bottom waters, manganous ions that accumulate in the hypolimnion are oxidized as follows:



It should be noted that oxidation of manganous ions is not always complete; therefore, reaction 16 is generalized. Moreover, this reaction can be slow and could take days or weeks, depending on the ambient pH. The slow rate of oxidation can result in elevated concentrations of dissolved manganese in the water column after fall overturn. The elevated concentration of dissolved manganese (0.15 mg/L) several weeks after the 1984 fall overturn illustrates the slow oxidation rate of manganous ions in Diascund Creek Reservoir (fig. 20). Oxidation of manganous ions and precipitation of manganese oxide are complete a few months after fall overturn, as evidenced by dissolved-manganese concentrations in water from Diascund Creek Reservoir that average 0.01 mg/L during the winter (table 25).

Overall, during isothermal periods in Diascund Creek Reservoir, the concentration of dissolved manganese near the raw-water intake is less than the secondary maximum contaminant level (U.S. Environmental Protection Agency, 1986b) of 0.05 mg/L (table 25). This level is occasionally exceeded at fall overturn because manganous ions mixed from the hypolimnion take from days to weeks to oxidize. As was previously discussed, water from the upper hypolimnion (or metalimnion) is periodically withdrawn during the summer because of the location of the raw-water intake. Dis-

solved-manganese concentrations in this water can range from 0.1 to 0.5 mg/L near the intake and can cause water-treatment problems.

Nutrients, Chlorophyll-a, and Reservoir Transparency

Nutrient (nitrogen and phosphorus) concentrations in water from Diascund Creek Reservoir are affected by a variety of sources and sinks. The primary nutrient sources are surface-water and ground-water inflows and wet and dry atmospheric deposition directly on the reservoir. Nutrient losses (sinks) include permanent sediment burial, spillway overflow, and withdrawal of raw water for treatment. In addition, nitrogen can be added or removed from the water column through nitrogen fixation by certain bluegreen algae and through denitrification of nitrate and nitrite in anaerobic environments.

Nitrogen and phosphorus are found in many forms in the water column, including particulate matter (living and detrital) and dissolved organic and inorganic forms. In addition, nitrogen is found in several oxidation states, including ammonium, nitrite, and nitrate. These nutrients can change forms and oxidation states rapidly in the water column and bottom sediments. Most of the nutrient transformations are biologically mediated. The most important biological processes affecting nutrient concentrations in water from Diascund Creek Reservoir follow a cycle of (1) algal assimilation of dissolved nutrients; (2) settling of algal cells (or algal-cell derived organic matter); and (3) bacterial mineralization of settled organic matter and subsequent release of dissolved nutrients to the water column.

Differences in the efficiency of this cycle account for the larger N:P ratio in the reservoir compared with tributary inputs. The N:P ratio in tributary inflows is about 7 (table 22); the N:P ratio in Diascund Creek Reservoir is about 24 (table 26). Apparently, much of the phosphorus released from mineralized organic matter is retained in reservoir bottom sediments. On the basis of average concentrations of phosphorus in tributary inflows (0.10 mg/L, table 22) and in the reservoir (0.03 mg/L, table 26), bottom sediments function as a sink for approximately 70 percent of the phosphorus entering Diascund Creek Reservoir. In contrast, nitrogen cycling is more efficient. Most of the nitrogen settling with organic matter is released to the water column as ammonium ions during mineral-

ization. The similarity of tributary and reservoir concentrations of total nitrogen indicates that bottom sediments function as a small sink for nitrogen in Diascund Creek Reservoir.

High concentrations of total phosphorus in water from Diascund Creek Reservoir tributaries indicate that reservoirs in relatively undisturbed watersheds in the Coastal Plain physiographic province can be naturally eutrophic. The U.S. Environmental Protection Agency (1986c) recommends reservoir-inflow concentrations of phosphorus of less than 0.05 mg/L to avoid nuisance algal blooms associated with eutrophication. Phosphatic deposits in the Yorktown Formation (Mixon and others, 1989) are probably the main source of phosphorus to Diascund Creek Reservoir.

Concentrations of nitrogen and phosphorus are similar in water from sampling sites 02042734 and 02042746 (table 26). This similarity probably results from (1) the similarity of nutrient inflows from major tributaries (table 22), (2) similar rates and efficiencies of nutrient cycling in different parts of the reservoir, and (3) horizontal mixing in the reservoir. Chlorophyll-*a* concentrations at the two sites are similar as well. Consequently, discussion of nutrients and chlorophyll-*a* in water from Diascund Creek Reservoir is primarily limited to measurements at sampling site 02042746, which is closer to the raw-water intake (fig. 3).

Seasonal variations in total nitrogen occur primarily in the hypolimnion of the reservoir (fig. 21). High concentrations of total nitrogen occur near reservoir bottom sediments during thermal stratification, commonly exceeding 2.0 mg/L. About one-half of the total nitrogen is in the form of particulate and dissolved organic nitrogen; the remainder is present as dissolved ammonia. Ammonia accumulates in anaerobic bottom water as settled particulate organic matter is mineralized. Summer concentrations of dissolved ammonia averaged 0.63 mg/L at the 18-ft depth (table 26). Consumption of hydrogen ions during the mineralization of organic nitrogen contributes to the accumulation of bicarbonate ions in the hypolimnion of Diascund Creek Reservoir.

Concentrations of total nitrogen in the epilimnion does not differ significantly among seasons; average concentrations range from about 0.6 to 0.8 mg/L (table 26). Slightly higher total-nitrogen concentrations in the summer of 1984 (fig. 21) probably resulted from larger inflow volumes and con-

Table 26.--Average seasonal concentrations of nutrients and chlorophyll-*a*, in water from sampling sites 02042734 and 02042746 in Diascund Creek Reservoir, following period of reservoir drawdown, April 1984 through October 1986

[Concentrations in milligrams per liter, except for chlorophyll-*a*, which is in micrograms per liter; <, less than]

Depth, in feet	Nitrate, plus nitrate, dissolved, as N		Ammonia, dissolved, as N		Ammonia plus organic nitrogen, total, as N		Nitrogen, total, as N		Phosphorus, total, as P		Phosphorus, ortho dissolved as P		Chlorophyll- <i>a</i>	
	02042734	02042746	02042734	02042746	02042734	02042746	02042734	02042746	02042734	02042746	02042734	02042746	02042734	02042746
January-March														
3	<0.1	<0.1	0.05	0.02	0.8	0.7	0.8	0.7	0.032	0.028	<0.01	<0.01	14	18
10	<.1	<.1	.02	.02	.7	.7	.7	.7	.033	.034	<.01	<.01	16	21
18	<.1	<.1	.05	.02	.7	.6	.7	.7	.031	.029	<.01	<.01	28	24
April-June														
3	<.1	<.1	.05	.05	.6	.8	.6	.8	.026	.028	<.01	<.01	7.6	7.4
10	<.1	<.1	.09	.04	.7	.6	.7	.6	.028	.023	<.01	<.012	9.3	9.7
18	<.1	<.1	.26	.23	.7	1.5	.7	1.5	.031	.029	<.01	<.010	6.5	6.5
July-September														
3	<.1	<.1	.04	.03	.6	.7	.6	.7	.029	.028	.010	<.01	13	17
10	<.1	<.1	.05	.04	.6	.6	.6	.6	.029	.028	<.01	<.010	13	14
18	<.1	<.1	.64	.63	1.2	1.5	1.2	1.5	.029	.030	<.01	<.01	17	15
October-December														
3	<.1	<.1	.07	.08	.8	.7	.8	.7	.029	.026	<.01	<.01	15	15
10	<.1	<.1	.04	.05	.8	.5	.8	.5	.030	.028	<.01	<.01	16	21
18	<.1	<.1	.08	.09	.7	.7	.7	.7	.027	.027	<.01	<.01	18	15
Annual average, excluding period of reservoir drawdown														
3	<.1	<.1	.05	.04	.7	.7	.7	.7	.029	.028	<.01	<.01	12	14
10	<.1	<.1	.05	.04	.7	.6	.7	.6	.030	.028	<.01	<.01	14	16
18	<.1	<.1	.26	.24	.8	1.1	.8	1.1	.030	.029	<.01	<.01	17	15

centrations as compared with the summers of 1985-86. Particulate and dissolved organic nitrogen comprise about 90 percent of the total nitrogen in oxygenated waters.

Although averaging only 0.05 mg/L (table 26), dissolved ammonia is probably the primary source of nitrogen for algal growth in the epilimnion of Diascund Creek Reservoir. This pool of ammonia is dynamic and reflects a balance among processes of algal uptake, nitrification (biological conversion of ammonia to nitrite and nitrate in aerobic environments), and replenishment from mineralization of

organic nitrogen in the water column and bottom sediments. Increased ammonia concentrations during reservoir drawdown, which averaged 0.18 mg/L at sampling site 02042746 (table 27), probably reflect a shift in the balance among these processes. For instance, replenishment of water-column ammonia could be more rapid during reservoir drawdown because of reservoir shallowness and lack of a hypolimnion. At full-pool conditions, the highest concentrations of dissolved ammonia in the epilimnion occurred after fall overturn because enriched bottom waters are mixed throughout the water column.

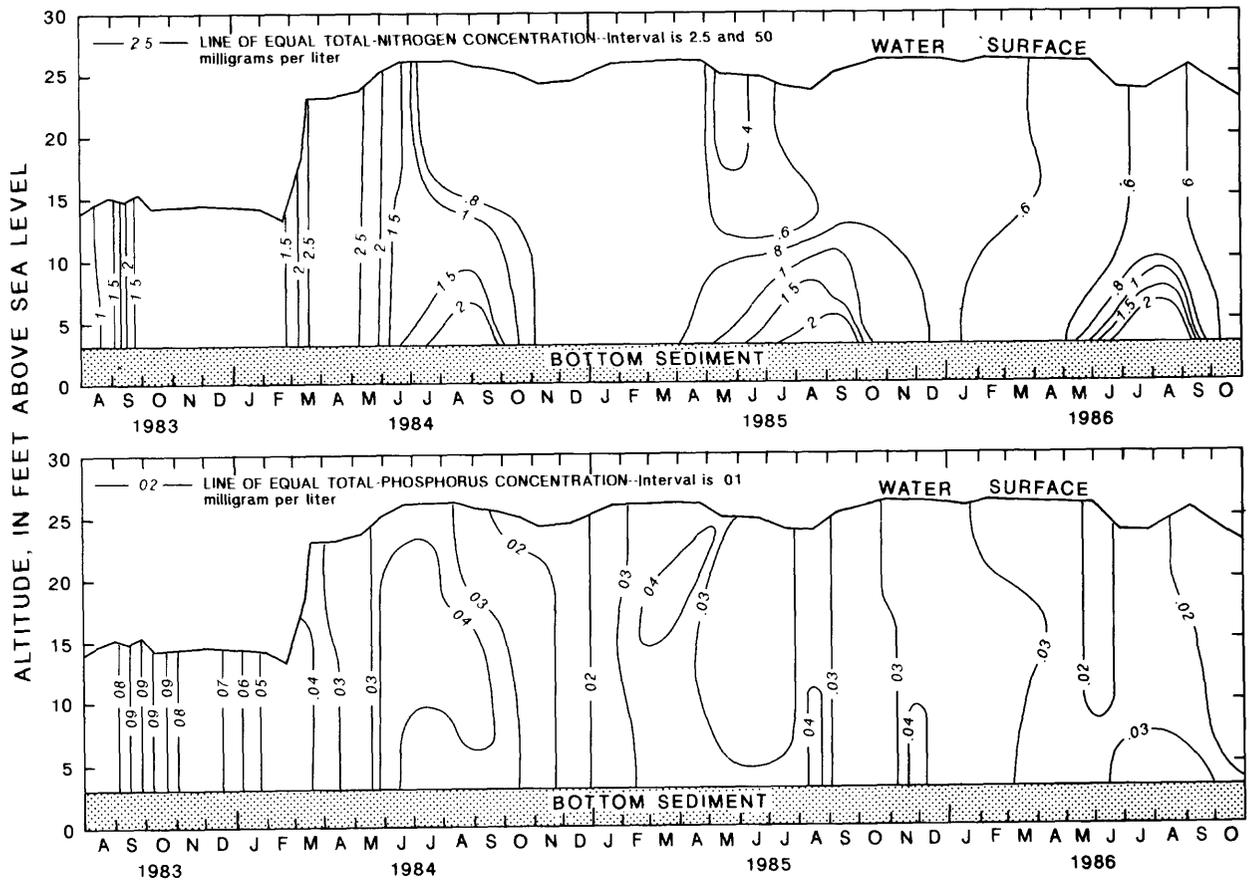


Figure 21.--Total-nitrogen and total-phosphorus concentrations in water from Diascund Creek Reservoir, 1983-86.

Table 27.--Average seasonal concentrations of nutrients and chlorophyll-*a* in water from sampling sites 02042734 and 02042746 in Diascund Creek Reservoir during period of reservoir drawdown, August 1983 through March 1984

[Concentrations in milligrams per liter, except for chlorophyll-*a*, which is in micrograms per liter; <, less than; no data available]

Depth in feet	Nitrite, plus nitrate, dissolved as N		Ammonia, dissolved as N		Ammonia plus organic nitrogen, total, as N		Nitrogen, total as N		Phosphorus, total as P		Phosphorus, ortho dissolved, as P		Chlorophyll- <i>a</i>	
	02042734	02042746	02042734	02042746	02042734	02042746	02042734	02042746	02042734	02042746	02042734	02042746	02042734	02042746
3	<0.1	0.11	0.15	0.15	-	1.1	-	1.2	0.094	0.100	<0.01	<0.01	91	83
10	<1	<1	.26	.21	-	1.2	.078	<.076	<.01	.076	<.01	<.01	68	65

Total-phosphorus and dissolved-orthophosphorus concentrations show little seasonal variability in water from Diascund Creek Reservoir. Total phosphorus averaged 0.029 mg/L for April 1984 through October 1986, differing by less than 10 percent among seasonal averages (table 26). Moreover, the variability among samplings was also small; concentrations typically ranged from 0.02 to 0.04 mg/L at full pool (fig. 21). As with total nitrogen, total-phosphorus concentrations were slightly higher in the summer of 1984 (fig. 21) because of larger inflow volumes and concentrations, compared with the summers of 1985-86. Dissolved-orthophosphorus concentrations were less than or equal to 0.01 mg/L for all seasons (table 26).

Concentrations of total phosphorus and dissolved orthophosphorus did not differ appreciably with depth in Diascund Creek Reservoir (table 26). The lack of vertical gradients was expected during isothermal periods because the reservoir is well mixed and bottom sediments are exposed to oxygen. Typically, phosphorus in oxidizing sediments is not released to the water column because it is tightly bound to ferric oxyhydroxide coatings on sediment grains. The lack of vertical phosphorus gradients during summer stratification in Diascund Creek Reservoir was unexpected, however. Typically, phosphorus concentrations in an anaerobic hypolimnion exceed epilimnion concentrations by one to two orders of magnitude as phosphorus is released from reducing bottom sediments (Wetzel, 1975). This release is attributed to the decreasing number and strength of iron-phosphate binding sites as ferric oxyhydroxide coatings are reduced to gel-like hydrated ferrous oxide or ferrous hydroxide (Patrick and Khalid, 1974). This "classical" phosphorus release mechanism, as first described by Mortimer in 1941 and 1942, does not operate in Diascund Creek Reservoir, because bottom sediments in this reservoir apparently function as a sink for phosphorus under all natural environmental conditions.

The lack of a net phosphorus release from anaerobic bottom sediments in Diascund Creek Reservoir could result from (1) an inadequate supply of phosphorus in sediments, (2) sediment phosphorus being in a form unavailable for release, and (3) a solid-phase equilibrium effect on dissolved phosphorus that keeps pore-water concentrations low. A phosphorus shortage in these bottom sediments seems unlikely, considering that about 70 percent of the phosphorus load carried by naturally enriched tributaries is deposited in the reservoir; however,

phosphorus in these bottom sediments could be retained in refractory, redox-insensitive, high-molecular-weight organic compounds, similar to those described by Schindler and others (1977) for Experimental Lake 227 in Canada. Formation of these compounds is thought to be mediated by bacterial uptake. Pore-water gradients of dissolved phosphorus in Lake 227 are absent or weak, and concentrations are low under both aerobic and anaerobic conditions.

Concentrations of dissolved orthophosphorus in pore water could be low because of equilibrium reactions with sediment solid phases, such as clay, ferric and ferrous complexes, or iron-phosphate minerals. Clay, particularly montmorillonite, has a large binding capacity for phosphate (Golterman, 1973). The binding capacity is pH-dependent and is strongest when pH is low. Apparently, hydroxide and phosphate ions compete for the same binding sites on clay minerals. At the pH values expected in pore water of Diascund Creek Reservoir sediments (6.5 to 6.9), about 300 mg of phosphate (as P) binds to 100 g of montmorillonite (Golterman, 1973). This binding is insensitive to redox conditions. Consequently, bottom-sediment clay could function as a large phosphate "buffer" that keeps pore-water concentrations low under nearly all environmental conditions. This effect on dissolved-phosphorus concentration in pore water is reasonable for Diascund Creek Reservoir because the Calvert Formation, which is the source of the clay in these bottom sediments, is predominantly montmorillonite, with small amounts of iron-rich illite.

Low dissolved-phosphorus concentrations in pore water could also be related to the iron-rich bottom sediments and a relatively small phosphorus-to-iron ratio. The bottom sediments of Diascund Creek Reservoir are undoubtedly iron-rich, as evidenced by bright orange soils in the area. As shown by Olsen (1964) and Patrick and Khalid (1974), the exchange of phosphorus on iron-inorganic complexes follows an adsorption isotherm in which the pore-water concentration is related to the ratio of bound phosphate molecules to the number of available binding sites.

In other words, pore-water concentrations of dissolved phosphorus tend to be lower in sediments with a large surface area of iron-inorganic phosphate binding sites. This phenomenon, noted by Stauffer (1981), could explain why certain lakes with

iron-rich sediments (like Diascund Creek and Little Creek Reservoirs) have less internal loading of phosphorus than lakes with iron-poor sediments.

Because of the elevated concentration of dissolved iron in sediment pore water, the dissolved-orthophosphorus concentration in pore water can be affected by an iron phosphate mineral, such as vivianite [$\text{Fe}_3(\text{PO}_4)_2 \cdot 8\text{H}_2\text{O}$]. Such an effect has been described by Emerson and Widmer (1978) and Nriagu and Dell (1974) in other lakes. The presence of vivianite in lake sediments examined by Tesse-now (1974) and Williams and others (1971) provides further evidence that dissolved phosphorus in pore water could be in equilibrium with this mineral.

WATEQF (Plummer and others, 1976) was used to evaluate the possible effect exerted by vivianite on the concentration of aqueous phosphorus in pore water. Water collected from a depth of 18 ft on September 4, 1984 (table 24), was assumed to closely reflect the pore-water chemistry in anaerobic bottom sediments. Dissolved-orthophosphorus concentrations ranging from 0.01 to 0.05 mg/L, rather than a single concentration, were evaluated. This range probably includes the actual dissolved-orthophosphorus concentration in pore water of Diascund Creek Reservoir. Hypolimnetic water (and presumably pore water) in Diascund Creek Reservoir appears to be saturated with respect to vivianite, as evidenced by log (AP/K) values ranging from -0.52 to 0.87 as the concentration of dissolved orthophosphorus ranged from 0.01 to 0.05 mg/L as P. Log (AP/K) is approximately zero when the orthophosphorus concentration is 0.02 mg/L.

Regardless of whether orthophosphorus concentrations in pore water are affected by a refractory organic molecule, sorption to montmorillonite or a iron-inorganic complex, or equilibrium with an iron phosphate mineral, the bottom sediments of Diascund Creek Reservoir are an excellent sink for phosphorus under all natural environmental conditions. Moreover, because this reservoir has been in existence for more than 20 years, it is reasonable to assume that the phosphorus-retention capacity of these bottom sediments will not decrease measurably in the near future.

The elevation of the reservoir pool has a major effect on the total-phosphorus concentration in water from Diascund Creek Reservoir. Total phosphorus averaged about 0.09 mg/L in water from sampling site 02042746 during reservoir drawdown

(table 27), three times the mean concentration (0.028 mg/L) at full-pool conditions (table 26). On the basis of these mean total-phosphorus concentrations, Diascund Creek Reservoir would be classified as hypereutrophic during reservoir drawdown, and mesotrophic/eutrophic at full pool (Carlson, 1977).

Increased total-phosphorus concentrations during drawdown can result from many factors, including (1) reduced settling time for tributary inflows of phosphorus because of a shortened hydraulic retention time; (2) increased inflows of phosphorus-enriched ground water because of a steeper hydraulic gradient in the surrounding aquifer; (3) the elimination of the hypolimnion, which sometimes serves as an effective phosphorus trap when the reservoir is at full pool; (4) increased direct uptake of phosphorus from bottom sediments by algae because of the reservoir shallowness; and (5) increased exposure of phosphorus-rich, fine-grained bottom sediments to resuspending forces, such as wave action, wind-induced mixing, and fish activity (Serruya, 1977; Ryding and Forsberg, 1977). Resuspension of bottom sediments can be the most important factor contributing to increased total-phosphorus concentrations during reservoir drawdown. Even moderate currents and wave action can readily suspend fine-grained bottom sediments. Phosphorus in resuspended sediment can be assimilated by algae in the euphotic zone (Williams and others, 1980; Golterman, 1977), resulting in increased concentrations of chlorophyll-*a* (Ryding and Forsberg, 1977).

Algal biomass in Diascund Creek Reservoir is probably phosphorus-limited, as indicated by an N:P ratio (by weight) in excess of 7 (table 22). The N:P ratio during full pool conditions averaged 25 (table 26). The ratio of the forms of nitrogen that are readily available to algae (dissolved nitrite, nitrate, and ammonia) to a readily available form of phosphorus (dissolved orthophosphorus) averaged 10.

Although algae in Diascund Creek Reservoir are potentially phosphorus-limited, a correlation between total-phosphorus and chlorophyll-*a* concentrations (at full pool) is not statistically significant ($p > 0.05$). The lack of correlation is probably related to the lack of variation in total-phosphorus concentrations among seasons (table 26) and among years during full pool conditions (figs. 21 and 22). Most of the variability in chlorophyll-*a* is related to factors other than the concentration of the limiting nutrient, factors such as water temperature, light

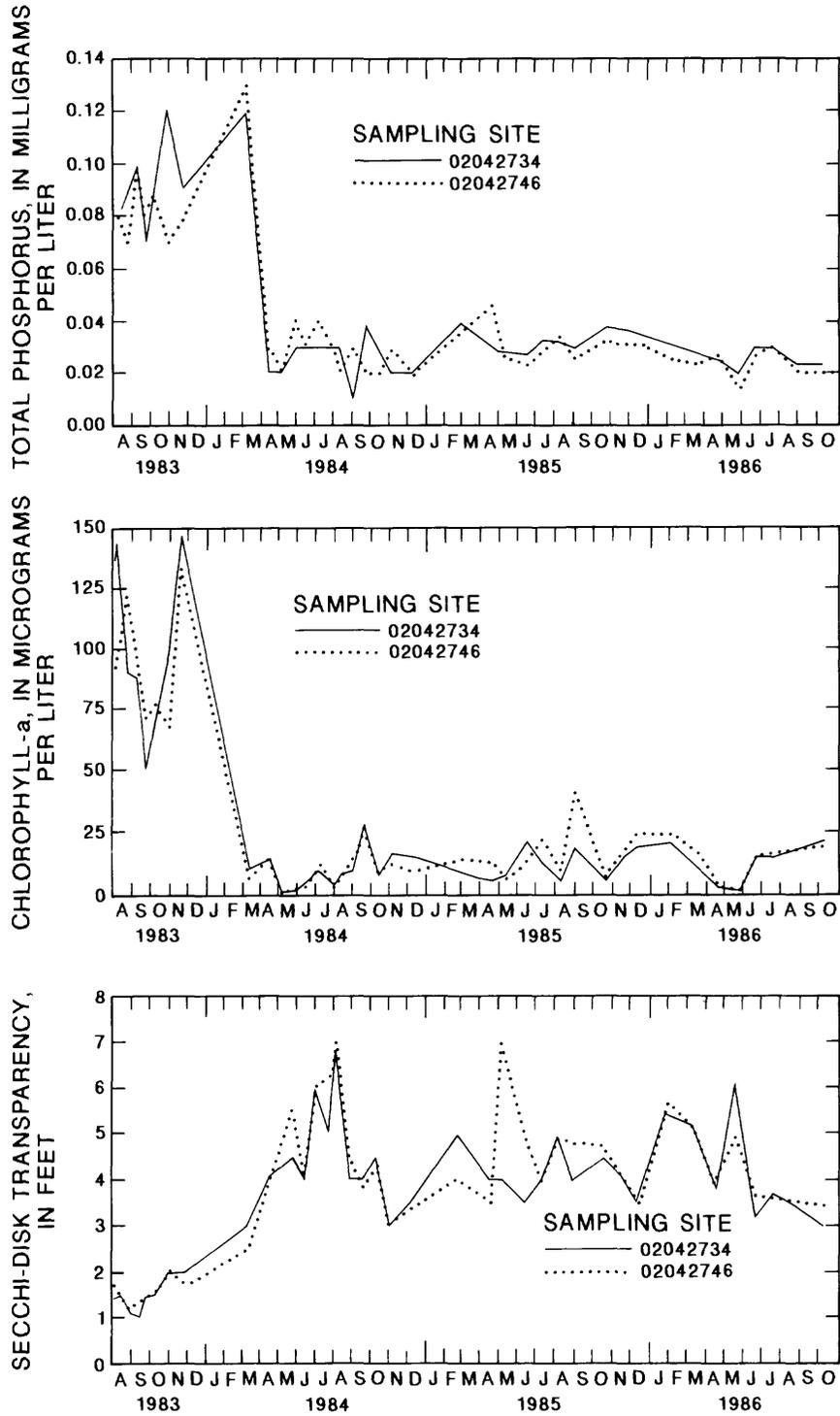


Figure 22.—Total-phosphorus and chlorophyll-a concentrations at a depth of 3 feet and Secchi-disk transparency depth in water from Diascund Creek Reservoir, 1983-86.

intensity, reservoir-mixing patterns, zooplankton grazing, and natural succession of algal populations. Variations in hydraulic retention time could also explain some of the variability in chlorophyll-*a* concentrations resulting from washout of algal cells. For instance, high flows from March through June 1984 could have been partly responsible for low chlorophyll-*a* concentrations in the early summer of that year (fig. 23). Because the hydraulic retention time of Diascund Creek Reservoir averages about 130 days, however, algal washout undoubtedly plays a much smaller role in affecting chlorophyll-*a* concentrations than is observed in Chickahominy Reservoir.

Evidence for phosphorus as the limiting nutrient in Diascund Creek Reservoir can be seen in figure 22, where chlorophyll-*a* concentrations are compared for the period of reservoir drawdown (August 1983 through March 1984) and the period of full pool operation (April 1984 through October 1986). High chlorophyll-*a* concentrations, which averaged 83 $\mu\text{g}/\text{L}$ in water at a depth of 3 ft (sampling site 02042746, table 27) during reservoir drawdown, mostly resulted from increased availability of phosphorus, the nutrient-limiting algal biomass (fig. 22). Phosphorus was potentially limiting during reservoir drawdown, as evidenced by a mean N:P ratio of 12 (table 27). Chlorophyll-*a* concentrations averaged only 14 $\mu\text{g}/\text{L}$ at a depth of 3 ft after the reservoir was refilled in March 1984 (table 26), presumably because of a substantial decrease in total-phosphorus concentrations. Although the increased concentrations of chlorophyll-*a* during drawdown cannot be linked conclusively to the increased concentrations of total phosphorus, it is certain that water quality is severely degraded in Diascund Creek Reservoir when operated at low pool.

During full pool, mean concentrations of chlorophyll-*a* in water from Diascund Creek Reservoir (sampling site 02042746) ranged from 6.5 to 24 $\mu\text{g}/\text{L}$ and averaged 15 $\mu\text{g}/\text{L}$ (table 26 and fig. 23). Although concentrations at this site did not differ markedly between seasons, short-term changes in chlorophyll-*a* concentrations can be large. Concentrations measured on individual days ranged from 0.5 to 40 $\mu\text{g}/\text{L}$ at a depth of 3 ft (fig. 22). The lowest concentrations measured in water from Diascund Creek Reservoir occurred in spring months, commonly in late April and May. The May concentration of chlorophyll-*a* averaged less than 3 $\mu\text{g}/\text{L}$. The rapid decline of chlorophyll-*a* in the spring corre-

sponds to the onset of thermal stratification and the subsequent settling of planktonic algae (primarily diatoms) from the epilimnion. Planktonic diatoms, which dominate the algal community in the winter and early spring, require cool (more viscous), well-mixed water to remain suspended in the water column. Loss of particulate phosphorus with settling diatom cells explains the slight decrease in total-phosphorus concentrations each May (fig. 22). After thermal stratification, green and blue-green algae begin to thrive, dominating the algal community by June. These algae, either through motility or buoyancy, remain suspended in the epilimnion even during periods of negligible mixing.

The vertical distribution of chlorophyll-*a* is fairly constant during isothermal periods, as illustrated by the nearly vertical lines of equal chlorophyll-*a* concentration from October through March in figure 23; however, the vertical distribution of chlorophyll-*a* differed considerably among years during thermal stratification. In 1984, chlorophyll-*a* concentrations changed little with water depth. Concentrations measured in the summer of 1985 decreased with depth; concentrations increased with depth in 1986, exceeding 50 $\mu\text{g}/\text{L}$ near the sediment-water interface. The reason for differences among years in the vertical distribution of chlorophyll-*a* during stratification is unclear.

In general, the transparency of Diascund Creek Reservoir is inversely correlated with chlorophyll-*a* concentration. High concentrations of chlorophyll-*a* at sampling site 02042746 during drawdown resulted in Secchi-disk transparency depths that ranged from 1.2 to 2.0 ft and averaged 1.6 ft in 1983 (fig. 22). During full-pool operation, when chlorophyll-*a* concentrations were lower, the Secchi-disk transparency depth ranged from 3.0 to 7.0 ft and averaged 4.6 ft. The relation between Secchi-disk transparency depth and chlorophyll-*a* concentration in Diascund Creek Reservoir is partly obscured, however, by nonalgal particulates and dissolved organic matter in the water column that also scatter and absorb light.

Little Creek Reservoir

The limnology of Little Creek Reservoir is similar to that of Diascund Creek Reservoir. The cycling of constituents within the water column, and between the water column and bottom sediments, is affected by many of the same physical, chemical, and biological processes. The quality of water in Lit-

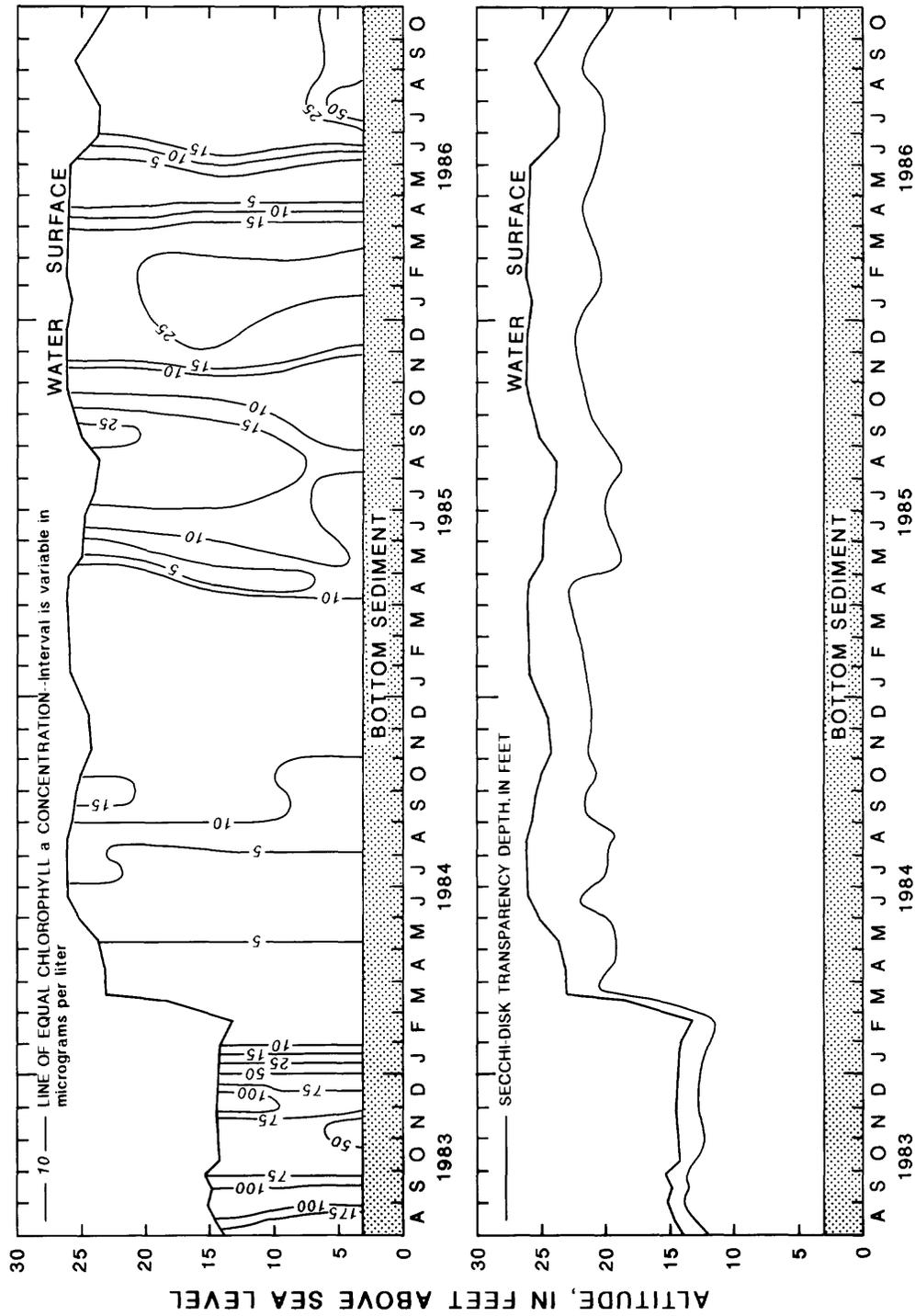


Figure 23.--Chlorophyll-a concentrations and Secchi-disk transparency depth in water from Diascund Creek Reservoir, 1983-86.

tle Creek Reservoir and Diascund Creek Reservoir is not the same, however, mostly because these reservoirs differ in external sources of constituents, hydraulic retention times, and depths.

The quality of the upper 10 ft of water at the five Little Creek Reservoir sampling sites is summarized in table 28. These data are compiled for August 1983 through December 1984, when these sites were sampled synoptically. For the most part, water-quality differences between site pairs are small and not statistically significant (Wilcoxon matched-pairs signed-ranks test, *p* values greater than 0.05; SPSS, Inc., 1988).

Specific Conductance

The vertical and horizontal distribution of specific conductance in Little Creek Reservoir is primarily affected by the water quality of inflows from Chickahominy and Diascund Creek Reservoirs, patterns of horizontal mixing, and cycling of constituents between bottom sediments and the hypolimnion. The increased specific conductance of hypolimnial water in Little Creek Reservoir results from the release of ferrous, manganous, ammonium, and bicarbonate ions from anaerobic bottom sediments. This release begins around June and contin-

ues until fall overturn in November (fig. 24). Specific conductance occasionally exceeds 200 $\mu\text{S}/\text{cm}$ near the sediment-water interface in October and November.

Ions released from anaerobic bottom sediments have little effect on the specific conductance of oxygenated water in Little Creek Reservoir. When released ions are exposed to oxidizing conditions after diffusing through the metalimnion or during fall overturn, they rapidly precipitate in aerobic environments.

Mean specific conductance of the upper 10 ft of water in Little Creek Reservoir differed less than 4 percent among the five sites sampled between August 1983 and December 1984 (table 28). Sampling sites 0204275430 and 0204275470, which were sampled during the entire study period, differed in mean specific conductance by less than 2 percent (table 29). None of the differences in specific conductance among the five stations were statistically significant (*p* > 0.05) according to results of the Wilcoxon matched-pairs signed-ranks test (SPSS, Inc., 1988). In addition, seasonal differences were less than 5 percent between sampling sites 0204275430 and 0204275470 (table 30). Presumably, daily inflows to Little Creek Reservoir are small compared with the reservoir volume, and because horizontal

Table 28.--Mean constituent values and measurements in water from the upper 10 feet at five sampling sites in Little Creek Reservoir, August 1983 through December 1984

[$\mu\text{S}/\text{cm}$, microsiemens per centimeter at 25 degrees Celsius; $\mu\text{g}/\text{L}$, micrograms per liter; mg/L , milligrams per liter; <, less than]

Constituent or property	Station number				
	0204275420	0204275430	0204275440	0204275470	0204275490
Water temperature, degrees Celsius	20.0	20.0	20.5	20.5	20.5
Specific conductance, $\mu\text{S}/\text{cm}$	106.0	102.0	103.0	106.0	105.0
pH	6.9	7.1	7.0	7.0	7.0
Oxygen, dissolved, mg/L	8.2	8.2	8.2	8.3	8.1
Secchi-disk transparency depth, feet	6.8	7.0	6.2	7.2	7.8
Chlorophyll- <i>a</i> , $\mu\text{g}/\text{L}$	9.8	8.4	11.9	7.7	8.1
Nitrite + nitrate, dissolved, mg/L as N	<.1	<.1	<.1	<.1	<.1
Ammonia, dissolved, mg/L as N	.08	.06	.05	.06	.08
Phosphorus, total, mg/L as P	.011	.015	.010	<.01	.011
Phosphorus, ortho dissolved, mg/L as P	<.01	<.01	<.01	<.01	<.01

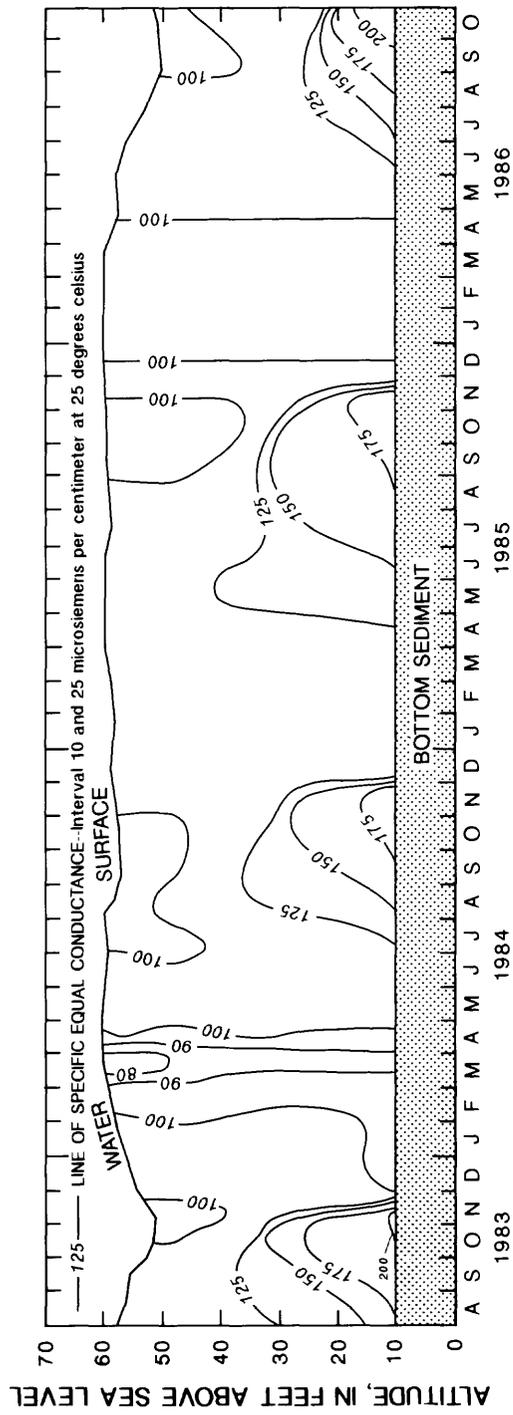


Figure 24.--Specific conductance in water from Little Creek Reservoir, 1983-86.

Table 29.--Mean constituent values and measurements in water from the upper 10 feet of water at sampling sites 0204275430 and 0204275470 in Little Creek Reservoir, 1983-86

[μ S/cm, microsiemens per centimeter at 25 degrees Celsius; μ g/L, micrograms per liter; mg/L, milligrams per liter; <, less than]

Constituent or property	Station number	
	0204275430	0204275470
Water temperature, degrees Celsius	20.0	20.0
Specific conductance, μ S/cm	103.0	105.0
pH	7.0	7.1
Oxygen, dissolved, mg/L	8.6	8.7
Secchi disk transparency, feet	8.4	8.9
Chlorophyll- <i>a</i> , μ g/L	7.9	7.4
Nitrite + nitrate, dissolved, mg/L as N	<.1	<.1
Ammonia, dissolved, mg/L as N	.05	.05
Phosphorus, total, mg/L as P	.014	.010
Phosphorus, ortho dissolved, mg/L as P	<.01	<.01

mixing is relatively rapid, the specific conductance of the upper 10 ft of water is spatially and temporally uniform.

The specific conductance of the upper 10 ft of water in Little Creek Reservoir represents a composite of the inflows from Chickahominy and Diascund Creek Reservoirs, surface-water and ground-water inflows from the drainage basin, and precipitation. By knowing the volume and quality of each inflow, and accounting for reservoir evaporation, one can determine if specific conductance (or any constituent in water) is conservative (nonreactive) in Little Creek Reservoir. If Little Creek Reservoir is not a net sink or source for a constituent, its mean water-column concentration should approximately equal the volume-weighted concentration of the inflows; if the reservoir is a net sink for a constituent, the water-column concentration will be less than the volume-weighted concentration of the inflows; if the reservoir is a net source of a constituent, the water-column concentration will be greater than the volume-weighted concentration of the inflows.

Volume-weighted inflows of specific conductance to Little Creek Reservoir (and several constituents discussed later) are calculated by use of the following mass-balance equation:

$$\frac{\sum (C_{li} * C_{Ci}) + \sum (D_{li} * D_{Ci}) + (T_{I} * T_{C}) + (P_{I} * P_{C})}{C_{I} + D_{I} + T_{I} + P_{I} - E}, \quad (7)$$

where

C_{li} is inflow volume from Chickahominy Reservoir during month i , in millions of gallons;

C_{Ci} is the specific conductance measured in Chickahominy Reservoir (sampling site 02042720) during or around month i ;

D_{li} is inflow volume from Diascund Creek Reservoir during month i , in million of gallons;

D_{Ci} is the specific conductance measured in Diascund Creek Reservoir (depth 10 ft at sampling site 02042746) during or around month i ;

T_{I} is total volume of terrestrial inflows, in millions of gallons;

T_{C} is average specific conductance of terrestrial inflows;

P_{I} is total volume of precipitation on Little Creek Reservoir, in millions of gallons;

P_{C} is average concentration or value of precipitation;

C_{I} is total inflow volume from Chickahominy Reservoir, in millions of gallons;

D_{I} is total inflow volume from Diascund Creek Reservoir, in millions of gallons;

E is total volume evaporated from Little Creek Reservoir, in millions of gallons.

Monthly inflows to Little Creek Reservoir from Chickahominy and Diascund Creek Reservoirs are listed in table 5. Volumes of terrestrial inflows, precipitation, and evaporative losses used in equation 17 are also listed in table 5. Because the quality of terrestrial inflows was not measured, it is assumed to equal the volume-weighted mean of inflows from Chickahominy and Diascund Creek Reservoirs.

Table 30.—Average seasonal specific conductance, pH, water temperature, and dissolved-oxygen concentration in water from sampling sites 0204275430 and 0204275470 in Little Creek Reservoir, 1983-86

[Sampling sites shown in figure 3]

Depth, in feet	Water temperature, in degrees Celsius		Dissolved oxygen, in milligrams per liter		Dissolved oxygen, in percent of saturation		Specific conductance, in microsiemens per centimeter		pH	
	0204275430	0204275470	0204275430	0204275470	0204275430	0204275470	0204275430	0204275470	0204275430	0204275470
January-March										
10	6.5	6.5	11.5	11.0	94	89	96	102	6.9	7.0
Bottom ^a	5.5	6.0	11.1	10.4	89	83	99	99	7.0	7.0
April-June										
10	20.5	20.5	9.0	9.0	101	100	105	109	7.0	7.0
Bottom ^a	10.0	9.5	4.2	4.2	35	37	119	116	6.5	6.6
July-September										
10	26.5	26.5	7.7	8.0	97	100	105	105	7.1	7.0
Bottom ^a	11.0	11.0	0.0	0.0	0	0	159	145	6.6	6.6
October-December										
10	15.5	16.0	7.9	8.0	79	80	103	106	7.0	7.0
Bottom ^a	11.5	11.0	3.4	2.0	32	18	152	154	6.7	6.8
Annual Average										
10	17.5	17.5	9.0	9.0	93	92	102	106	7.0	7.0
Bottom ^a	9.5	9.5	4.6	4.2	39	34	132	128	6.7	6.7

^a Measurements made at a depth of 40 feet at station 0204275430 and at a depth of 50 feet at station 0204275470.

Estimates of precipitation quality were obtained from published literature. The volume-weighted specific conductance of precipitation in this area of Virginia averages about 20 $\mu\text{S}/\text{cm}$ (Gambell and Fisher, 1966).

Specific conductance appears to be conservative in the upper 10 ft of water in Little Creek Reservoir. The mean specific conductance of the upper 10 ft of water at sampling site 0204275430 is 102 $\mu\text{S}/\text{cm}$ (table 30), which is almost identical to the predicted value of 100 $\mu\text{S}/\text{cm}$ obtained from equation 17 (fig. 25). Specific conductance is conservative in the oxygenated water because most of the charge-carrying ions entering the reservoir are conservative, and because the relation between ion concentrations and specific conductance is nearly linear in dilute waters (Hem, 1985).

pH and Water Temperature

The pH of the upper 10 ft of water of Little Creek Reservoir typically ranges from 6.5 to 7.5 and is relatively constant during the year. During isothermal conditions, such as January through March, pH is constant with depth and averages about 7.0 (table 30); however, when Little Creek Reservoir is stratified (from April through November), the hypolimnial pH averages 6.5 to 6.6, which is 0.4 to 0.5 standard units lower than that of the upper 10 ft of water. As in Diascund Creek Reservoir, the pH of the Little Creek Reservoir hypolimnion is rarely below 6.5. The hypolimnion pH represents a balance between acidity contributed by carbon dioxide from decomposing organic matter and alkalinity generated when organic matter is decomposed in an anaerobic environment.

The pH of the upper 10 ft of water periodically exceeds 7.5. The pH increases when uptake of carbon dioxide by photosynthesizing algae temporarily exceeds the rate of atmospheric infusion of carbon dioxide. The highest pH and the largest diurnal changes in pH probably occur on calm days when primary productivity is high.

Similar to Diascund Creek Reservoir, Little Creek Reservoir is a monomictic system (that is, it undergoes one primary period of thermal stratification). Stratification typically begins in April and continues until fall overturn in November (fig. 26). Fall overturn occurs about 1 month later than in Diascund Creek Reservoir because Little Creek Reservoir is deeper and its hypolimnion is 5 to 10 °C cooler by early fall. Consequently, the period of epilimnion cooling to reach isothermal conditions is longer for Little Creek Reservoir. Isothermal conditions generally persist from December through March, as illustrated by nearly vertical lines of equal temperature in figure 26; however, ice cover does form during some years and leads to short-term periods of slight thermal stratification. This is seen as 0 °C temperatures near the reservoir surface in January 1984 and February 1985. Compared with Diascund Creek Reservoir, ice covers are thinner and develop more slowly on Little Creek Reservoir because of its greater depth and heat content.

The annual maximum water temperature in Little Creek Reservoir typically occurs in July or August, periodically reaching 30 °C. The mean temperature of the upper 10 ft of water in the summer is 26.5 °C (table 30). Hypolimnion water temperatures in the summer range from 8 to 12 °C near the reservoir bottom (fig. 26), increasing slightly during the periods of stratification. The minimum annual water temperature in Little Creek Reservoir typically occurs in January and ranges from 0 to 4 °C. The water temperature averages 6 °C throughout the water column during the winter.

The progression of thermal stratification and destratification (fall overturn) is similar throughout Little Creek Reservoir. Vertical profiles of temperature are comparable at all sampling sites for the four dates illustrated in figure 27. For instance, on August 15, 1984, the epilimnion thickness was about 10 ft at all sampling sites; water temperatures near the reservoir surface and at a depth of 28 ft were about 30 and 12.5 °C, respectively (fig. 27). On November 18, 1984, during fall overturn, the epilimnion thickness was about 30 ft (reaching the bottom

at sampling sites 0204275420 and 0204275440), and water temperatures near the reservoir surface were about 16 °C at all sampling sites (fig. 27). Overall, the factors affecting heating, cooling, and mixing of water in Little Creek Reservoir have a nearly uniform effect throughout the reservoir.

Withdrawals from Little Creek Reservoir affect the summer thermal-density layering at sampling site 0204275430, which is near the pump-station intake. When withdrawals are small, such as in 1985 (table 5), thermal gradients tend to be steep, as illustrated by the closely spaced equal-temperature lines in figure 26. In contrast, when summer withdrawals are large, such as in 1986 (table 5), the thermal gradients are less steep, as illustrated by the widely spaced equal-temperature lines in figure 26. Vertical mixing in the water column associated with withdrawal-induced currents accounts for the less steep thermal gradients.

Dissolved Oxygen

The vertical distribution of DO in Little Creek Reservoir is affected by mixing patterns and thermal stratification. During isothermal periods, when mixing keeps the water column in contact with atmospheric oxygen, Little Creek Reservoir is nearly saturated with DO (averaging greater than 90 percent). Reservoir mixing also keeps DO concentrations uniform with depth, as illustrated by the nearly vertical lines of equal DO between December and March in figure 26. DO concentrations typically range from 8 to 14 mg/L during isothermal periods. During thermal stratification, the upper 10 ft of water in Little Creek Reservoir is also nearly saturated with DO because of contact with the atmosphere. Because this upper water is warmer, however, summer DO concentrations (typically 6 to 10 mg/L, fig. 26) are lower than those during isothermal periods. Water in the Little Creek Reservoir is periodically supersaturated with DO during the summer months, particularly when algal productivity is high and winds are calm. For instance, DO saturation in the upper 10 ft of water was 150 percent on August 12, 1985, averaging 11.7 mg/L of DO.

Hypolimnial DO in Little Creek Reservoir is depleted by decomposition of settling particulate organic matter and by benthic oxygen demand. Epilimnion sources of organic matter include inflows from Chickahominy and Diascund Creek Reservoirs and in-lake primary and secondary production of biomass. Depletion of hypolimnial oxy-

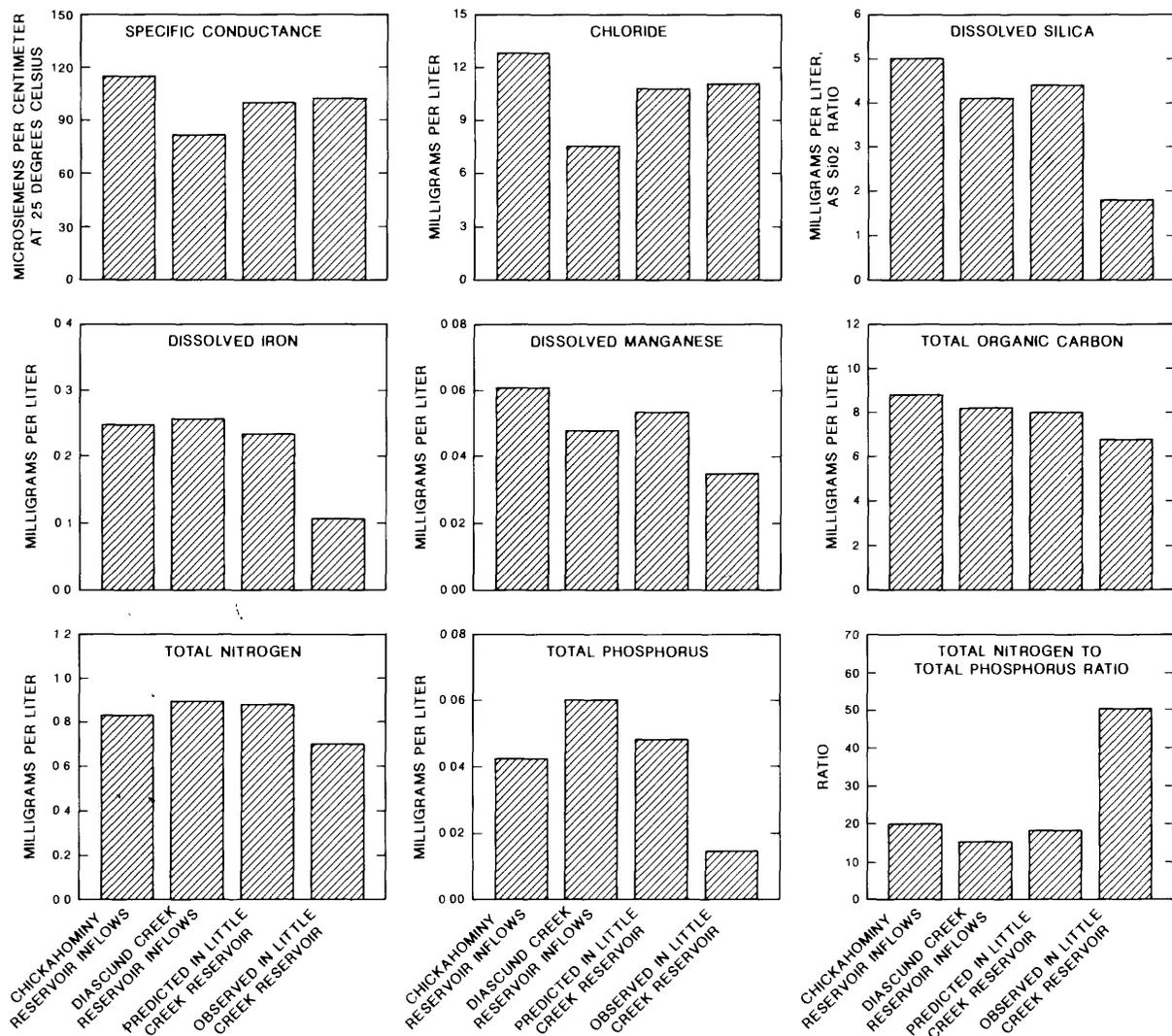


Figure 25.—Mean constituent values in water from the upper 10 feet of Chickahominy, Diascund Creek, and Little Creek Reservoirs, and predicted values from equation 17 for Little Creek Reservoir, 1983-86.

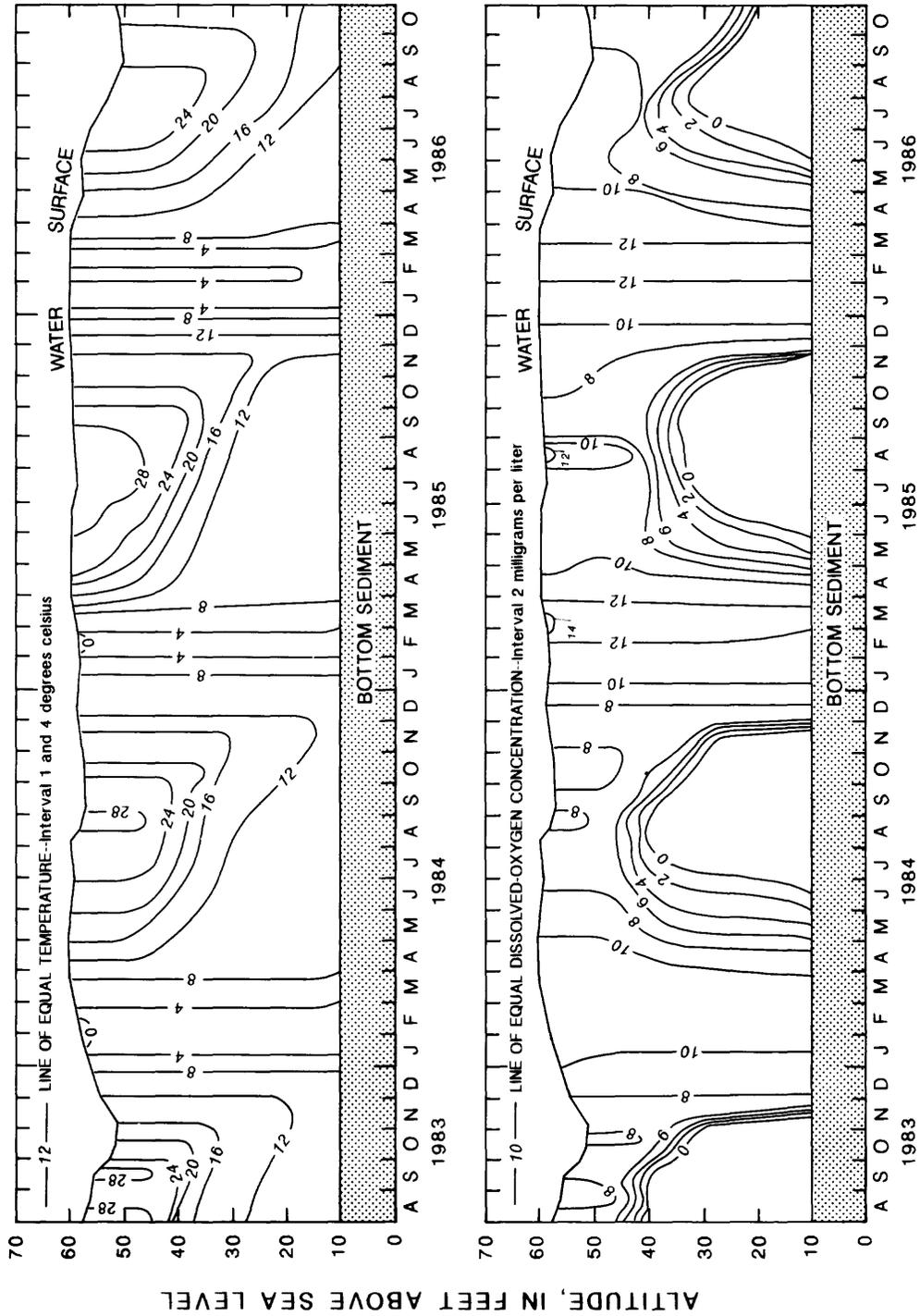


Figure 26.--Average seasonal concentrations of nutrients and chlorophyll-*a* in water from sampling sites and in Diascund Creek Reservoir following period of reservoir drawdown, April 1984 through October 1986.

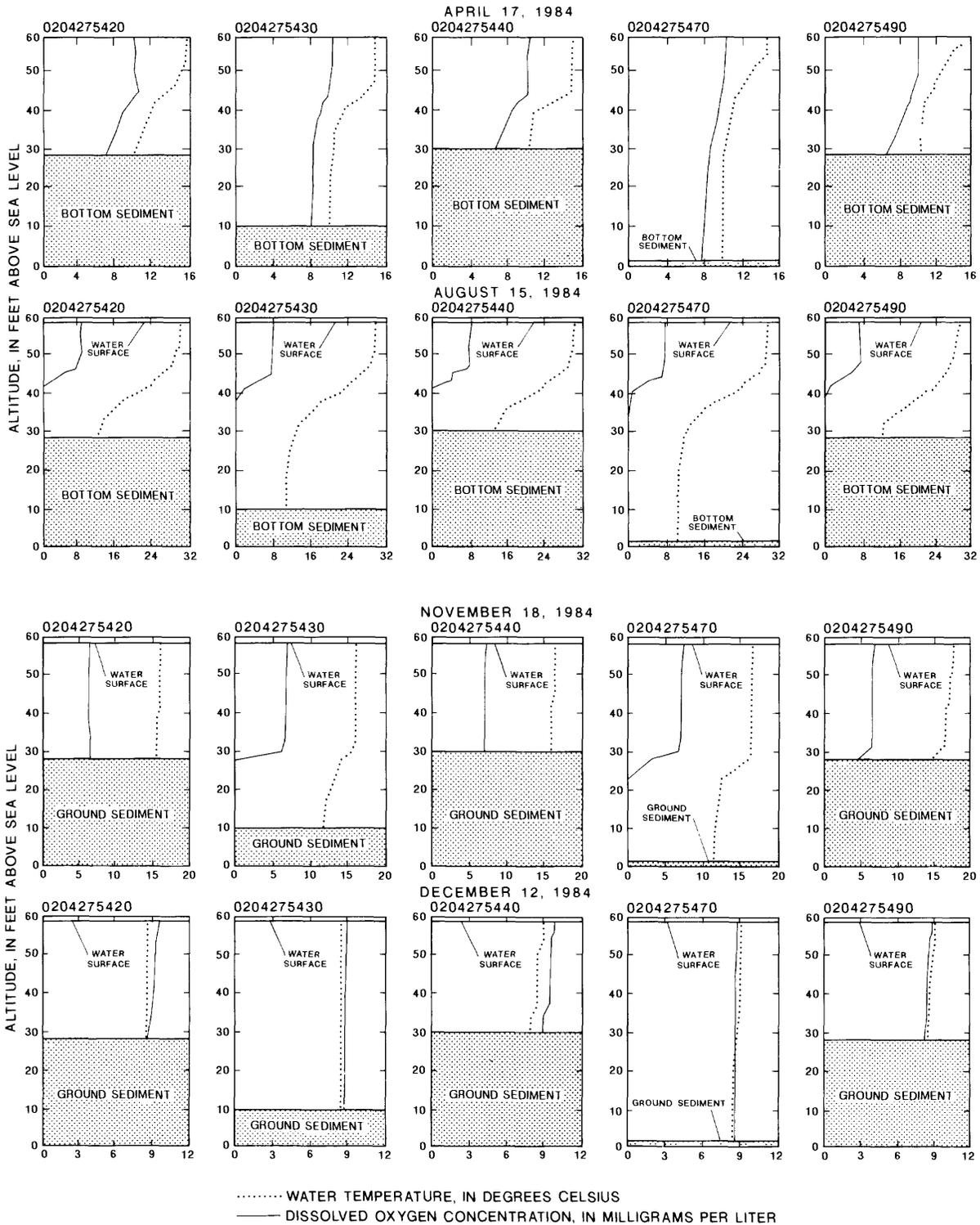


Figure 27.--Water temperature and dissolved-oxygen concentrations in water from Little Creek Reservoir on selected dates in 1984.

gen occurs by late June in Little Creek Reservoir, 1 month later than in Diascund Creek Reservoir. The Little Creek Reservoir hypolimnion requires longer for DO depletion because of its greater thickness and lesser amount of primary productivity. At full pool, the hypolimnion of Little Creek Reservoir averages 20 ft in thickness (comprising 50 percent of the reservoir volume), whereas the hypolimnion of Diascund Creek Reservoir averages only 4 ft in thickness.

Seasonal changes in vertical DO concentrations are consistent among the five sampling sites in Little Creek Reservoir. During early stages of stratification, as illustrated by the April 17, 1984, graphs in figure 27, epilimnion DO concentrations were nearly identical among sampling sites, and the hypolimnion DO concentration at each site was 1.5 to 3 mg/L less than the epilimnion concentration. Losses of DO from the hypolimnion were largest near the reservoir bottom at most sites, reflecting the effects of benthic oxygen demand. As evidenced by the DO concentrations for April 17, 1984 (fig. 27), hypolimnial DO concentration decreased more rapidly in the arms of the reservoir (sampling sites 0204275420, 0204275440, and 0204275490). In these shallow locations, the oxygen demand from settling organic matter and the benthos is exerted on a thinner part of the hypolimnion. The August 15, 1984, concentrations in figure 27 are typical of summer DO conditions in Little Creek Reservoir. These concentrations were also similar among sampling sites. DO concentrations in the 10- to 15-ft thick epilimnion were uniform, averaging 8.0 mg/L. Below this depth, DO concentrations decreased rapidly to zero through a thin (3 to 6 ft) metalimnion. The entire hypolimnion was anaerobic by August 15, 1984.

The November 18, 1984, DO concentrations in figure 27 show the uniform thickening of the epilimnion throughout the reservoir as fall overturn approaches. On this date, the epilimnion extended to a depth of about 30 ft, which was at or near the reservoir bottom at stations in the reservoir arms. Below a depth of 30 ft, anaerobic conditions persisted at sampling sites 0204275430 and 0204275470. The low epilimnial concentrations of DO on November 18, 1984, which averaged 6.5 mg/L (66 percent of saturation), resulted partly from the mixing of deeper, oxygen-depleted water into the epilimnion. Low DO concentrations during and after fall overturn result from a large oxygen demand created by organic matter and reduced species of nitrogen, iron, and manganese that accumulate in the anaerobic bottom water and sediments during thermal stratifi-

cation. Concentrations of DO measured on December 12, 1984, about 2 weeks after development of isothermal conditions (fig. 27), averaged 8.9 mg/L (77 percent of saturation). Undersaturation of DO several weeks after fall overturn persists because (1) reaeration of a deep water body occurs slowly, and (2) previously anaerobic bottom sediments continue to create an oxygen demand on the overlying water.

Major Inorganic Constituents

Concentrations of major ions in Little Creek Reservoir are strongly affected by inflows from Chickahominy and Diascund Creek Reservoirs. Sixty-seven percent of the total inflow of water in 1983-86 came from these two reservoirs (table 7). Consequently, the epilimnion of Little Creek Reservoir is dominated by calcium and bicarbonate (alkalinity) ions. Calcium concentrations averaged 9.5 mg/L between August 1983 and December 1984, whereas alkalinity concentrations averaged 23 mg/L as CaCO₃ in the epilimnion for the same study period (table 31). (Concentrations of some of the major ions were only measured during the first 17 months of the study.) Sodium, sulfate, and chloride ions make up a substantial percentage of the dissolved-solids concentration in the reservoir, however, at times rivaling calcium and bicarbonate as the dominant ions. For instance, water in Little Creek Reservoir changed from a calcium bicarbonate-type water in September 1983 to a calcium sulfate chloride bicarbonate-type water in March 1984. Inflows of calcium chloride sulfate-type water from Chickahominy and Diascund Creek Reservoirs, totalling more than 4.5 bgal (table 5), produced the change in water type during this 6-month period. Sulfate and chloride concentrations were higher in Chickahominy reservoir inflows, and alkalinity concentrations were lower during this 6-month period, compared with the remainder of the study (table 15).

Concentrations of magnesium and potassium in Little Creek Reservoir averaged 1.6 and 1.7 mg/L, respectively, during the first 17 months of the study, and showed little temporal variability. Similarly, concentrations of dissolved solids were relatively constant, ranging from 48 to 66 mg/L, and averaging 57 mg/L (table 31). As expected, concentrations of these constituents and other major ions in Little Creek Reservoir are similar to concentrations measured in inflows from Chickahominy and Diascund Creek Reservoir.

Table 31.--Concentrations of selected dissolved constituents in water at a depth of 10 feet in Little Creek Reservoir (0204275430), 1983-86

[Concentration in milligrams per liter; --, no data available; <, less than]

Date	Calcium	Magnesium	Sodium	Potassium	Alkalinity as CaCO ₃	Sulfate	Chloride	Fluoride	Silica, as SiO ₂	Hardness, as CaCO ₃	Dissolved Solids
83-08-03	10	1.5	6.1	1.7	26	8.1	11	<0.1	1.8	31	56
83-08-17	11	1.7	7.1	1.7	27	8.3	12	<.1	1.9	34	60
83-08-30	10	1.7	7.5	1.8	27	8.4	12	.1	2.0	32	60
83-09-13	9.8	1.7	8.9	1.5	28	8.5	14	.2	2.1	31	64
83-09-27	10	2.0	8.8	1.8	28	4.2	14	.1	2.0	33	60
83-10-14	10	1.7	8.0	1.7	28	4.8	13	.1	2.2	32	58
83-11-04	11	1.7	7.9	1.9	27	9.7	14	.2	2.2	34	65
83-11-29	10	1.7	7.3	2.0	23	14	13	.1	3.2	32	66
84-01-17	9.2	1.4	6.3	1.8	18	13	12	<.1	3.5	29	59
84-03-15	9.4	1.5	7.0	1.7	18	16	12	<.1	3.0	30	63
84-04-17	8.9	1.3	6.2	1.8	18	12	11	<.1	2.0	28	54
84-05-09	9.6	1.3	6.4	1.8	18	11	11	.1	1.1	29	54
84-06-01	8.7	1.4	6.0	1.7	19	9.4	11	<.1	.97	27	51
84-06-21	8.6	1.4	6.0	1.6	19	7.8	9.2	.1	1.7	27	48
84-07-11	9	1.4	6.0	1.6	21	9.3	10	<.1	1.6	28	52
84-08-02	9.0	1.4	5.7	1.6	22	9.3	9.8	<.1	1.5	28	52
84-08-15	9.1	1.4	5.9	1.6	21	9.4	9.1	.1	1.5	28	51
84-09-05	9.0	1.5	5.8	1.4	23	7.3	9.6	<.1	3.1	29	52
84-09-25	8.8	1.5	6.3	1.3	24	6.3	9.8	.1	3.0	28	53
84-10-18	9.4	1.6	6.5	1.5	24	6.8	11	.1	3.1	30	55
84-11-08	8.9	1.7	7.6	1.5	24	7.1	12	<.1	3.5	29	57
84-12-12	9.3	1.6	7.3	1.6	25	7.0	12	<.1	3.6	30	59
85-03-01	--	--	--	--	--	--	--	--	3.0	--	--
85-04-23	--	--	--	--	24	--	11	--	2.3	--	--
85-05-14	--	--	--	--	25	--	11	--	1.6	--	--
85-06-20	--	--	--	--	25	--	11	--	.36	--	--
85-07-17	--	--	--	--	26	--	12	--	.30	--	--
85-08-12	--	--	--	--	25	--	12	--	.39	--	--
85-09-04	--	--	--	--	25	--	12	--	.49	--	--
85-10-24	--	--	--	--	24	--	11	--	1.1	--	--
85-11-27	--	--	--	--	25	--	11	--	1.6	--	--
85-12-18	--	--	--	--	24	--	11	--	2.0	--	--
86-02-05	--	--	--	--	24	--	11	--	1.6	--	--
86-03-20	--	--	--	--	24	--	9.9	--	1.1	--	--
86-04-24	--	--	--	--	27	--	8.8	--	.81	--	--
86-05-28	--	--	--	--	27	--	10	--	.25	--	--
86-06-25	--	--	--	--	27	--	10	--	.50	--	--
86-07-23	--	--	--	--	26	--	9.7	--	.86	--	--
86-09-04	--	--	--	--	27	--	9.4	--	1.1	--	--
86-10-16	--	--	--	--	28	--	10	--	1.9	--	--
Mean	9.5	1.6	6.8	1.7	24	9.0	11	<.1	1.8	30	57
Maximum	11	2.0	8.9	2.0	28	16	14	.2	4.5	34	66
Upper quartile	10	1.7	7.5	1.8	27	9.7	12	.1	2.2	32	60
Median	9.4	1.5	6.4	1.7	25	8.4	11	<.1	1.8	30	56
Lower quartile	9.0	1.4	6.0	1.6	23	7.1	10	<.1	1.1	28	52

During periods when inflows to Little Creek Reservoir were small, such as December 1984 through February 1986 (table 5), major-ion concentrations in the upper 10 ft of water were stable. Alkalinity concentrations ranged from 24 to 26 mg/L, and chloride concentrations ranged from 11 to 12 mg/L during this period (table 31). These narrow concentration ranges, which were maintained for a 15-month period, attest to the conservative nature of these constituents in the epilimnion of Little Creek Reservoir.

The validity of equation 17 for distinguishing between conservative and nonconservative constituents in Little Creek Reservoir is substantiated fur-

ther by chloride, a known conservative tracer. The chloride concentration in the upper 10 ft of water at sampling site 0204275430 averaged 11 mg/L (table 31), which is close to the predicted concentration of 10.7 mg/L obtained from equation 17 (fig. 25). It was assumed that the volume-weighted chloride concentration in precipitation averaged 2 mg/L for this area of Virginia (Gambell and Fisher, 1966).

Concentrations of most major ions in Little Creek Reservoir do not differ appreciably between the epilimnion and hypolimnion. Samples collected in fall 1984 had similar concentrations of calcium, magnesium, sodium, potassium, sulfate, and chloride at depths of 10 and 40 ft (table 32). Only alka-

Table 32.--Typical fall values of selected constituents in water from epilimnion and hypolimnion of Little Creek Reservoir at sampling site 0204275430

[μ S/cm, microsiemens per centimeter; mg/L, milligrams per liter, less than; --, no data available]

Constituent or property	Depth, in feet			
	10		40	
	November 8, 1984		October 16, 1986	
Specific conductance, μ S/cm	108	168	102	211
Solids, sum of constituents, dissolved, mg/L ¹	57	96	--	--
pH [standard units]	6.9	6.6	6.9	6.8
Water temperature, degrees Celsius	16.0	12.0	19.0	13.0
Oxygen, dissolved, mg/L	6.5	0	6.3	0
Calcium, dissolved, mg/L	8.9	11	--	--
Magnesium, dissolved, mg/L	1.7	1.7	--	--
Sodium, dissolved, mg/L	7.6	7.0	--	--
Potassium, dissolved, mg/L	1.5	2.0	--	--
Alkalinity, mg/L as CaCO ₃	24	58	28	76
Bicarbonate, mg/L	29	71	34	93
Sulfate, dissolved, mg/L as SO ₄	7.1	6.3	--	--
Chloride, dissolved, mg/L	12	13	10	11
Fluoride, dissolved, mg/L	<.1	<.1	--	--
Silica, dissolved, mg/L as SiO ₂	3.5	4.5	1.9	4.8
Nitrite plus nitrate, dissolved, mg/L as N	<.1	<.1	<.1	<.1
Ammonia, dissolved, mg/L as N	.05	1.0	.03	1.9
Ammonia plus organic nitrogen, total, mg/L as N	.8	1.7	.9	3.1
Nitrogen, total, mg/L	.8	1.7	.9	3.1
Phosphorus, total, mg/L as P	.01	.03	.01	.02
Phosphorus, ortho dissolved, mg/L as P	<.01	.01	.004	.006
Iron, dissolved, mg/L	.17	14	.05	28
Manganese, dissolved, mg/L	.14	1.3	.04	1.3
Organic carbon, total, mg/L	--	8.4	6.3	9.4

¹ Includes dissolved iron and manganese in the calculation.

linity (bicarbonate) concentrations differed markedly with depth. Concentrations at 10 and 40 ft were 24 and 58 mg/L (as CaCO₃), respectively, on November 8, 1984 (table 32). As observed in Diascund Creek Reservoir, alkalinity concentrations increase because bicarbonate ions accompany the release of ferrous, manganoous, and ammonium ions from anaerobic bottom sediments.

A mass balance of dissolved silica (equation 17) indicates that Little Creek Reservoir provides a net sink for about 60 percent of the dissolved silica in inflows (fig. 25). (It is assumed that precipitation contains a negligible concentration of dissolved silica.) The dissolved-silica concentrations measured in upper waters averaged 1.8 mg/L (table 31); the predicted concentration, assuming silica is conservative, was 4.4 mg/L (fig. 25). Presumably, dissolved silica is lost during the uptake by diatoms and the subsequent settling of their frustules to bottom sediments.

Dissolved silica follows a seasonal cycle in both the epilimnion and hypolimnion of Little Creek Reservoir as it does in Diascund Creek Reservoir. Water-column concentrations are nearly uniform with depth during isothermal periods (November through March) (fig. 28); however, concentrations in the upper 10 ft of water decline rapidly during the first few months of stratification because of diatom uptake and settling. By May or June of each year, concentrations of dissolved silica decrease to less than 1.0 mg/L. An increase in dissolved silica in the upper 10 ft of water during the remainder of thermal stratification corresponds to a shift in the algal population away from diatoms. With this reduction in the silica sink, the upper 10 ft of water is slowly replenished with dissolved silica from internal and external sources. The summer and fall replenishment of dissolved silica in 1985 was slow because inflows of water from Chickahominy and Diascund Creek Reservoirs were negligible (table 5); these inflows are the primary external sources of dissolved silica to Little Creek Reservoir. The size of the diatom population that develops each spring in Little Creek Reservoir, and the timing of the shift to

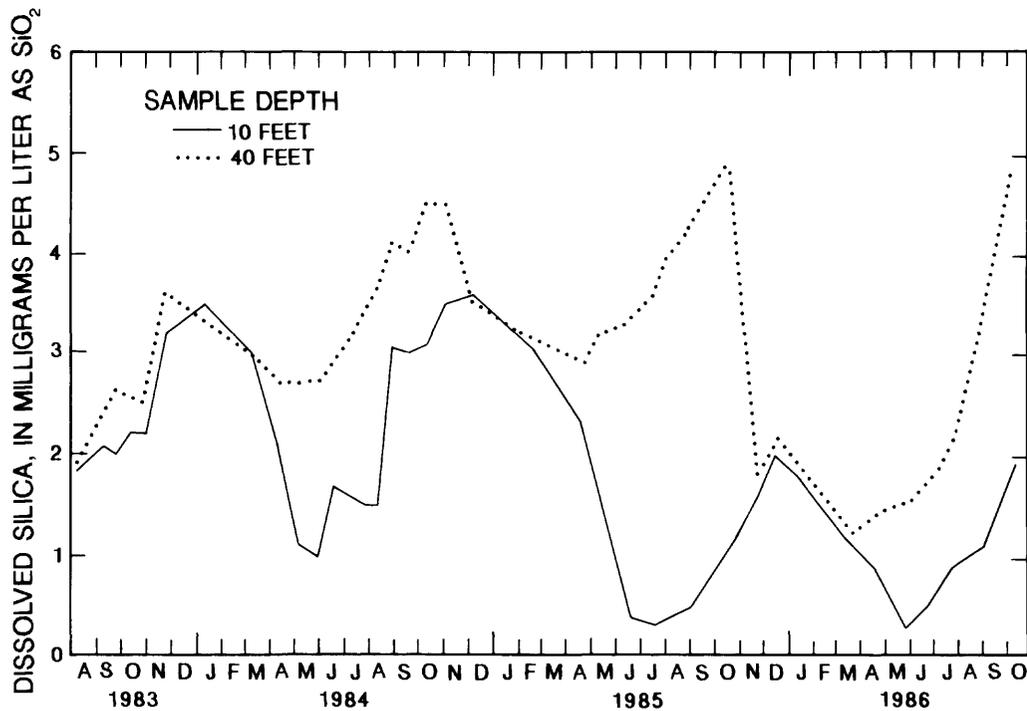


Figure 28.--Dissolved-silica concentrations in water at depths of 10 and 40 feet in Little Creek Reservoir, 1983-86.

other algal species in the summer, may depend largely on the quantity of dissolved silica supplied to the reservoir from external sources during the previous fall and winter.

The concentration of dissolved silica increases markedly in the hypolimnion soon after stratification. Dissolution of diatom frustules, which are either settling from the epilimnion or buried in bottom sediments, probably is the main source of this dissolved silica. Hypolimnetic concentrations commonly exceed 4.0 mg/L (as SiO₂) by the fall (fig. 27). This enriched hypolimnetic water, when mixed throughout the water column at fall overturn, undoubtedly is another important source of dissolved silica for diatom growth in Little Creek Reservoir during the following year.

Iron and Manganese

The processes affecting iron and manganese in Little Creek Reservoir are similar to the processes described earlier for Diascund Creek Reservoir. Concentrations of these metals result from a balance between sources and losses (sinks). Sources of iron and manganese include external inflows (primarily from Chickahominy and Diascund Creek Reservoirs) and internal sources (released from anaerobic bottom sediments). Losses include sorption of dissolved metals to bottom sediments and settling of metals in particulate matter. In addition, metals are removed from Little Creek Reservoir in raw-water withdrawals and spillway overflow and leakage.

Concentrations of dissolved iron and manganese in the upper 10 ft of Little Creek Reservoir are low, averaging 0.10 and 0.035 mg/L, respectively (table 33). These concentrations are less than the secondary maximum contaminant levels for drinking water of 0.3 mg/L for iron and 0.05 mg/L for manganese (U.S. Environmental Protection Agency, 1986b). Moreover, these concentrations are less than expected if these constituents are conservative. On the basis of external loadings (using equation 17), and the assumption that atmospheric sources of these metals were negligible, Little Creek Reservoir contained 55 percent less iron and 35 percent less manganese than expected (fig. 25). Losses of iron and manganese undoubtedly result from sorption to bottom sediments and settling of precipitates that form in the reservoir. Percentages of total iron and manganese lost from inflows to Little Creek Reservoir are probably similar to losses measured for the dissolved forms.

Routing water from Chickahominy and Diascund Creek Reservoirs through Little Creek Reservoir can substantially reduce the cost of water treatment by (1) decreasing the mean concentration of iron and manganese in raw water to be treated, and (2) diminishing the peak concentrations of these constituents in raw water that occur seasonally and under certain flow conditions. Peak concentrations in raw water are diminished when many months of inflows from Chickahominy and Diascund Creek are mixed and diluted in Little Creek Reservoir. (The hydraulic retention time of Little Creek Reservoir is 7 months when inflows from these reservoirs average 30 Mgal/d.) It should be noted, however, that the efficiency of iron and manganese removal from raw water could decrease as more water is routed through Little Creek Reservoir.

The highest concentrations of dissolved iron in the upper 10 ft of Little Creek Reservoir occurred December 1984 through March 1985 (table 33). The concentration on December 12, 1984, was 0.6 mg/L. This period of elevated dissolved-iron concentrations was preceded by 3 months of inputs from Chickahominy Reservoir that were relatively large (averaging 36 Mgal; table 5) and contained elevated concentrations of dissolved iron (averaging 0.7 mg/L; table 17). There appears to be a quantifiable link between external loading of iron to Little Creek Reservoir and the concentration of iron in the upper 10 ft of the water column.

Thermal stratification and the ensuing hypolimnetic DO depletion result in elevated concentrations of dissolved iron near the sediment-water interface (fig. 29). By October, concentrations near reservoir bottom sediments commonly exceeded 12 mg/L at sampling site 0204275430 (table 33). A concentration of 28 mg/L was measured at a depth of 40 ft on October 16, 1986 (table 32). The October 16, 1986, sample was collected within 1 or 2 ft of the reservoir bottom and could approximate the pore-water concentration of dissolved iron as well as other constituents. As in Diascund Creek Reservoir, greater than 90 percent of the total iron in the hypolimnion of Little Creek Reservoir is dissolved, presumably present as ferrous ions. Ferrous ions, along with bicarbonate ions, are released from anaerobic bottom sediments as ferric oxyhydroxide is reduced during the decomposition of organic matter (reaction 9). This release corresponds to the depletion of DO at the sediment-water interface, beginning around June (fig. 26).

Table 33.--Dissolved-iron and -manganese concentrations in water from multiple depths below the surface at sampling site 0204275430 in Little Creek Reservoir, 1983-86

[Concentrations in milligrams per liter; --, no data available; <, less than]

Date	Dissolved iron					Dissolved manganese				
	3 feet	10 feet	20 feet	30 feet	40 feet	3 feet	10 feet	20 feet	30 feet	40 feet
08-03-83	--	0.050	--	--	6.4	--	<0.001	--	--	0.66
08-17-83	--	.038	--	--	8.2	--	<.001	--	--	.71
08-30-83	--	.031	--	--	9.4	--	<.001	--	--	.76
09-13-83	--	.044	--	--	12	--	<.001	--	--	.81
09-27-83	--	.038	--	--	12	--	.003	--	--	.83
10-14-83	--	.032	--	--	13	--	<.001	--	--	.90
11-04-83	--	.15	--	--	14	--	.28	--	--	.96
11-29-83	--	.15	--	--	.19	--	.12	--	--	.096
01-17-84	.26	.28	.26	.20	.15	.03	.032	.02	.02	.012
03-15-84	.06	.11	.06	.11	.12	<.01	.005	<.01	<.01	.006
04-17-84	.13	.08	.11	.15	.15	.01	.008	.01	.05	.073
05-09-84	.08	.06	.06	.13	.09	.02	.006	.06	.10	.13
06-01-84	.12	.11	.06	.06	.10	.01	.006	.13	.15	.37
06-21-84	.16	.16	.06	.25	.29	.03	.017	.25	.62	.63
07-11-84	.20	.15	.07	.37	.89	.07	.004	.21	.97	.93
08-02-84	.10	.10	.16	3.2	5.7	.01	.002	.47	1.1	1.1
08-15-84	.10	.08	.08	4.3	6.9	.03	.006	.76	1.1	1.1
09-05-84	--	.17	--	--	10	--	.005	--	--	1.2
09-25-84	.11	.11	.13	7.6	12	.02	.003	.30	1.3	1.3
10-18-84	.12	.11	.15	9.9	14	.08	.064	.06	1.6	1.3
11-08-84	.14	.17	.08	4.1	14	.15	.14	.14	1.4	1.3
12-12-84	.60	.61	.52	.58	.58	.28	.26	.28	.27	.25
03-01-85	.30	.28	.28	.31	.32	.055	.055	.055	.062	.063
04-23-85	.031	.035	.045	.046	.059	.014	.003	.002	.015	.022
05-14-85	.035	.029	.024	.037	.064	.075	.013	.017	.17	.47
06-20-85	.074	.022	.020	.12	1.3	.087	.012	.073	.61	.77
07-17-85	.035	.039	.012	2.6	3.4	.044	.018	.042	.84	.89
08-12-85	.032	.033	.026	3.1	7.1	.028	.009	.16	.98	1.0
09-04-85	.052	.038	.032	1.6	9.5	.028	.003	.003	1.3	1.0
10-24-85	--	.050	--	--	14	--	.011	--	--	1.3
11-27-85	.11	.12	.11	.18	.20	.032	.032	.028	.042	.050
12-18-85	.080	.071	.09	.067	.098	.011	.010	.011	.011	.010
02-05-86	.015	.024	.015	.016	.014	.004	.004	.004	.004	.004
03-20-86	.030	.022	.017	.013	.015	.005	.005	.003	.004	.012
04-24-86	.067	.029	.022	.019	.021	.033	.031	.032	.20	.22
05-28-86	.070	.11	.027	.032	.026	.017	.004	.004	.27	.44
06-25-86	.067	.071	.12	.077	1.9	.003	.003	.047	.29	.97
07-23-86	.059	.050	.25	2.7	8.5	.004	.010	.27	.44	.89
09-04-86	.024	.018	.032	12	18	.005	.001	.013	.93	1.1
10-16-86	.073	.054	.085	22	28	.044	.038	.036	1.1	1.3

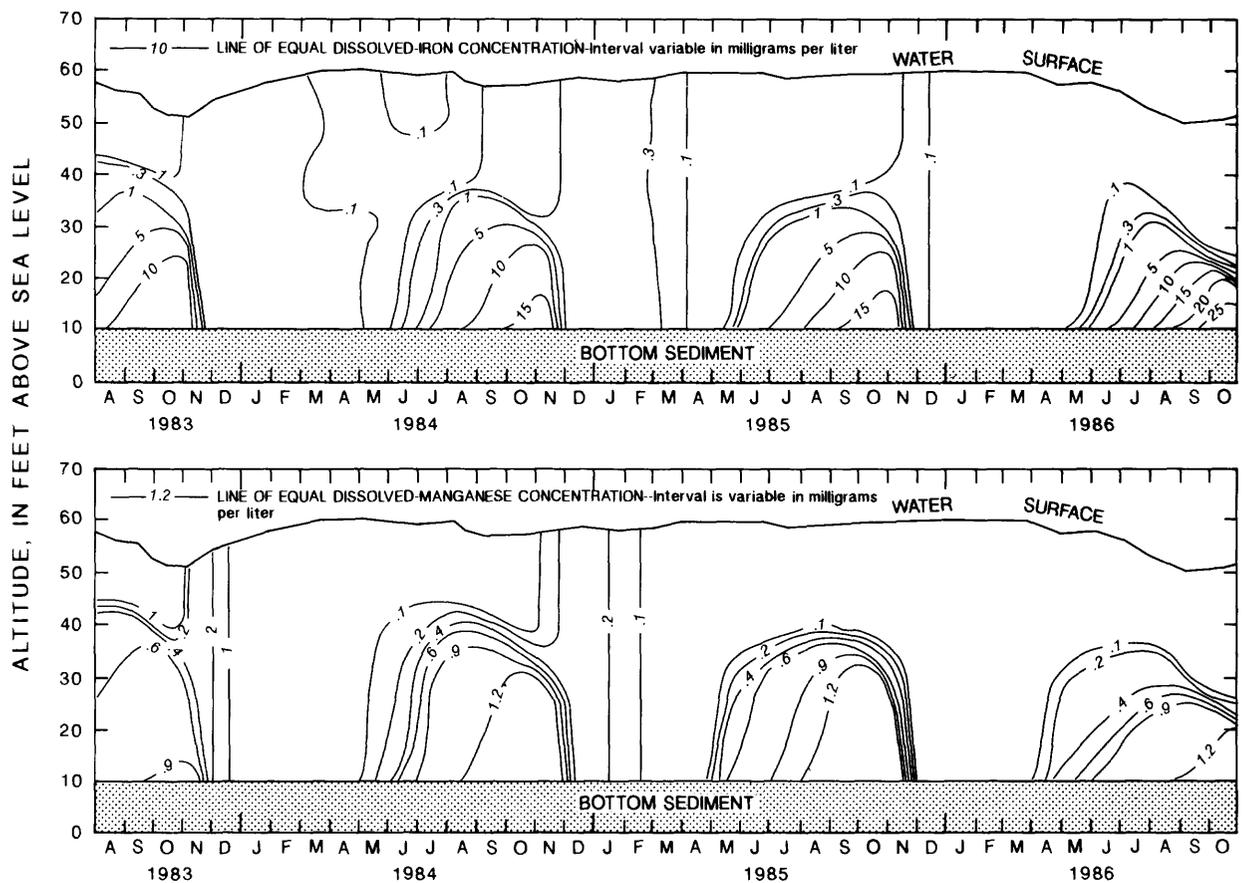


Figure 29.--Dissolved-iron and dissolved-manganese concentrations in water from Little Creek Reservoir, 1983-86.

The concentration of ferrous ions in the pore water of anaerobic bottom sediments is probably affected by a ferrous-carbonate solid phase, such as siderite (reaction 10). If one assumes that water collected from a depth of 40 ft on November 8, 1984, reasonably represents the chemistry of pore water in Little Creek Reservoir sediments (table 32), it appears that this water is nearly saturated with respect to siderite, as evidenced by a log (AP/K) value of -0.15. This result was generated from WATEQF, a mineral-solution equilibrium program (Plummer and others, 1976). Saturation would be indicated (that is, a log (AP/K) value of zero), if dissolved iron and bicarbonate concentrations were 20 percent greater. If water collected within 1 or 2 ft of the sediment-water interface on October 16, 1986, is assumed to represent the chemistry of anaerobic pore water, then this water is supersaturated (log AP/K value of 0.5) with respect to siderite. Concentrations of dissolved iron and bicarbonate on this date were 28 and 93 mg/L, respectively, and the pH was 6.8 (table 32).

The amount of ferric oxyhydroxide reduced in Little Creek Reservoir bottom sediments, and thus the amount of dissolved iron available for release or formation of siderite, is affected by the availability of organic matter that drives reaction 9. Organic matter driving this reaction comes primarily from algal cells and other organic detritus settling from the water column. If the supply of organic matter is small and the activity product of ferrous and bicarbonate ions in pore water does not reach the equilibrium constant for siderite, the quantity of dissolved iron in the hypolimnion would be limited solely by reaction 9. Thus, siderite would not affect the ferrous ion concentration in pore water (reaction 10), and siderite would not form in bottom sediments. In contrast, a relatively large supply of organic matter to bottom sediments increases the potential for release of ferrous ions to the hypolimnion and formation of siderite in bottom sediments. As discussed for Diascund Creek Reservoir, siderite formed in anaerobic bottom sediments can exert

substantial chemical oxygen demand (reaction 11) on the entire water column following fall overturn. This demand can be exerted over a period of weeks or months, which may partly explain why water in Little Creek Reservoir is undersaturated with DO (less than 80 percent) during December.

Release of manganous ions from bottom sediments in Little Creek Reservoir occurs in May, about a month earlier than release of ferrous ions. Although oxygen is not necessarily depleted at the sediment-water interface in May, bottom sediments could be sufficiently reducing for the accumulation of manganous ions in pore water. Because manganous ions oxidize slowly, particularly in environments with low DO and pH (Stumm and Morgan, 1981), these ions can diffuse from the bottom sediments to the hypolimnion and can persist for weeks in the water column before anaerobic conditions develop.

Concentrations of dissolved manganese near the bottom sediments of Little Creek Reservoir commonly exceed 1.2 mg/L by August (fig. 29). Greater than 95 percent of the total manganese in the hypolimnion is dissolved, presumably present as manganous ions. As suggested for Diascund Creek Reservoir, the concentration of manganous ions in the pore water of anaerobic bottom sediments could be affected by a manganous carbonate solid phase, such as rhodochrosite; however, water near the sediment-water interface of Little Creek Reservoir is undersaturated with respect to rhodochrosite, as evidenced by a log (AP/K) value of -0.96. The undersaturation with respect to rhodochrosite indicates a possible limited source of manganese in bottom sediments or a microbial preference for ferric-containing substrates as electron acceptors during decomposition of organic matter.

Nutrients, Chlorophyll-a, and Reservoir Transparency

Nutrients in Little Creek Reservoir are affected by a variety of sources and sinks. Inflows from Chickahominy and Diascund Creek Reservoirs are the largest nutrient source. During the study period, the combined inflows from these reservoirs provided 73 and 75 percent of the external loads of total nitrogen and phosphorus, respectively. Precipitation and inflows from the terrestrial drainage basin provided the remaining major sources of nutrients. Fixation of atmospheric nitrogen by blue-green algae can provide an additional source of nitrogen; however, this source is thought to be negligible because algae

in Little Creek Reservoir are not thought to be nitrogen-limited. Generally, nitrogen fixation does not occur in phosphorus-limited algae. Net losses of nutrients include permanent sediment burial, spillway overflow and leakage, and withdrawal of raw water for treatment. Additionally, nitrogen could be removed from Little Creek Reservoir by denitrification of nitrite and nitrate in anaerobic bottom water and sediments.

Nitrogen and phosphorus are cycled with different efficiencies in Little Creek Reservoir. Nitrogen cycling is efficient, with only 20 percent of the incoming load permanently lost from the water column through denitrification or sediment burial (fig. 25). In calculating this loss with equation 17, it is assumed that (1) the volume-weighted total-nitrogen concentration in precipitation is 0.50 mg/L (Lynch and Dise, 1986) and (2) nitrogen fixation in Little Creek Reservoir is negligible. Total-nitrogen concentrations in the upper 10 ft of water at sampling site 0204275430 averaged 0.70 mg/L; a concentration of 0.85 mg/L would be expected if total nitrogen was conservative (fig. 25). Presumably, nitrogen contained in settling algal cells and other organic matter is efficiently mineralized at the sediment-water interface, and the released ammonium ions are not retained by the bottom sediments.

In contrast to nitrogen, phosphorus cycling is less efficient, with 70 percent of the incoming load to Little Creek Reservoir permanently lost to bottom sediments (fig. 25). This loss, calculated by use of equation 17, is based on an annual precipitation contribution of 2.3 milligrams per square foot (Rast and Lee, 1983), or a volume-weighted concentration of 0.022 mg/L for precipitation falling on Little Creek Reservoir. The estimate by Rast and Lee (1983) for phosphorus loading is an average of published values throughout the United States. Total-phosphorus concentrations in the upper 10 ft of water averaged about 0.014 mg/L; a concentration of about 0.048 mg/L would be expected if phosphorus was conservative (fig. 25). Losses of phosphorus include direct sorption of dissolved phosphorus to bottom sediments and settling of particulate phosphorus to bottom sediments. Bottom sediments in Little Creek Reservoir have a capacity to sorb and retain phosphorus under aerobic and anaerobic conditions.

The N:P ratio (by weight) in water from Little Creek Reservoir is greater than from any of the inflow sources because the reservoir retains phos-

phorus more efficiently than it retains nitrogen. The N:P ratios of inflows from Chickahominy and Diascund Creek Reservoirs averaged 20 and 15, respectively, or less than one-half the ratio of 50 observed in the upper 10 ft of water from sampling site 0204275430 (fig. 25). The N:P ratio was rarely less than 20, which indicates that phosphorus is the nutrient that potentially limits algal growth in Little Creek Reservoir.

Overall, concentrations of nutrients and chlorophyll-*a* in the upper 10 ft of water in Little Creek Reservoir are uniformly distributed. Similarly, Secchi-disk transparency depths do not differ significantly among sampling sites. Table 28 summarizes data from between August 1983 through December 1984 when all five sites were sampled synoptically. Table 29 summarizes data collected during the entire study for sampling sites 0204275430 and 0204275470. Most of the concentration differences among sampling sites were not statistically significant ($p > 0.05$), based on a Wilcoxon matched-pairs signed-ranks test (SPSS, Inc., 1988). These data indicate that nutrient inflows are well mixed horizontally in Little Creek Reservoir and that in-reservoir cycling of nutrients is evenly distributed. One noteworthy exception is total phosphorus. Concentrations in the upper 10 ft of water at sampling sites 0204275430 and 0204275470 differed significantly, averaging 0.014 and 0.010 mg/L, respectively (table 29). Total-phosphorus concentrations could be lower at sampling site 0204275470, compared with 0204275430, because of its greater distance from the inflow from Chickahominy and Diascund Creek Reservoirs (fig. 3), which allows time for phosphorus to sorb to bottom sediments or settle out of the water column. Total-phosphorus concentration at a sampling site may be related to its distance from shore, however. Nearshore sources could account for the higher concentrations of phosphorus at sampling sites 0204275420, 0204275430, 0204275440, and 0204275490 compared with the open-water sampling site 0204275470 (table 28). Contributions from nearshore areas include partial recycling of particulate phosphorus settling in shallow areas, surface-water and ground-water inflows, resuspension of bottom sediments, and direct uptake of phosphorus by algae in contact with bottom sediments.

The concentration of total nitrogen in the hypolimnion of Little Creek Reservoir varies seasonally. In May or June, hypolimnion concentrations increase steadily until fall overturn in November (fig. 30). Maximum total-nitrogen concentrations

occurred near the bottom sediments in October and November, periodically exceeding 2.0 mg/L. The increasing hypolimnial concentrations result from mineralization of organic nitrogen in or near bottom sediments, and the subsequent release of ammonia. Algal cells and other organic debris settling from the water column provide the organic-nitrogen source. Released ammonia comprises about 60 percent of the total nitrogen concentration in the hypolimnion during the summer, which averages 0.76 at a depth of 40 ft (table 34) and periodically exceeds 1.0 mg/L. The mean concentration of organic nitrogen in Little Creek Reservoir did not differ significantly with season or depth, ranging from 0.5 to 0.6 mg/L. Organic nitrogen concentrations are obtained by subtracting ammonia concentrations from ammonia plus organic nitrogen concentrations in table 34.

Total-nitrogen concentration in the upper 10 ft of water in Little Creek Reservoir showed little seasonal variation, averaging 0.7 mg/L (table 34); however, concentrations of dissolved nitrite-plus-nitrate and ammonia in the upper 10 ft of water varied seasonally, the highest concentrations occurring from January through March. Mixing of hypolimnial waters at fall overturn accounted for the elevated winter (October-March) concentrations of ammonia, which averaged 0.10 mg/L. Increased nitrite-plus-nitrate concentrations during the winter probably resulted from biological nitrification of ammonium ions that are mixed from the hypolimnion or released from bottom sediments during isothermal periods. Nitrite-plus-nitrate concentrations averaged 0.12 mg/L during the winter months, concentrations during the remainder of the year averaged less than 0.10 mg/L. Minimum dissolved-ammonia concentrations in the upper 10 ft of water occurred during the summer (July-September), averaging 0.02 mg/L (table 34). The low summer concentrations probably resulted from the cumulative loss of nitrogen as algal cells settled out of the upper 10 ft of water during this optimal growth period.

Concentrations of dissolved nitrite-plus-nitrate and ammonia are subject to short-term variations as well. These dynamic pools of nitrogen, which are the primary nitrogen sources for algal growth, change hourly or daily in response to algal uptake and release, mineralization of organic nitrogen, and biological processes of nitrification and denitrification. Although these forms of nitrogen make up less than 20 percent of the total nitrogen in the upper 10 ft of water, they do not limit the biomass of algae. Phosphorus limitation is indicated, as evidenced by

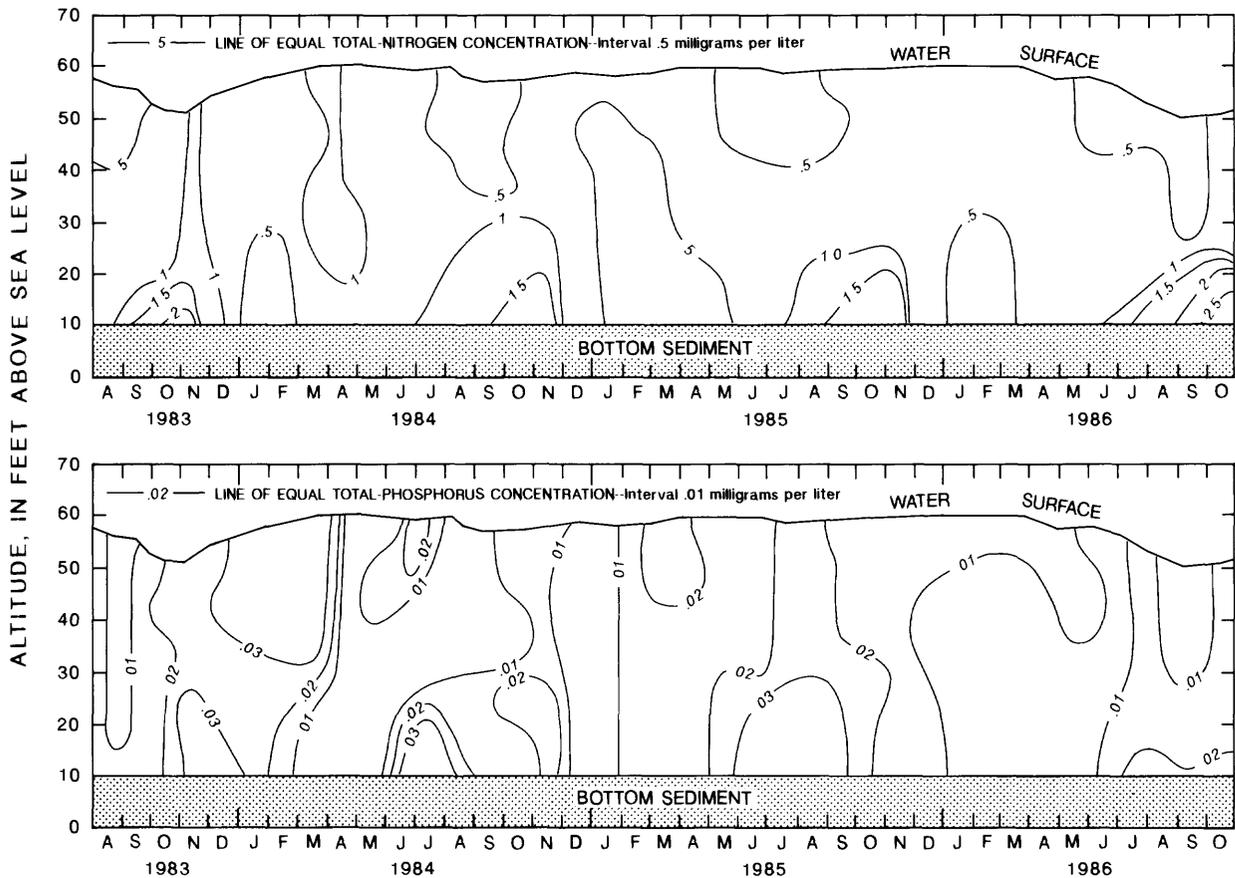


Figure 30.— Total-nitrogen and total-phosphorus concentrations in water from Little Creek Reservoir, 1983-86.

an N:P ratio of 23 for nutrients readily available for algal assimilation (dissolved nitrite, nitrate, ammonia, and orthophosphorus).

Dissolved-orthophosphorus concentrations averaged less than 0.01 mg/L in Little Creek Reservoir, differing little with season or water depth (table 34). These extremely low concentrations are typical of a reservoir with a phosphorus-limited algal population. During the last 19 months of the study, when the orthophosphorus detection limit was 0.001 mg/L, the concentration of dissolved orthophosphorus throughout the water column averaged 0.004 mg/L at sampling site 0204275430.

Seasonal variability of total phosphorus is most prominent in the bottom water of Little Creek Reservoir. A slight amount of total phosphorus accumulates in bottom water during thermal stratification, as evidenced by a total-phosphorus

concentration of 0.018 mg/L in water at a depth of 40 ft (sampling site 0204275430) compared with 0.014 mg/L at a depth of 10 ft. Concentrations exceeding 0.030 mg/L, however, were measured near the sediment-water interface during the summers of 1984 and 1985 (fig. 30). The accumulation of phosphorus in bottom water results from settling of particulate phosphorus from the epilimnion, not from a net release of phosphorus from anaerobic bottom sediments.

Similar to Diascund Creek Reservoir, but unlike most productive reservoirs described in the literature, the bottom sediments of Little Creek Reservoir are a net sink of phosphorus throughout the year. The lack of a net phosphorus release, even under anaerobic conditions, precludes the large hypolimnial accumulation of phosphorus that triggers algal blooms at fall overturn in many reservoirs. The

Table 34.—Average seasonal concentrations of nutrients and chlorophyll-*a* in water from sampling sites 0204275430 and 0204275470 in Little Creek Reservoir, 1983-86

[Concentrations in milligrams per liter, except for chlorophyll-*a* which is in micrograms per liter; <, less than; locations of sampling, figure 3]

Depth in feet	Nitrite, plus nitrate, dissolved as N		Ammonia, dissolved as N		Ammonia plus organic nitrogen, total, as N		Nitrogen, total as N		Phosphorus, total as P		Phosphorus, ortho dissolved, as P		Chlorophyll- <i>a</i>	
	0204275430	0204275470	0204275430	0204275470	0204275430	0204275470	0204275430	0204275470	0204275430	0204275470	0204275430	0204275470	0204275430	0204275470
January - March														
10	0.12	0.11	0.10	0.12	0.6	0.7	0.7	0.8	0.019	0.011	<0.01	<0.01	10.3	7.4
Bottom ^a	.11	.11	.12	.10	.5	.6	.6	.7	.012	.010	<.01	<.01	6.2	6.7
April - June														
10	<.1	<.1	.04	.04	.9	.5	.9	.5	.012	.010	<.01	<.01	5.0	4.9
Bottom ^a	.10	.11	.25	.24	.7	.7	.8	.8	.014	.013	<.01	.01	1.4	1.8
July - September														
10	<.1	<.1	.02	.02	.6	.5	.6	.5	.013	.011	<.01	<.01	8.4	7.8
Bottom ^a	<.1	<.1	.76	.66	1.2	1.0	1.2	1.0	.017	.016	<.01	<.01	4.8	2.7
October - December														
10	<.1	<.1	.08	.11	.7	.6	.7	.6	.014	.010	<.01	<.01	14.3	12.0
Bottom ^a	<.1	<.1	.76	.90	1.4	1.4	1.4	1.4	.019	.016	<.01	<.01	14.9	17.1
Annual Average														
10	<.1	<.1	.06	.07	.7	.6	.7	.6	.014	.011	<.01	.01	9.5	8.0
Bottom ^a	<.1	<.1	.47	.48	1.0	.9	1.0	1.0	.016	.014	<.01	<.01	6.8	7.1

^a Samples collected at a depth of 40 feet from station 0204275430 and at a depth of 50 feet from station 0204275470.

mechanism(s) of phosphorus retention in bottom sediments is probably the same as described earlier for Diascund Creek Reservoir. Phosphorus could be retained in refractory, high molecular-weight organic compounds in sediments, similar to those described by Schindler and others (1977). More likely, however, phosphorus could be retained in sediments by a solid-phase equilibrium reaction involving either (1) sorption to clays, (2) sorption to ferric and ferrous complexes, or (3) formation of an iron phosphate mineral, such as vivianite.

WATEQF (Plummer and others, 1976) was used to evaluate the possible effect of vivianite on the pore-water concentration of orthophosphorus in

Little Creek Reservoir. Water collected from a depth of 40 ft (sampling site 0204275430) on October 16, 1986 (table 32), was assumed to closely reflect the pore-water chemistry of anaerobic bottom sediments. This sample was used because it was collected within 1 to 2 ft of the sediment-water interface and near the end of the summer stratification period. On the basis of a pH of 6.8 and respective concentrations of orthophosphorus and dissolved iron of 0.006 and 28 mg/L, water near the sediment-water interface (and presumably in pore water) was nearly saturated with respect to vivianite, as evidenced by a log (AP/K) value of -0.4. As suggested by Paces (1972), log (AP/K) values between -0.5 and 0.5 indicate possible equilibrium with the mineral consid-

ered. The orthophosphorus concentration for saturation with respect to vivianite would be only 0.010 mg/L for the sample collected.

Chlorophyll-*a* concentrations in Little Creek Reservoir did not differ appreciably among sampling sites. Mean concentrations in the upper 10 ft of water ranged from 7.7 µg/L at sampling site 0204275470 to 11.9 µg/L at sampling site 0204275440 during the first 17 months of the study (table 28). Concentrations at sampling sites 0204275430 and 0204275470 were nearly identical during the entire study, averaging 7.9 and 7.4 µg/L, respectively (table 29).

Although chlorophyll-*a* concentrations are similar among sampling sites, concentrations do differ considerably with reservoir depth and season, as is evident in figure 31. Because of better light conditions, the upper 10 ft of water supports a larger,

more viable algal population than does bottom water, and contains higher concentrations of chlorophyll-*a*; however, relatively high chlorophyll-*a* concentrations do occur with depth during the latter part of thermal stratification, typically from August through November. These high chlorophyll-*a* concentrations with depth, which at times exceed 25 µg/L near the sediment-water interface (fig. 31), could result from either (1) an accumulation of algal cells settling from the epilimnion; or (2) the slow growth of algae or photosynthetic bacteria in this low-light environment. Algae and photosynthetic bacteria commonly produce high cellular chlorophyll-*a* concentrations in low-light environments to maximize photosynthesis.

Low concentrations of chlorophyll-*a* during the spring months, which averaged 5.0 and 1.4 µg/L at depths of 10 and 40 ft (sampling site 0204275430), respectively, result from algal succession (table 34).

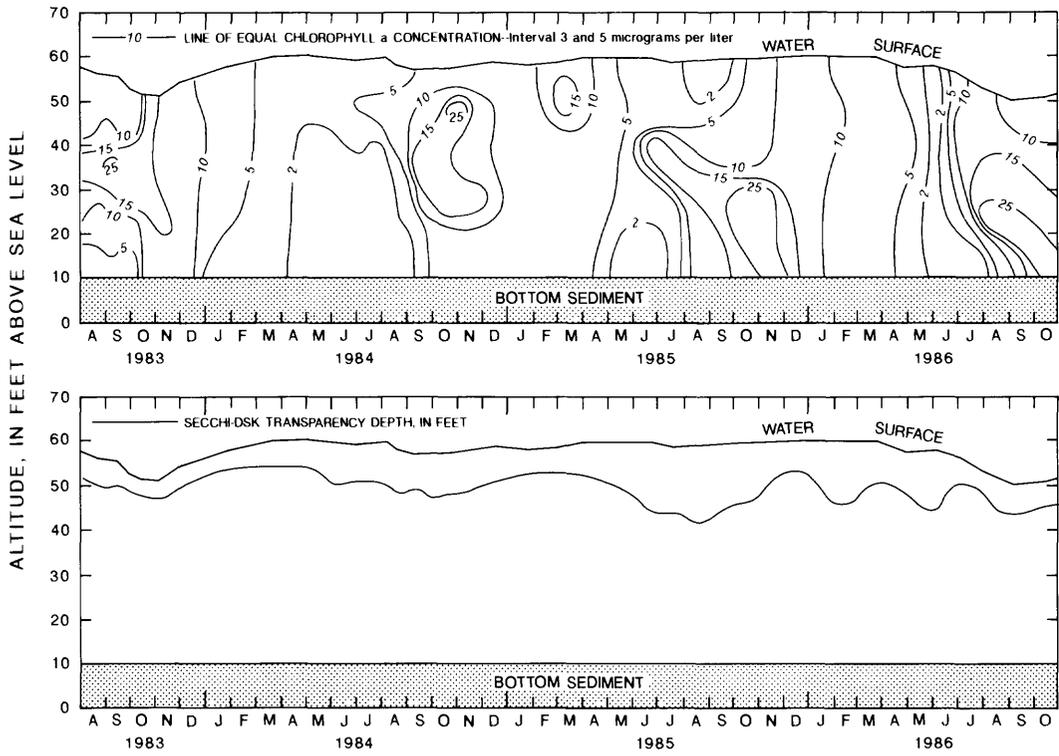


Figure 31.--Chlorophyll-*a* concentrations and Secchi-disk transparency depth in water from Little Creek Reservoir, 1983-86.

Diatoms, which dominate the algal population during isothermal periods, settle out of the water column in April and May as the epilimnion warms and thermal stratification is established. Diatoms rely on cool (more viscous), well-mixed water to remain suspended in the water column and exposed to adequate light for growth. A mixed algal population, more adept than diatoms at controlling flotation and light exposure, slowly replaces diatoms during the first few months of stratification. The lag time between the decline of the diatom population and the replacement by more motile algae results in an annual chlorophyll-*a* minimum about May (fig. 32). The May concentrations for 1984 through 1986 averaged about 3 µg/L in the upper 10 ft of water at sampling site 0204275430.

Chlorophyll-*a* concentrations in Little Creek Reservoir appear to be partly related to variations in the quantity and quality of inflows from Chickahominy and Diascund Creek Reservoirs. For example, large inflows from these reservoirs from July 1983 through February 1984 (table 5) resulted in increased concentrations of phosphorus in Little Creek Reservoir, which led to increased concentrations of chlorophyll-*a* and decreased Secchi-disk transparencies (fig. 32). Inflows during this period, particularly those from Diascund Creek Reservoir in its drawn-down state, contained high concentrations of phosphorus. Concentrations of total phosphorus and chlorophyll-*a* in Little Creek Reservoir averaged 0.021 mg/L and 11 µg/L, respectively, during this period. These concentrations are about 60 percent greater than concentrations measured during the same months (August through February) in subsequent years. Secchi-disk transparency depths in Little Creek Reservoir decreased steadily during this period of increased phosphorus and chlorophyll-*a* concentrations. From August 1983 through February 1984, the transparency depth averaged 5 ft, or about one-half the depth measured during the same months in subsequent years.

In contrast to the first 7 months of the study, inflows from Chickahominy and Diascund Creek Reservoirs were negligible in 1985 (table 5). The small external load of phosphorus to Little Creek Reservoir in 1985 resulted in low concentrations of chlorophyll-*a* in upper waters and the greatest Secchi-disk transparency depths (fig. 32). Transparency

depths at sampling site 0204275430 were greater than 10 ft for the entire period of stratification in 1985, reaching a maximum of 18 ft in August 1985.

Although a relation may exist between concentrations of total phosphorus and chlorophyll-*a* in Little Creek Reservoir, the correlation is not statistically significant ($p > 0.05$). No correlation was obtained because the phosphorus concentration in Little Creek Reservoir varies only within a narrow range. Consequently, little of the chlorophyll-*a* variability can be explained by the potentially limiting nutrient. Most of the chlorophyll-*a* variability in Little Creek Reservoir is probably explained by light conditions, water temperature, vertical mixing patterns, zooplankton, grazing, and algal succession.

The lack of a correlation between phosphorus and chlorophyll-*a* concentrations in Little Creek Reservoir, as well as in Chickahominy and Diascund Creek Reservoirs, does not preclude phosphorus as the limiting nutrient. As discussed earlier, the phosphorus concentration defines the upper boundary for chlorophyll-*a* (algal biomass) in these reservoirs and indirectly affects the mean reservoir concentration as well. This relation is apparent when the three reservoirs are compared on a plot of mean total-phosphorus concentration as a function of mean chlorophyll-*a* concentration. The resulting relation is both positive and linear (fig. 33). (Only samples collected during base-flow periods are included for Chickahominy Reservoir.) Mean concentrations of total phosphorus and chlorophyll-*a* are lowest in Little Creek Reservoir, are intermediate in Diascund Creek Reservoir, and are highest in Chickahominy Reservoir. If Diascund Creek Reservoir in a drawn-down condition is considered separately, mean concentrations of total phosphorus and chlorophyll-*a* would be highest (fig. 33). Similarly, a relation exists between the mean chlorophyll-*a* concentration in a reservoir and the mean Secchi-disk transparency depth. As expected, this relation is negative and nonlinear (fig. 33). Taken together, the data presented in figure 33 indicate that increasing the mean phosphorus concentration in any of these reservoirs would both increase the mean concentration of chlorophyll-*a* and decrease the Secchi-disk transparency depth in a predictable manner.

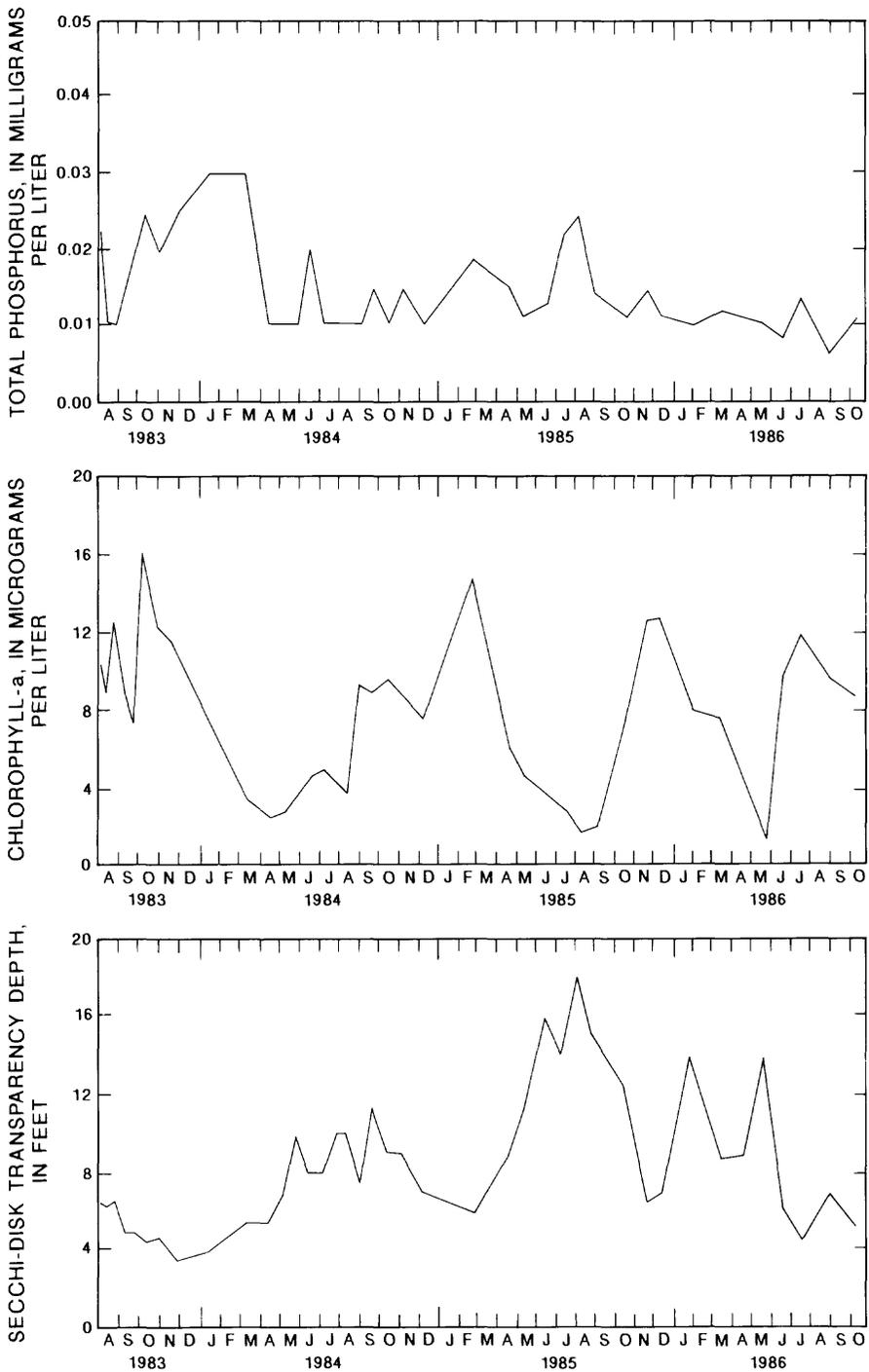


Figure 32.--Total-phosphorus and chlorophyll-a concentrations in the upper 10 feet of water and Secchi-disk transparency depth for Little Creek Reservoir, 1983-86.

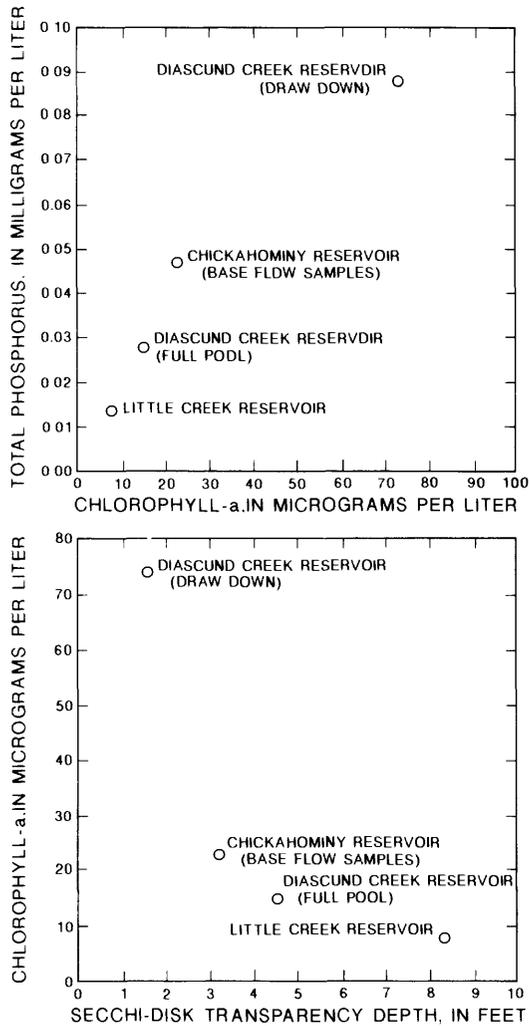


Figure 33.—Relations between average chlorophyll-*a* and total phosphorus concentrations, and average Secchi-disk transparency depth and chlorophyll-*a* concentrations in the upper 10 feet of water in Chickahominy, Diascund Creek, and Little Creek Reservoirs.

EVALUATION OF RAW-WATER ROUTING SCENARIOS

A nomograph was developed to estimate phosphorus concentration in Little Creek Reservoir, given differing amounts and concentrations of inflows from Chickahominy and Diascund Creek Reservoirs. The nomograph, shown in figure 34, is based on the following empirical model (Reckhow and others, 1980):

$$P = \frac{L}{11.6 + 1.2(qs)}, \quad (18)$$

where

P is estimated mean total-phosphorus concentration in reservoir, in milligrams per cubic meter;

L is annual phosphorus loading in reservoir, in milligrams per square meter; and

qs is annual hydraulic loading in reservoir, in meters.

This empirical model incorporates data from 47 temperate lakes and reservoirs in the United States. The following assumptions were made in developing the nomograph in figure 34 from equation 18:

- (1) Annual hydraulic loading (*qs*) from natural inflows equaled 3.8 ft (1.16 m), which was based on terrestrial inflows, plus precipitation, minus evaporation, for 1982 through 1987 (table 5);
- (2) Annual hydraulic loading (*qs*) from Chickahominy and Diascund Creek Reservoirs equaled 1.18 ft (0.36 m) when the combined pumpage averaged 1 Mgal/d;
- (3) Annual phosphorus loading (*L*) from precipitation equaled 0.21 mg/m² (Rast and Lee, 1983); and
- (4) Annual phosphorus loading (*L*) from terrestrial inflows equaled 0.30 mg/m²; it is assumed that phosphorus concentration in terrestrial inflows was similar to that in Chickahominy and Diascund Creek Reservoirs (0.05 mg/L).

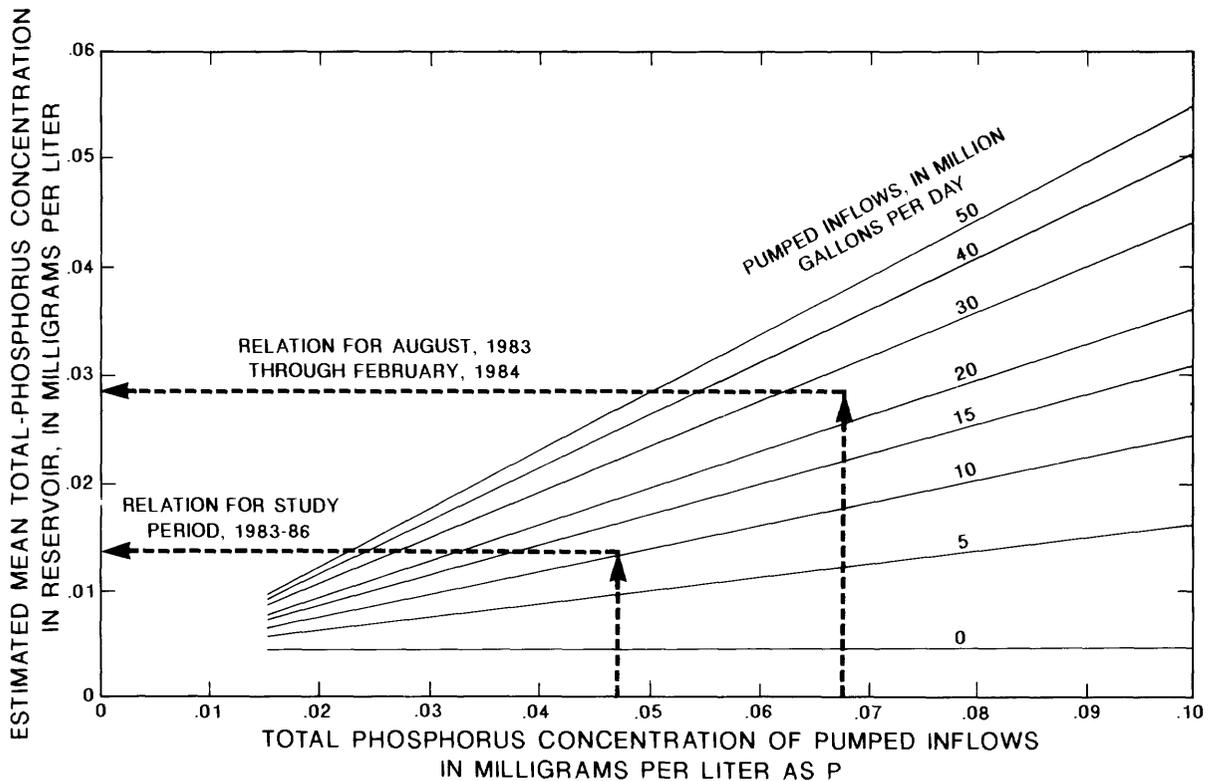


Figure 34.—Estimation of average total-phosphorus concentrations in Little Creek Reservoir at different rates of inflow and phosphorus concentrations in water from Chickahominy and Diascund Creek Reservoirs.

The mean total-phosphorus concentration in the upper 10 ft of water at sampling site 0204275430, which averaged 0.014 mg/L (table 29), was precisely estimated by the nomograph for the study period (fig. 34). The nomograph was based on a combined inflow from Chickahominy and Diascund Creek Reservoirs of 11.2 Mgal/d and a volume-weighted total-phosphorus concentration of 0.047 mg/L (fig. 34). Moreover, the nomograph accurately estimated the response of Little Creek Reservoir for relatively short time periods. For instance, large hydraulic and phosphorus loadings during the first 7 months of the study resulted in steadily increasing total-phosphorus concentrations in Little Creek Reservoir that peaked at 0.030 mg/L in early 1984 (fig. 32). Pumped inflows averaged 27 Mgal/d, and had a mean total-phosphorus concentration of 0.068 mg/L during the 7-month period. On the basis of these figures, the nomograph estimated a total-phosphorus concentration of 0.029 mg/L (fig. 34), which resembled the observed concentration. As expected, when inflows from Chickahominy and Diascund Creek

Reservoirs were substantially reduced in March and April 1984 (table 5), the total-phosphorus concentration in Little Creek Reservoir decreased rapidly (fig. 32).

The nomograph in figure 34 is used to evaluate the effects of different raw-water routing scenarios on the total-phosphorus concentrations of water in Little Creek Reservoir and raw water delivered to terminal reservoirs. For purposes of the five raw-water routing scenarios listed in table 35, concentrations of total phosphorus in water pumped from Chickahominy and Diascund Creek Reservoirs are assumed to average 0.049 and 0.028 mg/L, respectively, to reflect the concentrations measured during the study period. In each scenario, 33.7 Mgal/d is delivered to the terminal reservoirs. Withdrawals from Chickahominy and Diascund Creek Reservoirs are assumed to equal 17.5 and 12.6 Mgal/d, respectively, to reflect actual withdrawals during the study period (fig. 4). Withdrawals from Little Creek Reservoir differed in the five scenarios, depending on

Table 35.--Estimated effects of raw-water routing scenarios on total-phosphorus concentrations of water in Little Creek Reservoir and water delivered to terminal reservoirs

[mg/L, milligrams per liter; routing figures are in million gallons per day]

Routing	Scenario ¹				
	1	2	3	4	5
Chickahominy Reservoir to terminal reservoirs	17.5	0	9.8	0	17.5
Chickahominy Reservoir to Little Creek Reservoir	0	17.5	7.7	17.5	0
Diascund Creek Reservoir to terminal reservoirs	12.6	0	9.1	12.6	0
Diascund Creek Reservoir to Little Creek Reservoir	0	12.6	3.5	0	12.6
Little Creek Reservoir to terminal reservoirs	3.6	33.7	14.8	21.1	16.2
Predicted total-phosphorus concentration in Little Creek Reservoir, in mg/L	.005	.019	.019	.018	.010
Predicted total-phosphorus concentration of raw water delivered to terminal reservoirs, in mg/L	.036	.019	.028	.022	.030

¹ Scenarios based on total-phosphorus concentrations of 0.049 and 0.028 mg/L for Chickahominy and Diascund Creek Reservoirs, respectively, and the following raw-water management schemes: (1) no raw water routed through Little Creek Reservoir; (2) all raw water routed through Little Creek Reservoir; (3) raw-water routing for the study period, as illustrated in figure 3; (4) only Chickahominy Reservoir water routed through Little Creek Reservoir; and (5) only Diascund Creek Reservoir water routed through Little Creek Reservoir.

whether withdrawals from Chickahominy and Diascund Creek Reservoirs are routed through Little Creek Reservoir or pumped directly to terminal reservoirs. In each scenario, however, withdrawals from Little Creek Reservoir equal inputs from Chickahominy and Diascund Creek Reservoirs, plus 3.6 Mgal/d derived from the Little Creek Reservoir drainage basin. The estimated mean total-phosphorus concentration delivered to terminal reservoirs are the volume-weighted concentrations of the direct inputs from the three contributing reservoirs.

Scenario 1 depicts a raw-water route that bypasses Little Creek Reservoir (table 35). All withdrawals from Chickahominy and Diascund Reservoirs are delivered directly to terminal reservoirs, and 3.6 Mgal/d (with an estimated mean total-phosphorus concentration of 0.003 mg/L, as shown in figure 33) is delivered from Little Creek Reservoir. This routing scenario results in the highest total-phosphorus concentration delivered to terminal reservoirs (estimated at 0.036 mg/L) because Little Creek Reservoir is not used as a phosphorus sink. In scenario 2, all raw water is routed through Little

Creek Reservoir. This scenario delivers one of the lowest total-phosphorus concentrations to terminal reservoirs (estimated at 0.019 mg/L) because Little Creek Reservoir reaches maximum capacity as a phosphorus sink. Compared with scenario 1, routing all raw-water through Little Creek Reservoir (scenario 2) would result in nearly a 50 percent decrease in the amount of phosphorus delivered to terminal reservoirs, which would result in the following benefits: (1) improved quality of finished water, (2) reduced water-treatment costs, and (3) improved esthetic quality of the water in terminal reservoirs. Scenario 3 depicts the average raw-water routing observed during the study period (fig. 4), with about 37 percent of the withdrawals from Chickahominy and Diascund Creek Reservoirs pumped through Little Creek Reservoir. As expected, this scenario delivered water with an intermediate total-phosphorus concentration to terminal reservoirs, estimated at 0.028 mg/L.

A large decrease in total phosphorus delivered to terminal reservoirs (compared to scenario 1) can also be achieved by routing all water from Chicka-

hominy Reservoir through Little Creek Reservoir and delivering water from Diascund Creek Reservoir directly to terminal reservoirs, as depicted by scenario 4 (table 35). The estimated total-phosphorus concentration of delivered raw water is 0.022 mg/L for scenario 4. Routing only Diascund Creek Reservoir water through Little Creek Reservoir (scenario 5 in table 35) results in raw water with an estimated total-phosphorus concentration of 0.030 mg/L delivered to terminal reservoirs. Scenario 4 is more efficient than scenario 5 at removing phosphorus from raw water because the volume of water routed through Little Creek Reservoir, as well as its total-phosphorus concentration, is greater. As shown in figure 34, the quantity of phosphorus removed from raw water routed through Little Creek Reservoir increases as the inflow concentration increases.

Although scenario 4 does not quite achieve the same improvements in raw water delivered to terminal reservoirs as scenario 2 (table 35), it could represent a practical routing scheme. The low cost of routing raw water in scenario 4 compared with scenario 2 (because of less pumping between reservoirs) may compensate for delivery of a slightly larger total-phosphorus load to terminal reservoirs.

The quantity and quality of raw water routed through Little Creek Reservoir affects its trophic status. As more raw water is routed through Little Creek Reservoir, concentrations of total phosphorus and chlorophyll-*a* increase, and the depth of light penetration decreases. For example, if Little Creek Reservoir did not receive raw water from Chickahominy and Diascund Creek Reservoirs (scenario 1 in table 35), it would probably be classified as oligotrophic (Carlson, 1977), on the basis of an estimated total-phosphorus concentration of 0.005 mg/L. In contrast, if all raw water were routed through Little Creek Reservoir (scenario 2 in table 35), it would probably be classified as mesotrophic on the basis of an estimated total-phosphorus concentration of 0.019 mg/L.

SUMMARY

The Department of Public Utilities, city of Newport News, provides water to more than 330,000 people and to many industries on the York-James Peninsula in southeastern Virginia. Raw water for this supply was primarily withdrawn from Chickahominy, Diascund Creek, and Little Creek Reservoirs, and amounted to 33.7 Mgal/d

during 1983-86. Chickahominy, Diascund Creek, and Little Creek Reservoirs supplied 17.5, 12.6, and 3.6 Mgal/d, respectively. Because of its small drainage area, Little Creek Reservoir does not supply a large amount of raw water; however, it serves as a critical storage facility for water pumped from Chickahominy and Diascund Creek Reservoirs.

All three supply reservoirs are monomictic and have similar surface areas, ranging from about 1.5 to 2.0 mi². Chickahominy and Diascund Creek Reservoirs are both relatively shallow, with mean depths of 5.5 and 9.3 ft respectively; the mean depth of Little Creek Reservoir is 24 ft. Hydraulic retention times of Chickahominy, Diascund Creek, and Little Creek Reservoirs averaged about 9, 130, and 490 days, respectively, during the study. Depending on the volume of inputs from Chickahominy and Diascund Creek Reservoirs, however, the hydraulic retention time of Little Creek Reservoir varied considerably, ranging from 7 months (inflows averaging 30 Mgal/d) to 6 years (no inflows). During 1982-87, inflows to Little Creek Reservoir from Chickahominy Reservoir, Diascund Creek Reservoir, precipitation, and the terrestrial drainage basin averaged 7.7, 3.5, 3.2, and 2.3 Mgal/d, respectively. Of the total inflow to Little Creek Reservoir, 86 percent was withdrawn for treatment, 12 percent evaporated, and 2 percent was lost by leakage through the overflow structure.

An analysis of the hydrology of the three reservoirs produced two unexpected findings. First, the flow of Chickahominy River during dry periods has decreased significantly during the past 45 years. For example, the 7-day low-flow value declined 16 ft³/s between 1943 and 1987. This decline could have resulted from increased withdrawals of ground water in the area, or it could correspond to the elimination of sewage discharges that once supplemented flows in the Chickahominy River. Second, the hydraulic connection between Little Creek Reservoir and the surrounding aquifer is close. About 400 to 500 Mgal of water move in and out of ground-water storage when the reservoir water level is raised and lowered 10 ft. Thus, ground-water storage results in a reservoir capacity that is 5 to 10 percent greater than the bathymetry of Little Creek Reservoir would indicate.

On the basis of a mean total-phosphorus concentration of 0.049 mg/L, Chickahominy Reservoir is classified as eutrophic. This elevated total-phosphorus concentration results in elevated concentra-

tions of chlorophyll-*a*, periodically exceeding 40 µg/L. Elevated concentrations of chlorophyll-*a* only occur when the hydraulic retention time is adequate for development of an algal population. When average inflows exceeded 300 ft³/s for 20 days before sampling, which equated to a hydraulic retention time of about 8 days, reservoir chlorophyll-*a* concentrations averaged only 3 µg/L and never exceeded 6 µg/L. In contrast, when inflows averaged less than 300 ft³/s, the chlorophyll-*a* concentration in Chickahominy Reservoir averaged 21 µg/L, and periodically exceeded 50 µg/L.

One of the most common water-quality problems in Chickahominy Reservoir is periodic high concentrations of dissolved solids (specific conductance) resulting from flow reversals of salty water through the locks and over the spillway of Walkers Dam. Flow reversals that cause problems occur most frequently when the following conditions are met: (1) reservoir inflows have been less than 90 ft³/s for 3 months, thereby allowing salty water to reach Walkers Dam, (2) the reservoir water level is low because withdrawals have exceeded inflows for a number of days, and (3) the water level below the dam is high because of a storm surge and (or) an exceptionally high tide. The specific conductance of water in Chickahominy Reservoir exceeded 500 µS/cm on several occasions after flow reversals.

The trophic status of Diascund Creek Reservoir depends on the reservoir level. For example, when the reservoir level was drawn down 11 ft during the first 8 months of the study, Diascund Creek Reservoir was hypereutrophic, on the basis of a mean total-phosphorus concentration of 0.09 mg/L. Concentrations of phosphorus are higher during reservoir drawdown because of (1) decreased settling time for tributary inflows of phosphorus; (2) increased exposure of fine-grained, phosphorus-rich bottom sediments to resuspending forces; and (3) increased algal uptake of phosphorus directly from bottom sediments. At full pool, however, Diascund Creek Reservoir was mesotrophic/eutrophic, on the basis of a total-phosphorus concentration of 0.028 mg/L. Concentrations of phosphorus in tributaries of the reservoir are naturally high (averaging 0.1 mg/L) because they drain the Yorktown Formation, which is commonly phosphatic and glauconitic. Chlorophyll-*a* concentrations averaged 83 µg/L when Diascund Creek Reservoir was drawn down and averaged 14 µg/L when Diascund Creek Reservoir was at full pool.

Although the hypolimnion of Diascund Creek Reservoir becomes anaerobic during the first few months of stratification, the bottom sediments are not a net source of phosphorus. This lack of release, which is contrary to observations commonly reported in the literature, explains why phosphorus fails to accumulate in the hypolimnion during the summer, and why algal blooms do not develop after fall overturn. Bottom sediments in Diascund Creek Reservoir retain about 70 percent of the incoming phosphorus. Retention of phosphorus could result from (1) formation of refractory organophosphorus compounds, (2) sorption of phosphate ions to clays or iron complexes, or (3) formation of an iron phosphate mineral, such as vivianite. In contrast, bottom sediments retain little of the incoming nitrogen. Particulate organic nitrogen settling from the water column is efficiently recycled at the sediment-water interface and released as ammonium ions.

During the study, anaerobic bottom sediments of Diascund Creek Reservoir released large amounts of dissolved bicarbonate, ammonium, manganese, and iron to the hypolimnion, which resulted in concentrations that commonly reached 98, 0.7, 2.4, and 22 mg/L, respectively, near the sediment-water interface. Concentrations of bicarbonate, iron, and manganese in the hypolimnion appear to be in equilibrium with ferrous carbonate and manganous carbonate solid phases. Higher hypolimnial concentrations of iron and manganese generally do not cause water-quality problems. At fall overturn, these constituents oxidize, form a precipitate, and settle from the water column.

The trophic state of Little Creek Reservoir depends on the quantity and quality of inflows from Chickahominy and Diascund Creek Reservoirs. If inflows were negligible, Little Creek Reservoir would probably be classified as oligotrophic on the basis of an estimated total-phosphorus concentration of 0.005 mg/L; however, with inflows averaging 11.2 Mgal/d during the study, Little Creek Reservoir was mesotrophic on the basis of a mean total-phosphorus concentration of 0.014 mg/L. If all raw water from Chickahominy and Diascund Creek Reservoirs (30.1 Mgal/d) were routed through Little Creek Reservoir, it would be classified as mesotrophic, on the basis of an estimated total-phosphorus concentration of 0.019 mg/L.

Anaerobic bottom sediments in Little Creek Reservoir are a large source of dissolved bicarbonate, ammonium, manganese, and iron to the

hypolimnion, as they are in Diascund Creek Reservoir. By late summer, hypolimnial concentrations of dissolved iron and manganese commonly exceeded 10 and 1 mg/L, respectively, near the sediment-water interface. Elevated hypolimnial concentrations of iron and manganese in Little Creek Reservoir do not cause water-supply problems because these constituents are quickly lost from the water column at fall overturn.

On the basis of a detailed mass-balance analysis, Little Creek Reservoir is a significant sink for several constituents. Raw water routed through Little Creek Reservoir lost 20 and 70 percent of its load of nitrogen and phosphorus, respectively. Retention of phosphorus in anaerobic sediments is probably related to sorption or reaction with iron. Loads of dissolved silica, iron, and manganese were reduced by 60, 55, and 35 percent, respectively. The quality of raw water is measurably improved when withdrawals from Chickahominy and Diascund Creek Reservoirs are routed through Little Creek Reser-

voir. For example, routing all raw water through Little Creek Reservoir would reduce the load of phosphorus delivered to terminal reservoirs by nearly 50 percent, which would help alleviate some of the water-quality problems in these reservoirs.

On the basis of N:P ratios (by weight) that exceeded 7.0, algae in all three reservoirs are phosphorus-limited. Phosphorus concentration defines the upper bound for chlorophyll-*a* concentrations in each reservoir, and indirectly affects the mean reservoir chlorophyll-*a* concentration as well. Consequently, increasing the phosphorus concentration in any of these reservoirs would result in increased chlorophyll-*a* concentrations and decreased Secchi-disk transparency depths. During the study period, chlorophyll-*a* concentrations averaged 21, 14, and 8 µg/L in Chickahominy (base flow), Diascund Creek, and Little Creek Reservoirs, respectively, and were approximately proportional to the mean total-phosphorus concentrations in these reservoirs.

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