WATER-QUALITY ASSESSMENT OF ARVADA RESERVOIR, DENVER METROPOLITAN AREA, COLORADO

By Linda J. Britton and Neville G. Gaggiani

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DONALD PAUL HODEL, Secretary

U.S. GEOLOGICAL SURVEY

Dallas L. Peck, Director

For additional information
write to:

District Chief
U.S. Geological Survey
Water Resources Division
Box 25046, Mail Stop 415
Federal Center
Denver, CO 80225-0425

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CONVERSION FACTORS

Metric units (International System) in this report may be converted to inch-pound units by using the following conversion factors:

Multiply metric unit	*by*	*To obtain inch-pound unit*
centimeter (cm)	0.3437	inch
meter (m)	3.281	foot
micrometer (μm)	0.00003937	inch
millimeter (mm)	0.03937	inch
cubic meter (m^3)	35.31	cubic foot

Inch-pound units in this report may be converted to metric units (International System) by using the following conversion factors:

Multiply inch-pound unit	*by*	*To obtain metric unit*
acre-foot (acre-ft)	0.001233	cubic hectometer
cubic foot per second (ft^3/s)	0.02832	cubic meter per second
foot (ft)	0.3048	meter
mile (mi)	1.609	kilometer

Degree Celsius (°C) may be converted to degree Fahrenheit (°F) by using the following equation:

$$°F=9/5(°C+32)$$

The following terms and abbreviations also are used in this report:

cells per milliliter (cells/mL)
microgram per gram (μg/g)
microgram per liter (μg/L)
microsiemens per centimeter at 25 degrees Celsius (μS/cm)
milligram per kilogram (mg/kg)
milligram per liter (mg/L)
organisms per cubic meter (organisms/m^3)

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ABSTRACT

Construction of Arvada Reservoir began during 1982; the reservoir, which was completely filled in May 1984, is a raw-water-supply source for the city of Arvada. Water-quality data were collected from Arvada Reservoir and its major inflows, Ralston Creek and Croke Canal, to assess the physical, chemical, and biological quality of the reservoir; to evaluate the effect of various source waters on the reservoir; to estimate the trophic state of the reservoir; and to provide information to assist with future water treatment and recreational and esthetic management of the reservoir. Water temperature, dissolved oxygen, specific conductance, and pH were measured at all sampling sites; measurements in the reservoir were made at every 0.5 to 1 meter of depth, and measurements in Ralston Creek and Croke Canal were made near the centroid of flow. Secchi-disk-depth measurements also were made in the reservoir. In addition, water samples were analyzed for concentrations of major-chemical constituents, nutrients, trace elements, and uranium; for densities and relative abundance of phytoplankton; for concentrations of chlorophyll *a*; for determination of algal-growth potential; and for densities and relative abundance of zooplankton.

Data were collected at five sites in Arvada Reservoir, one site in Ralston Creek, and two sites in Croke Canal. The study began in June 1983 and continued through September 1985.

The reservoir was thermally stratified on most sampling dates, generally from April through September during the study period. Dissolved-oxygen concentrations ranged from 0 to 12.0 milligrams per liter, and the reservoir was anaerobic below the 10-meter depth during most of the summer. Specific-conductance measurements indicated that the reservoir has small dissolved-solids concentrations (generally less than 250 milligrams per liter), and effects from Croke Canal were apparent as indicated by decreased specific conductance within a 4-meter zone where the Croke Canal pipeline extends into the reservoir. However, water from Croke Canal was pumped to the reservoir only during 2 months in 1985. Secchi-disk-depth measurements ranged from 0.9 to 5.5 meters and generally increased during the study period, possibly because of decreases in nonalgal turbidity after the reservoir was filled.

The results of chemical analyses indicated that water from the reservoir generally is of suitable quality for a raw-water-supply source and for maintenance of aquatic life. The exceptions were that dissolved-manganese concentrations occasionally exceeded secondary drinking-water standards, and total-mercury and total-zinc concentrations occasionally exceeded criteria for aquatic life. In addition, total-uranium concentrations occasionally exceeded standards specified for the reservoir. Total-nitrogen and total-phosphorus concentrations were small, and ratios of total-nitrogen and total-phosphorus concentrations generally were larger than 26:1 in the reservoir.

The phytoplankton community was diverse in the reservoir; densities ranged from 1,400 to 29,000 cells per milliliter, and diversity-index values were as much as 4.65. Diatoms dominated most often; whereas, blue-green algae rarely dominated. Chlorophyll *a* concentrations ranged from 0.0 to 20.4 micrograms per liter, and relations between phosphorus and chlorophyll *a* indicated that predicted values of chlorophyll *a*, based on measured values of phosphorus, rarely were within the confidence limits of mean measured chlorophyll *a* concentrations. Algal-growth-potential determinations indicated that spikes of phosphorus, as well as nitrogen, increased algal growth. Therefore, phosphorus-chlorophyll *a* relations may be affected by nitrogen as a limiting nutrient, phytoplankton patchiness, chlorophyll *a* content of algal cells, and inherent variability in the assumptions used to develop the relation.

Calculations of trophic state were based on three types of data: Secchi-disk-depth measurements, total-phosphorus concentrations, and chlorophyll *a* concentrations. Trophic-state-index values predicted using Secchi-disk-depth measurements and chlorophyll *a* concentrations were similar. Values predicted using total-phosphorus concentrations were markedly different; this indicates that other factors, possibly nitrogen limitation, affect light transparency and algal biomass.

INTRODUCTION

During 1982, Blunn Reservoir, now called Arvada Reservoir, was constructed in the northwestern part of the Denver metropolitan area, Colo. (fig. 1), and filling began during 1983 and continued during 1984. Water stored in the reservoir is used as an additional source of drinking water for the city of Arvada during the peak demand period, May through September. Because it is near a densely populated urban environment, the water quality of the reservoir could become a potential issue, especially if watershed use substantially increases and the reservoir becomes used for multiple purposes.

During the spring of 1983, a study of Arvada Reservoir was begun by the U.S. Geological Survey in cooperation with the city of Arvada. Because the study began when the reservoir was being filled and before the water-treatment plant was completed, an opportunity existed to monitor the water quality prior to use of the reservoir as a drinking-water supply. In many instances (Duthie and Ostrofsky, 1982), filling and newly impounded reservoirs are in a nonsteady or nonequilibrium state and demonstrate immediate trophic surges that are not applicable to long-term trophic conditions. However, this condition occurs predominantly among those reservoir basins that are densely vegetated and subsequently flooded. The flooded drainage basin for Arvada Reservoir was dry agricultural land that would not be considered a potential internal nutrient source. Nonetheless, the question of the reliability of forecasting future water quality of the reservoir, based on possible nonsteady-state conditions during the filling period was considered in the study. In addition, background data on ambient water quality of inflows was needed and used to assess potential effects on future water stored in the reservoir.

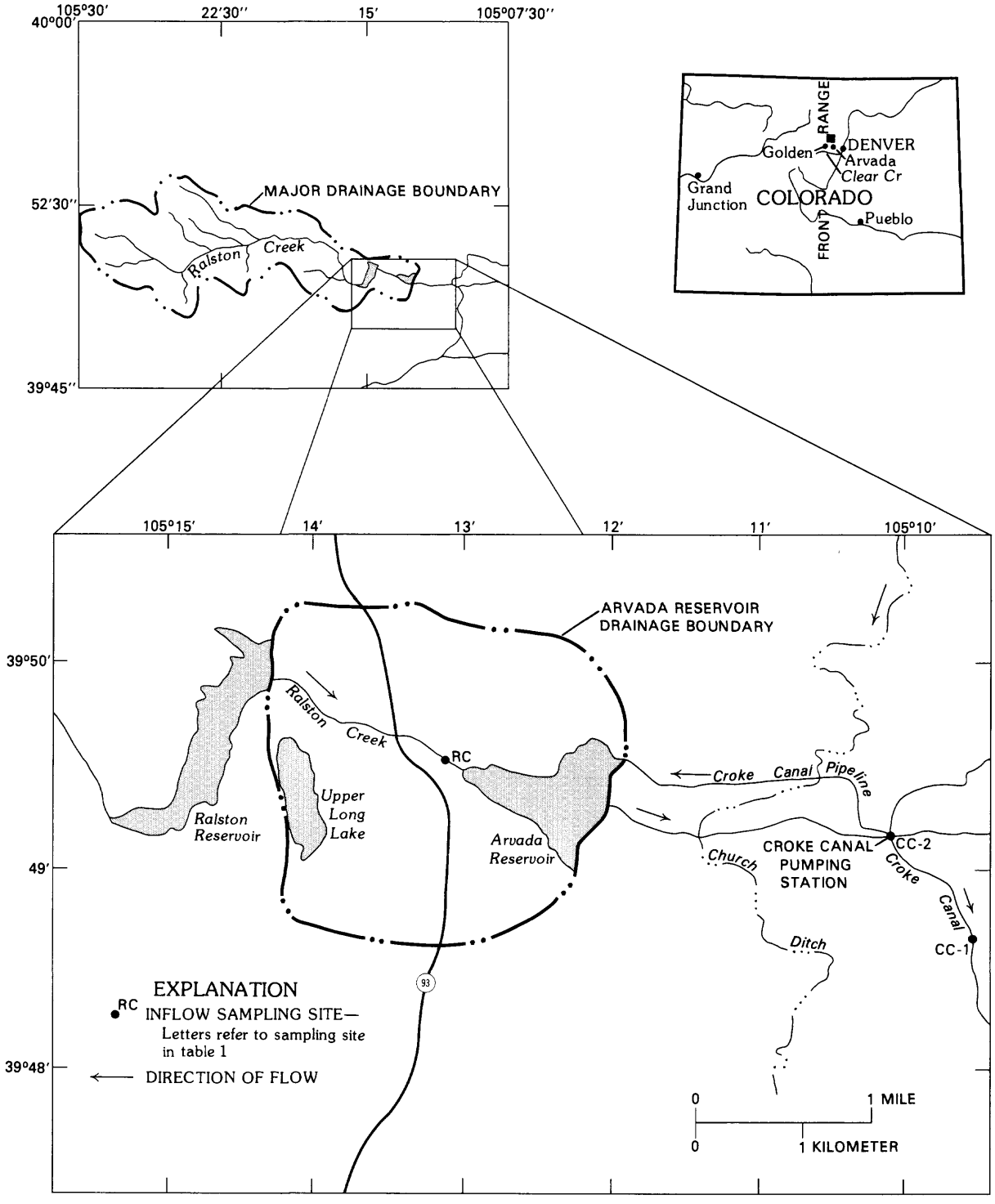

Figure 1.--Major drainage basin, Arvada Reservoir drainage basin, and inflow sampling sites.

The study of Arvada Reservoir was designed to achieve the following objectives:

1. Assess the physical, chemical, and biological quality of Arvada Reservoir;
2. Estimate the quantity of algal growth that the reservoir may support, as well as the substances that may limit or stimulate growth;
3. Evaluate the effect of various source waters on the reservoir and estimate possible water-quality changes after addition of Croke Canal water to the reservoir;
4. Estimate the trophic state of the reservoir; and
5. Identify future data-collection efforts necessary to monitor possible degradation of the reservoir.

In addition, water-quality data collected and evaluated from the reservoir will contribute to a data base to be used for evaluating lakes in the Denver metropolitan area.

Purpose and Scope

This report describes the qualitative and quantitative results of physical, chemical, and biological water-quality data collected from Arvada Reservoir and its inflows, Ralston Creek and Croke Canal. Water temperature, dissolved oxygen, specific conductance, and pH were measured at all sampling sites; measurements in the reservoir were made at every 0.5 to 1 m of depth, and measurements in Ralston Creek and Croke Canal were made near the centroid of flow. Secchi-disk-depth measurements also were made in the reservoir. In addition, water samples were analyzed periodically for concentrations of major-chemical constituents, nutrients, trace elements, and uranium; for densities and relative abundance of phytoplankton; for concentrations of chlorophyll a; for determination of algal-growth potential; and for densities and relative abundance of zooplankton.

The report describes these data in terms of objectives 1 through 5 listed in the "Introduction" section for five sites in Arvada Reservoir, one site in Ralston Creek, and two sites in Croke Canal. The data collected for this study during June 1983 through September 1985 are in a report by Britton and Gaggiani (1986). This report may be useful to city planners, water-treatment-plant operators, and local government personnel.

Acknowledgments

The authors extend their appreciation to Scott Daniels and Leonard Rossi of the city of Arvada for their assistance in collecting the samples. In addition, the city of Arvada provided sampling equipment during the last year of the study, which is gratefully acknowledged.

DESCRIPTION OF STUDY AREA AND OPERATION OF RESERVOIR

Arvada Reservoir is about 25 mi northwest of the city of Denver near the foothills of the Front Range in Jefferson County. The reservoir is an impoundment on Ralston Creek, about 1 mi downstream from Ralston Reservoir (fig. 1). Impoundment of water began during May 1982 with water released from Ralston Reservoir. The treatment and distribution of water from Arvada Reservoir began in June 1985.

The two primary sources of water for Arvada Reservoir are Ralston Creek and Clear Creek. In addition, water enters Arvada Reservoir from Upper Long Lake through a pipeline to Ralston Creek downstream from Ralston Reservoir. Occasionally water can and has entered Arvada Reservoir from Gross Reservoir (northwest of Ralston Reservoir on South Boulder Creek), which receives water from the Moffatt Tunnel (a transmountain diversion). This water can either: (1) Pass from Gross Reservoir to Ralston Reservoir and flow downstream to Arvada Reservoir, or (2) enter a diversion canal and bypass Ralston Reservoir to the Ralston Reservoir spillway and flow downstream to Arvada Reservoir. Clear Creek water enters the reservoir from a pumping station located downstream from the reservoir at the confluence of Ralston Creek and Croke Canal. Clear Creek water can be diverted through Church Ditch, Farmers Highline Canal, and Croke Canal to the pumping station. The timing, source, and volume of flow into Arvada Reservoir are subject to normal precipitation and runoff patterns, allocation of water rights, and ultimately management decisions by the city of Arvada and the Denver Water Board.

Treatment of water from Arvada Reservoir for distribution to consumers in the city of Arvada began on June 5, 1985. The Croke Canal pumping station began operation during June 1985 and began pumping water to Arvada Reservoir through a pipeline on June 20, 1985. The pumping station operated until July 22, 1985. Between June 20 and June 30, 1985, about 189 acre-ft of Croke Canal water entered the reservoir, and between July 1 and July 22, 1985, about 502 acre-ft of water entered the reservoir. Water can be withdrawn from the reservoir through a multiple-release system (outflow in fig. 2) consisting of three intake gates. At full pool, the midpoints of the intake gates are located about 19, 34, and 49 ft below the water surface.

SAMPLING-SITE LOCATIONS

After an initial reconnaissance of Arvada Reservoir was made on June 17, 1983, three locations in the reservoir were selected as sampling sites (sites A, B, C, fig. 2). These sites were selected based on depth characteristics to ensure that deep-, medium-, and shallow-depth sites were included. The proximity of the sites to the major inflow from Ralston Creek (fig. 2) and the future inflow from the Croke Canal pipeline (fig. 2) also was a factor. Because buoys were not established in the reservoir during 1983, sampling was not always at exactly the same location. Hence, depths of sampling vary at a site during the 1983 sampling period. In addition, even after buoys were established at sampling sites, intense winds caused some movement of the buoys, and, thus, depths also varied slightly during the 1984-85 sampling periods. Therefore, the sites delineated in figure 2 are approximate and only relative to the outlined contours.

Prior to the 1984 sampling period, permanent buoys were established at the sampling sites in the reservoir. After March 1984, site B was discontinued and site D (fig. 2) was established closer to the inflow from Ralston Creek (table 1). Data were collected at sites A, C, and D during most of the 1984 sampling period. During 1985, sites C and D were discontinued, and site E (fig. 2) was established. Site E was established to identify possible short-circuiting effects upstream from site A because of input of water to the reservoir from Croke Canal. Data were collected at sites A and E during the 1985 sampling period.

During 1983, in addition to the data-collection sites in the reservoir, a sampling site (RC, fig. 1) was located in Ralston Creek, the major inflow to the reservoir, and a sampling site (CC-1, fig. 1) was located in Croke Canal. During the later part of the 1983 sampling period, site CC-1 in Croke Canal was moved to its permanent diversion site (CC-2, fig. 1). Although two sites were sampled in Croke Canal, the data are stored according to one identification number and discussed as one site in this report. The sampling site name and number, location, identification number [for access to the U.S. Geological Survey National Water Data Storage and Retrieval System (WATSTORE)], period of data collection for each sampling site, and ranges and means of deepest sampling points are listed in table 1.

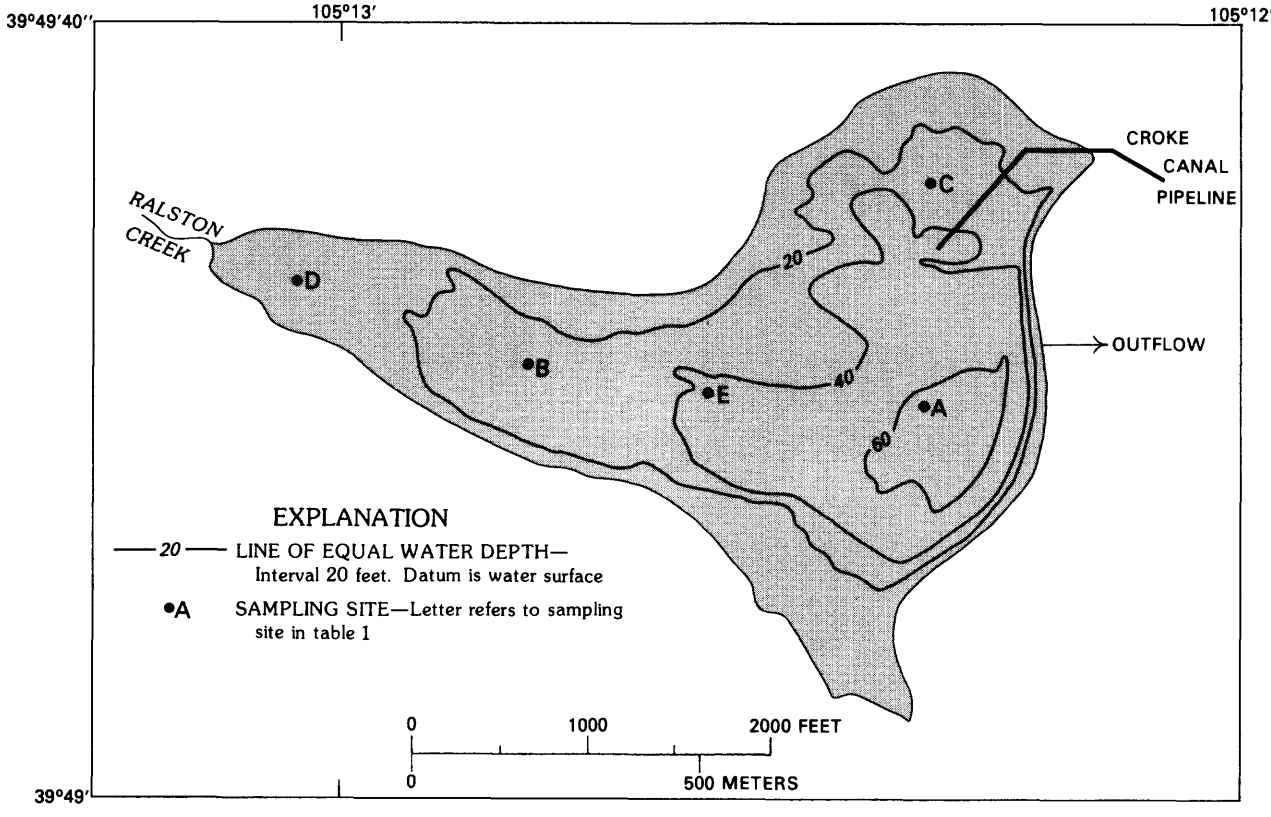

Figure 2.--Sampling sites in Arvada Reservoir

Table 1.--*Sampling sites*

[---, not applicable]

Sampling site name and number (figs. 1 and 2)	Location		Identification number	Period of data collection	Deepest sampling point (meters)	
	Latitude	Longitude			Range	Mean
Arvada Reservoir						
A	39°49'19"	105°12'21"	394919105122100	06-17-83 - 09-25-85	14.0 - 20.0	17.5
B	39°49'23"	105°12'46"	394923105124600	06-17-83 - 03-21-84	4.5 - 7.5	6.0
C	39°49'31"	105°12'20"	394931105122000	06-17-83 - 09-26-84	7.0 - 17.0	11.4
D	39°49'25"	105°13'01"	394925105130100	03-01-84 - 09-26-84	6.0 - 7.5	6.7
E	39°49'21"	105°12'30"	394921105123000	04-10-85 - 09-25-85	9.5 - 13.5	11.0
Inflows						
Ralston Creek (RC).	39°49'34"	105°13'17"	394934105131700	06-17-83 - 09-25-85	---	---
Croke Canal (CC-1).	39°48'46'	105°09'57"	394846105095700	07-05-83 - 07-27-83	---	---
Croke Canal (CC-2).	39°49'14"	105°10'23"	394846105095700	09-13-83 - 07-17-85	---	---

METHODS OF DATA COLLECTION AND ANALYSES

The results of all water-quality data collected during this study are in Britton and Gaggiani (1986). In-situ measurements of water temperature, dissolved oxygen, specific conductance, and pH were recorded at every 0.5 to 1 m of depth in the reservoir using a multiparameter unit. These in-situ measurements also were made in Ralston Creek and Croke Canal near the centroid of flow. Light transparency in the reservoir was measured using a Secchi disk, a black-and-white, flat, circular disk about 20 cm in diameter. The Secchi-disk measurement consisted of recording the depth at which the disk disappeared from view in a shaded area of water surface (Welch, 1948, p. 159). These measurements generally were made biweekly from June 1983 through September 1983, biweekly from May 1984 through September 1984, weekly from May 1985 through September 1985, monthly during the spring, and periodically during the winter, throughout the study period. Frequency of sampling generally was controlled by allocation of funds. On some sampling dates, a property was not measured because of equipment failure.

Water samples for analyses of concentrations of major-chemical constituents, trace elements, and uranium were collected from or near the water surface (0.5-m depth) of the reservoir and periodically from near the bottom of the deepest site, during periods of thermal stratification. Water samples for analyses of concentrations of nutrients were collected at 1-m depth intervals throughout the euphotic zone and composited (to correspond with the method used for collection of phytoplankton samples), or samples were collected from or near the surface (0.5-m depth) of the reservoir (to correspond with a change in the method used for collection of phytoplankton samples). The euphotic zone is defined in this study as twice the measured Secchi-disk depth (Moss, 1980). The samples were collected using a standard van Dorn-type polyethylene water-sampling bottle and using the procedures described by Brown and others (1970). Water samples for chemical analyses were collected near the centroid of flow in Ralston Creek and Croke Canal. Bed-material samples for analyses of nutrients and selected trace elements were collected from the reservoir using a weighted Ekman grab and using the procedures described by Guy and Norman (1970).

During 1983, 1984, and April 1985, water samples for phytoplankton, chlorophyll *a*, and algal-growth-potential analyses were collected and composited using the standard water-sampling bottle at 0.5- to 1-m depth intervals throughout the euphotic zone. At site A, during the remainder of 1985, water samples for phytoplankton and chlorophyll *a* analyses were collected from or near: (1) The water surface (0.5-m depth), (2) middle depth of the euphotic zone (equal to the Secchi-disk depth), and (3) bottom depth of the euphotic zone (twice the Secchi-disk depth). In addition, samples were collected at 1-m depth intervals throughout the euphotic zone and composited. Each of these four samples was analyzed separately. Although several methods were used for collection of phytoplankton and chlorophyll *a* samples, only the results of composited samples are discussed in this report. At site E, during 1985, water samples for chlorophyll *a* analyses were collected at 1-m depth intervals throughout the euphotic zone and composited. Water samples for chlorophyll *a* and algal-growth-potential analyses were collected near the centroid of flow in Ralston Creek and Croke Canal.

Water samples for zooplankton analyses were collected using a standard plankton net (80-μm mesh) that was: (1) Vertically towed from the bottom of the euphotic zone to the water surface, and referenced as photic tows (1983, March 1984, and April 1985 samples); or (2) vertically towed from the bottom of the reservoir to the water surface, and referenced as total-depth tows (June 1984 through September 1984 samples). On most dates, replicate samples were collected and analyzed separately. After April 1985, two samples for zooplankton analyses were collected from photic tows and two samples were collected from total-depth tows. Each replicate sample was analyzed separately. Zooplankton densities are reported as number of organisms per cubic meter, and the volume sampled is determined using the following formula:

$$V = \pi r^2 d,$$

where V = volume, in cubic meters;
r = radius of the mouth of the net, in centimeters; and
d = tow length through the water column, in meters.

Zooplankton densities were based on the assumption that the filtration efficiency of the net was 100 percent. The data need to be interpreted as relative density values rather than true density values, because the net's true efficiency probably was less than 100 percent (Rawson, 1956). In addition, the total-depth-tow values are questionable because, compared to the photic-tow values collected on the same dates, the total-depth-tow values were always less than the photic-tow values. The net possibly was towed to the water surface too quickly, causing backflushing and decreased densities of zooplankton. Therefore, in most instances (except when only total-depth-tow values were available) only the results of photic-tow samples are discussed in this report.

The chemical samples were analyzed at the U.S. Geological Survey laboratory in Arvada, Colo., using the procedures described by Guy (1969) and Fishman and Friedman (1985). Specifically, trace elements analyzed to determine total concentrations were done using the total-recoverable method. The biological samples were analyzed at a private laboratory in Littleton, Colo., using the procedures described by Greeson and others (1977). Specifically, the chlorophyll *a* analyses were done using acetone extraction and the spectroscopy method, with a correction for pheophytin. The algal-growth-potential determinations were done using *Selenastrum capricornutum* as the test organism following the procedures described by the U.S. Environmental Protection Agency (1971). Finally, diversity-index values were calculated for the phytoplankton samples collected in the reservoir. The diversity index, as proposed by Shannon and Weaver (1949), was used and the diversity per individual (H') is:

$$-\sum_{i=1}^{s} (n_i/N) \log 2\ n_i/N,$$

where s = total number of taxa;
n_i = number of individuals of each single taxa in the sample; and
N = total number of individuals of all taxa in the sample.

SIGNIFICANCE OF MEASURED WATER-QUALITY VARIABLES

The following section is presented as background for those readers who are not familiar with the properties and processes of lakes. Although this report is about a reservoir, the term lake more commonly is used in general literature and, therefore, sometimes is used in this report when discussing general properties and processes. The term lake also is used in reference to certain cited studies that dealt primarily with lakes rather than reservoirs. Data derived from studies of lakes generally are applicable to reservoirs. Those readers who are more familiar with the properties and processes of lakes and interpretation of measurements of water-quality variables may wish to continue reading with the "Results of Water-Quality Analyses of Arvada Reservoir" section of this report.

Water Temperature

One of the most important properties of a lake is water temperature because it is a major controlling factor for physical, chemical, and biological processes. For example, the solubility of dissolved oxygen is inversely related to temperature--that is, the warmer the water temperature, the less oxygen may be dissolved in the water. This fact has obvious biological implications because at warmer water temperatures, organisms have an increased metabolic rate but have less oxygen available for their physiological needs. Also, water-temperature measurements generally are used as a determinant or guide for the collection of other limnological data because differences in chemical quality in a lake commonly are related to temperature differences.

The density of freshwater primarily is temperature dependent; the effect of dissolved and suspended materials on density is relatively less important, except where chemical-density stratification occurs. Freshwater is unique in that its maximum density occurs at about 4 °C. At water temperatures either greater or less than 4 °C, the density of water decreases. This property of water is a principal factor affecting thermal stratification of lakes. Thermal stratification is a natural condition whereby less dense, warm water overlies more dense, cold water in deeper lakes during the warmer months, although stratification also can occur under ice cover. Wind mixing of the water layers is hindered by the density differences. The isolation of the water layers, caused by temperature and light-penetration differences, enables various chemical and biochemical transformations to occur and results in differences in quality of the water layers.

Most temperate-zone lakes have a regular pattern of seasonal mixing and stratification. Warming of the surface water during the summer produces thermal or density differences between the upper and lower water layers. Most of the radiant energy is absorbed in the water near the surface of the lake, and this water is heated more quickly and becomes less dense than water near the bottom. The lake becomes thermally stratified and is in the summer-stagnation period (Hutchinson, 1957). The result is the formation of three water layers. The upper layer containing the warmest water is the *epilimnion* (fig. 3). The central transition layer, in which water temperature decreases rapidly with depth, is the *metalimnion*. The *thermocline*, as defined by Wetzel (1983) is the plane of maximum rate of decrease of temperature with depth. The lower layer containing the coldest water is the *hypolimnion*.

As air temperature decreases during autumn, the surface-water temperature decreases. The surface water becomes more dense than the lower water, sinks, and is replaced by warmer, less dense water from below. This mixing of water layers continues until all the water has the same temperature and density. During this season of the year, the lake is in the autumn-circulation period, and even slight winds may mix the entire water mass and oxygenate the deeper water. Continued cooling during the winter may decrease the surface-water temperature until the lake surface is covered with ice. During this season, the lake normally is stratified having warmer (as much as 4 °C) water on the bottom and colder (0 °C) water immediately below the ice. When the ice melts during the spring, the surface water warms until all the water again has about the same temperature and density. This is the spring-circulation period. The

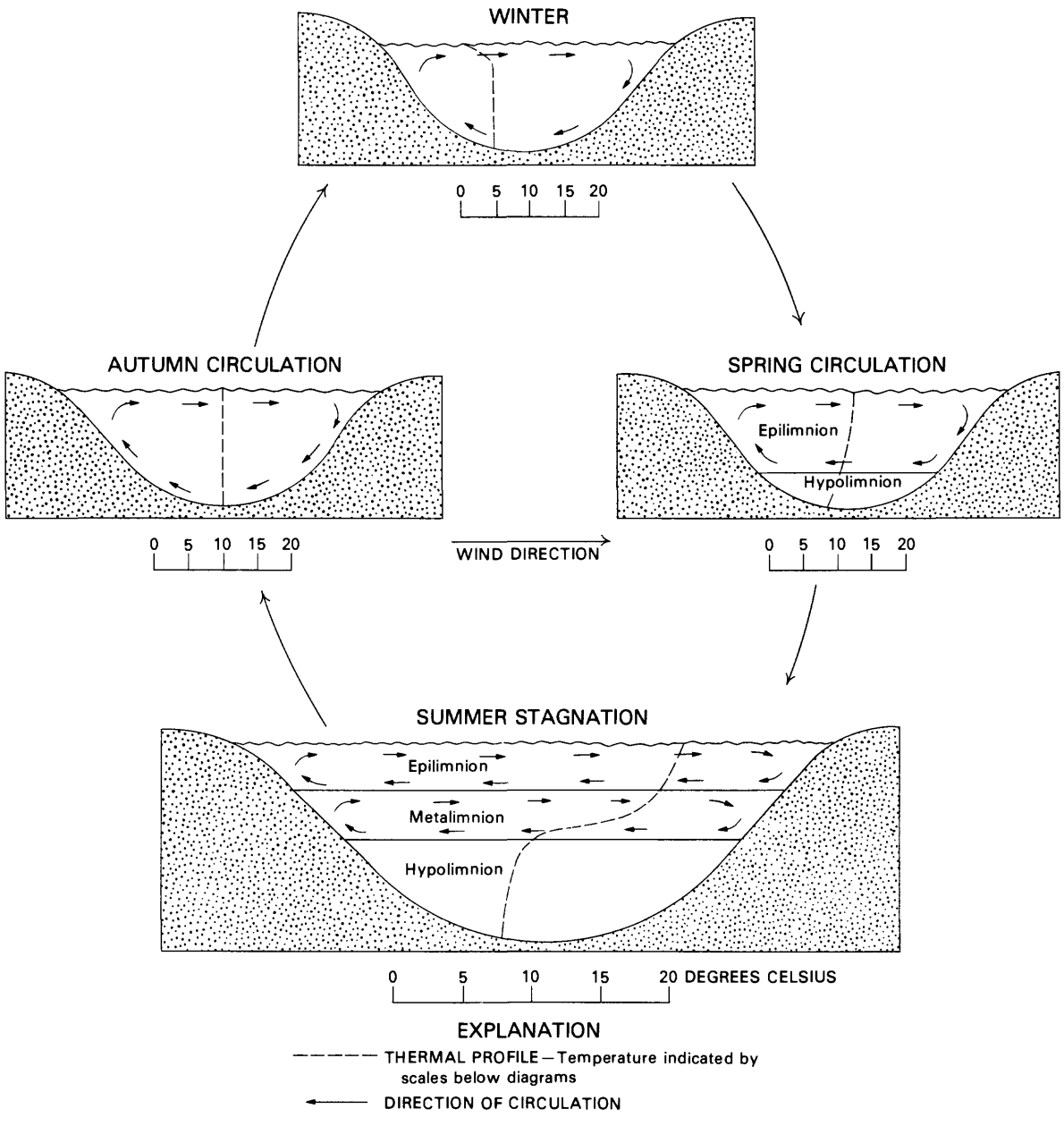

Figure 3.--Seasonal thermal profiles of a typical temperate-zone lake.

result of each circulation period is that nutrients released into the hypolimnion from decaying organic matter in the bottom sediments can become available for algal growth and production throughout the lake. Often this internal supply of nutrients produces algal blooms during the spring, summer, or autumn.

There are many variations in the thermal regime of lakes. For example, many shallow lakes become thermally stratified during periods of calm but may be completely mixed at any time of the year by moderate winds. Other lakes are continuously mixed and thermal stratification does not occur.

Dissolved Oxygen

The dissolved-oxygen concentration is another important constituent in water. The concentration of oxygen that dissolves in water is affected by the temperature, the atmospheric pressure, and the salinity of the water. Oxygen solubility is inversely related to temperature and salinity--the warmer and more saline the water, the smaller the dissolved-oxygen concentration. The relation of oxygen solubility to atmospheric pressure is direct. The maximum theoretical dissolved-oxygen concentration for a specific temperature, pressure, and salinity is the saturation concentration, but supersaturated or undersaturated conditions may exist. Supersaturated conditions occur from photosynthetic activity in excess of losses to the atmosphere. Undersaturated conditions can result from respiration and decomposition processes which exceed production of oxygen.

The dissolved-oxygen concentration primarily is determined by the physical interaction of the water with the atmosphere (aeration from wave action) and by biological activity. During photosynthesis, phytoplankton and other green plants in the water use carbon dioxide from the water, synthesize carbohydrates, and release oxygen. The released oxygen dissolves and may increase the total oxygen to a concentration greater than the saturation or equilibrium concentration. The water then is supersaturated with oxygen.

The associated process whereby dissolved oxygen is used by organisms is called respiration. Undersaturated or depleted dissolved-oxygen concentrations occur when aerobic bacteria or respiring plants and animals use the dissolved oxygen at a faster rate than it can be replaced by physical or biological activities.

When thermal stratification occurs in a lake, a density barrier is formed that prevents transfer of dissolved oxygen from the surface water to the bottom water. The bottom water then may lose, by natural respiration of plants and animals and chemical oxidation, all the dissolved oxygen gained during the last mixing period. A dissolved-oxygen minimum can occur in the metalimnion due to rapid rates of decomposition by bacteria that sometimes accumulate in this layer or because of respiration by plankton. When large quantities of decomposable organic wastes are introduced into lakes, oxygen depletion can occur throughout the lake, causing the death of aquatic organisms--primarily those that have the greatest oxygen requirements, such as fish. A dissolved-oxygen maximum also may occur in the metalimnion due to locally increased populations of phytoplankton that accumulate in this layer of increased density. Increased dissolved-oxygen concentrations in the bottom of some lakes probably result from inflow of stream water containing relatively large dissolved-oxygen concentrations.

Different types of fish require different concentrations of dissolved oxygen to survive. Cold-water fish, such as trout, require larger concentrations of dissolved oxygen (more than 5 mg/L) than do warm-water fish, such as carp and catfish (more than 4 mg/L) (U.S. Environmental Protection Agency, 1986b).

Specific Conductance

Specific conductance is a measure of the ability of water to conduct an electric current. In natural water, specific conductance results primarily from dissolved ions and, thus, provides an estimate of the concentration of dissolved solids. Frequent measurement of dissolved-solids concentration in an individual lake usually is not necessary because changes in dissolved-solids concentrations can be detected by measuring the specific conductance.

Differences in specific conductance throughout the water column are sometimes used to distinguish water layers in lakes. Although only small vertical differences usually occur in specific conductance, a larger specific conductance sometimes will be measured near the bottom of a lake because of dissolution of suspended particles. Dissolved-oxygen depletion and reducing conditions in the hypolimnion are mechanisms that contribute to the dissolution of suspended particles. Similar increases in specific conductance can be restricted to the metalimnion. These increases may result from relatively rapid rates of decomposition caused by bacterial populations.

pH

In most freshwater, pH, which is a measure of the hydrogen-ion (H^+) concentration (activity), usually is equal to the hydrogen-ion concentration. Values on the pH scale range from 0 to 14 and determine whether a solution is acidic or basic (alkaline). A solution that has a pH of 7 is neutral and contains 10^{-7} moles per liter of hydrogen ions. Solutions that have pH ranging from 0 to 7 are acidic, and solutions that have pH ranging from 7 to 14 are basic. The solubility of many chemical constituents and the biological activity of many organisms in water are pH dependent; thus, pH is an important factor in controlling concentrations of many chemical constituents and populations of organisms in water. The pH also may affect the suitability of water for various uses. A range of pH between 6.5 and 9.0 is necessary for most fish and for water used as a source of public-water supplies (U.S. Environmental Protection Agency, 1986a).

The pH of water in lakes may be altered by photosynthesis and respiration, as well as by inflowing water from surface- and ground-water sources. The use of carbon dioxide by plants during photosynthesis increases the pH of the water; whereas, the release of carbon dioxide during respiration decreases the pH. Therefore, pH values commonly are greatest near the water surface, where photosynthesis occurs, and least near the lake bottom, where respiration predominates. The pH sometimes decreases to a minimum in the metalimnion where settling materials are decomposed by large concentrations of bacteria.

Light Transparency

Light transparency is the ability of water to transmit light. Photosynthetic activity is restricted to depths where sufficient light is available. Interference resulting from materials that scatter or absorb light, such as algae, suspended sediment, or naturally colored materials in solution, will

confine the transmission of light to the uppermost layer of water. Usually, there is an inverse relation between algal density and light transparency, and decreased light penetration into waterbodies usually is the result of increased algal densities.

Light transparency, measured using a Secchi disk, is one of the simplest and most commonly made limnological measurements. Secchi-disk-depth measurements have been used as an index of trophic state and of plankton density in lakes. In reservoirs, light-transparency measurements commonly are used as indicators of suspended-sediment concentrations.

Chemical and Biological Constituents

Chemical constituents in water consist of dissolved and suspended minerals, such as calcium and sulfate; gases, such as oxygen and carbon dioxide; and organic compounds, such as pesticides. The presence and concentrations of the various chemical constituents determine the chemical-quality characteristics of the water in lakes. The chemical constituents discussed in this report include major-chemical constituents, nutrients, trace elements, and uranium.

Lakes also contain a variety of biological organisms including bacteria, plankton, complex plants, insects, and fish. Plant and animal life are affected by changes in water quality. Measurements of the populations of phytoplankton (algae) and zooplankton and analyses of chlorophyll a and algal-growth potential can be used as indicators of biologic conditions of a lake.

Major Chemical Constituents

The major chemical constituents comprise most of the dissolved solids in water. The concentrations of major-chemical constituents usually are expressed in milligrams per liter. In freshwater, the principal cations are calcium, magnesium, sodium, and potassium. These cations are associated with the principal anions bicarbonate, carbonate, sulfate, and chloride. These cations and anions are all essential nutrients; but, unlike nitrogen and phosphorus, they usually occur in concentrations that are not limiting to plant growth.

During natural conditions, the concentrations of the major-chemical constituents primarily come from the minerals in rocks that are in the drainage basin upstream from the lake. However, waste materials from human activities may contribute substantial quantities of the constituents to water. Carbon dioxide, bicarbonate, and carbonate are the principal constituents that control the buffering of pH in lakes. However, during short periods and in the absence of other factors, the concentrations of the major-chemical constituents change minimally.

Nonuniform vertical mixing of major-chemical constituents can occur because of differences in wind velocity and water temperature, the shape of the lake and its basin, biological activity, and many other factors. Therefore, water samples for major-chemical-constituent analyses that are

collected at several depths in a lake and at several areally separated sites will indicate more accurately the distribution of major-chemical constituents than will samples collected from a single point. However, the epilimnion and hypolimnion of a lake usually are well mixed, and one sample collected from each in the summer usually is sufficient for a general characterization of the concentrations of the major-chemical constituents during stratification.

Nutrients

A nutrient is any substance necessary for growth, repair of tissue, or energy needs of an organism (Fruh, 1967). Nitrogen and phosphorus (in several forms) are the major nutrients because they commonly have substantial concentrations in water (reported in milligrams per liter) and are most likely to be depleted by phytoplankton, thus limiting further growth. The concentrations of these nutrients are altered readily by biological activity, and even short-term storage of water samples before analysis may result in substantial changes in concentration and form of the nutrients (Fishman and Friedman, 1985). The focus on nitrogen and phosphorus originates from the fact that these two macronutrients (especially phosphorus) are controllable from the perspective of eutrophication-control measures. However, phytoplankton also need many micronutrients and trace elements, generally reported in micrograms per liter.

For many years, scientists have categorized lakes in terms of nutrient concentrations (Lee and Fruh, 1966; Vollenweider, 1968, 1976; Rast and Lee, 1978). A lake that has water containing relatively small nutrient concentrations throughout an annual cycle is oligotrophic. A lake that has water containing relatively large nutrient concentrations is eutrophic. Mesotrophic lakes are those that contain moderate concentrations of nutrients. As a result of increased nutrient concentrations, the productivity and the biomass (plants and animals) of a lake increase. The process of lake eutrophication commonly is subdivided into natural and cultural (artificial) eutrophication. Nutrients naturally enter a lake from the atmosphere (principally through precipitation and dry fallout), ground-water inflow, streamflow, and surface runoff. Human activities, such as the application of agricultural fertilizers and discharge of treated municipal-sewage effluent in the drainage basin, can result in an increased nutrient load to a lake; this, in turn, may accelerate the eutrophication process.

Nutrient recycling in a lake, particularly the release of phosphorus from bottom sediments, also substantially increases the productivity and eutrophication of a lake. Chemical reactions in bottom sediments may release phosphorus to the overlying water. This phosphorus usually remains in solution in the oxygen-depleted hypolimnion and is mixed throughout the lake during periods of circulation as described in the "Trace Elements" section.

The productivity in a lake will be limited by the nutrient or other element present in the smallest concentration compared to plant and animal needs. Although nitrogen and phosphorus are key limiting nutrients (Lee and others, 1978), other elements and compounds can limit algal growth. For example, nitrogen and phosphorus may be the primary control for total-algal biomass in most lakes, but trace elements and even vitamins and organic

compounds may be the control for the types of algae present (Patrick, 1978). Also, silica is needed by diatoms (a group of algae) because their cell walls are composed of this constituent. In water containing relatively small silica concentrations, diatom growth can be limited (Wetzel, 1983). A literature review by Greeson (1971) indicated that there are at least 21 elements, in some form of chemical combination, essential for the growth of phytoplankton.

Trace Elements

Trace elements occur in water in relatively small concentrations, generally less than 1 mg/L. Trace-element concentrations usually are expressed in micrograms per liter, which is one thousandth of 1 mg/L. Many trace elements are essential for plant and animal growth. However, some trace elements may be toxic to plants and animals even in relatively small concentrations. For example, a concentration of total iron greater than 1,000 µg/L exceeds the recommended criterion for aquatic life (U.S. Environmental Protection Agency, 1986b).

Trace elements commonly are transported with sediment and are removed rapidly from the water when the sediment settles to the lake bottom. Reducing conditions exist in the bottom sediments at depth where the water filling the void spaces between the sediment particles contains little or no dissolved oxygen. These conditions commonly dissolve and mobilize the trace elements, which then move upward toward the surface of the bottom sediments. As the trace elements approach or reach the surface of the bottom sediments, they are oxidized by the dissolved oxygen contained in the lake water and precipitate on the upper sediment particles. These processes eventually may cause depletion of dissolved oxygen in the hypolimnion. The resulting reducing conditions again can dissolve and mobilize the trace elements. In lakes having reducing conditions in the hypolimnion, trace elements may dissolve readily, causing large concentrations in the hypolimnion compared to the epilimnion where most algal growth occurs. During periods of rapid mixing of the hypolimnion with the epilimnion, such as during the spring and autumn, trace elements from the hypolimnion are brought near the water surface. At these times, the trace elements may have toxic effects on the biota (Delfino, 1979).

Plankton

The organisms that inhabit the water of lakes and that are suspended and drift with the currents are called plankton. The phytoplankton, or plant part of the plankton, commonly are known as algae, and can reproduce prolifically in nutrient-rich aquatic environments. Unattached, visible, and sometimes extensive accumulations of algae, at or near the surface of the water, are called algal blooms.

Excessive densities of algae can cause tastes and odors, clog intake screens in water-treatment plants, produce slime conditions, and be toxic to animals (Palmer, 1977). Because of large production, great numbers of algae die as nutrients needed to maintain growth and reproduction are depleted. As a result of algal decomposition after death, dissolved oxygen may become depleted, causing fish kills.

The most common groups of planktonic algae are the diatoms, green algae, and blue-green algae. Usually, the blue-green algae become overabundant during the warmer months and cause problems in lakes. A diversity index commonly is determined for biological populations to assess the structure of the community. The diversity-index values range from 0 to some positive number; maximum diversity occurs when each individual in the sample belongs to a different species, and minimum diversity occurs when all individuals belong to the same species. Generally, in areas of organic loading or where toxic substances are present, the biotic community has minimum diversity; in areas where organic loading or toxic substances are not present, the biotic community has maximum diversity.

The primary photosynthetic pigment of all oxygen-producing photosynthetic organisms, including all algae, is chlorophyll a. The measurement of this pigment can indicate the quantity of algae present, although size of algal cells, quantity of pigment per cell, and physiological condition of the algae need to be considered.

The knowledge of algal-growth potential is important because algal growth primarily is affected by the availability of growth substances. By measuring algal-growth potential of lake-water samples, differentiation can be made between the growth substances that are determined by chemical analysis and the growth substances that actually are available for algal growth. The addition of growth substances to the sample can indicate which growth substances are limiting for algal growth. For this study, which used the procedure described by the U.S. Environmental Protection Agency (1971), *Selenastrum capricornutum* was the test organism.

Algal-growth-potential data that are derived in the laboratory in controlled conditions of light and temperature do not necessarily indicate conditions in the natural aquatic environment from which the sample was collected. Algal-growth potential at a specific time can be measured. In nature, daily and seasonal variations in the intensity and duration of light occur, which can affect algal growth. Growth can be inhibited by any suspended material, living or nonliving, that interferes with the effective penetration of light essential to the algae. Grazing by invertebrates and fish, toxic materials entering the water, plant debris or animal wastes, or decay products from the algae are all additional examples of factors that may inhibit algal-growth potential.

The animal part of the plankton are the zooplankton. Zooplankton are secondary producers that feed on bacteria and phytoplankton and are, in turn, consumed by fish or invertebrate predators. Because they are the grazers in the aquatic environment, zooplankton are a vital part of the aquatic food web.

RESULTS OF WATER-QUALITY ANALYSES OF ARVADA RESERVOIR

The physical, chemical, and biological water-quality data collected at Arvada Reservoir, Ralston Creek, and Croke Canal are in a report by Britton and Gaggiani (1986). A discussion of these data, an evaluation of the effects of source waters on the reservoir water quality, estimates of trophic state of

the reservoir, and identification of future data-collection needs are included in this section. Generally, the water quality of Arvada Reservoir is discussed in terms of site A, where most data were collected. However, spatial variations in the reservoir are identified and discussed.

Reservoir Profile and Stream Measurements

A total of 39 profile measurements were made at site A (fig. 2), 7 at site B, 19 at site C, 12 at site D, and 18 at site E in Arvada Reservoir. Each of these profile measurements will not be discussed individually but will be evaluated generally in terms of temporal and spatial trends.

Arvada Reservoir was thermally stratified during the warmer sampling periods, and the maximum metalimnion extended from depths of about 3 to 10 m at site A (fig. 4*C*). The earliest time that thermal stratification can be documented is for the April 17, 1985, measurement (fig. 4*J*) when the ice cover melted and the surface water began to warm. The latest fall measurement was made on October 5, 1983 (fig. 4*E*), and the reservoir was still thermally stratified although the reservoir was isothermal (equal temperature) at about 15 °C from the water surface to a depth of 12 m. During the winter, the reservoir is isothermal at about 4 °C, as indicated by measurements on February 2, 1984 (fig. 4*F*), which were made through a hole drilled through the ice at site A. Water temperature at site A ranged from 2.7 °C at the 5-m depth on March 1, 1984 (not in fig. 4), to 26.4 °C at the water surface on August 9, 1983 (fig. 4*C*).

Among the other sites in Arvada Reservoir, water temperature ranged from 2.9 °C throughout the depth profile at site C on March 1, 1984, to 27.0 °C at the water surface at the same site on August 9, 1983. Of course, time of sampling determined the ranges of water temperature among the sites for a particular sampling date. The reservoir was thermally stratified during the warmer sampling periods (July through August) at all sampling sites.

Slight supersaturation of dissolved oxygen and a larger pH value near the surface at site A indicated photosynthetic activity on many sampling dates during July through August. By early July (fig. 4*A*), the dissolved oxygen had decreased in the metalimnion, and by late July (fig. 4*B*), the reservoir at site A was nearly devoid of dissolved oxygen below the 5-m depth. Because the reservoir was thermally stratified, a density barrier formed preventing transfer of dissolved oxygen from the surface to the bottom. In addition, the anaerobic condition of the reservoir probably was enhanced by respiration and decomposition of algal material that was produced during the spring and summer. By September and October (fig. 4*E*), the reservoir was becoming isothermal, and the dissolved-oxygen concentration was almost zero only below the 10-m depth. The autumn-circulation period occurs during these months, as indicated by mixing of the water masses.

The dissolved-oxygen profile that was measured through the ice on February 2, 1984 (fig. 4*F*), indicates the effect of the ice cover. According to Hutchinson (1957), when the summer dissolved-oxygen curve is markedly clinograde (dissolved-oxygen concentration decreases to zero), as in Arvada Reservoir, the reservoir will have a similar distribution during the winter.

Photosynthetic production of organic matter, although less than in other seasons, continues throughout the winter and commonly is vigorous at the later stages of winter ice cover. Light penetration is variable, but the photic zone generally is confined to the epilimnion. As during summer stratification, respiration and chemical oxidation increase with depth, although at a slower rate because of lower temperatures. Because water in the littoral areas of Arvada Reservoir can be heated slightly through the ice by solar radiation, this warmer, slightly denser water will sink and move in profile-bound density currents along the sediments to the deeper parts of the reservoir. Generally, this movement is slow enough so the dissolved-oxygen concentration of this water is decreased or depleted as it passes over the sediment-water interface where some decomposition of organic material is still occurring. The result is a dissolved-oxygen profile as shown in figure 4*F*.

On a few sampling dates during late spring, a dissolved-oxygen maximum in the metalimnion causes a slightly positive heterograde curve (fig. 4*G*). This slightly positive curve probably occurred as a result of dissolved oxygen produced by algae that grow more rapidly than they sink.

Dissolved-oxygen concentrations at site A ranged from 0.0 mg/L near the reservoir bottom on several sampling dates to 12.0 mg/L at the 1-m depth on February 2, 1984 (fig. 4*F*). The earliest date that the dissolved-oxygen concentration was depleted was August 1, 1984 (fig. 4*H*), and the shallowest depth at which it was depleted was 8 m, also on this date.

Ranges and means of dissolved-oxygen concentrations averaged throughout the measured depths for each sampling date at site A are shown in figure 5. The mean dissolved-oxygen concentration averages between 8 to 10 mg/L during the winter and spring, but the mean dissolved-oxygen concentrations are markedly decreased during the summer and early autumn. As expected, the dissolved-oxygen concentrations were always smaller at the bottom than at the surface of the reservoir, and concentrations decreased rapidly to zero throughout late spring and early summer. During March and April, the reservoir apparently is nearly homothermous and well mixed, as indicated by the small variation in dissolved-oxygen concentrations. Throughout the sampling period, some of the variability in dissolved-oxygen concentrations probably is due to wind action and photosynthetic activity.

The decreased dissolved-oxygen concentration at lower depths in the reservoir may restrict vertical distribution of fish. A minimum allowable dissolved-oxygen concentration of 6.0 mg/L has been established for Arvada Reservoir to fulfill spawning requirements (Colorado Department of Health, Water Quality Control Commission, 1981).

Among the other sampling sites in Arvada Reservoir, the dissolved-oxygen concentration became depleted at sites C and E (fig. 2) on several sampling dates beginning in August and continuing through September. Probably because of their shallow depths, water at sites B and D did not become completely anaerobic, although the dissolved-oxygen concentration was decreased to only 0.1 mg/L at site B in late July 1983. The dissolved-oxygen concentration among all the other sampling sites (sites B-E) ranged from 0.0 mg/L at sites C and E on several sampling dates to 10.7 mg/L near the water surface at site C on March 1, 1984.

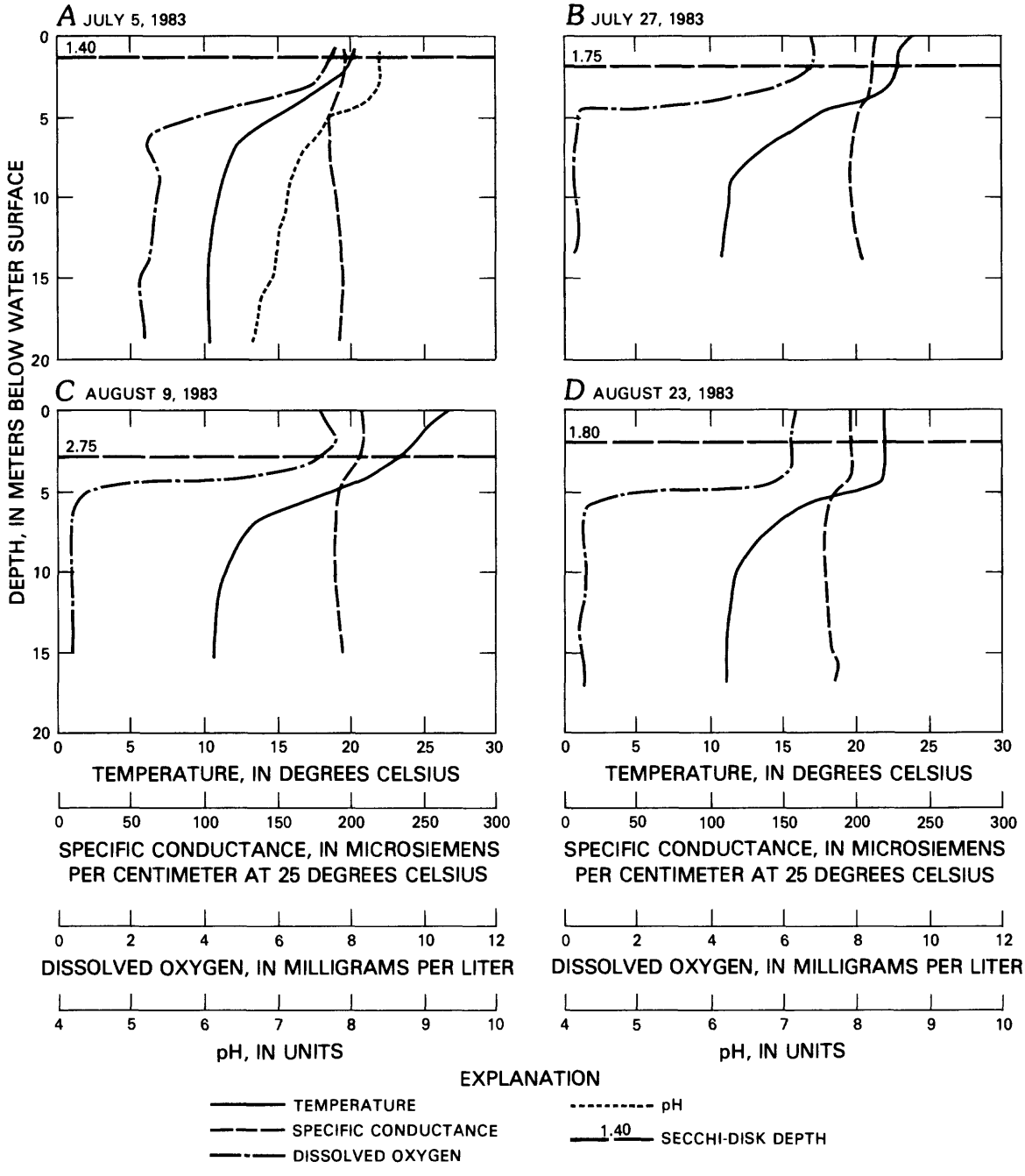

Figure 4.--Selected profile measurements for site A, 1983-85.

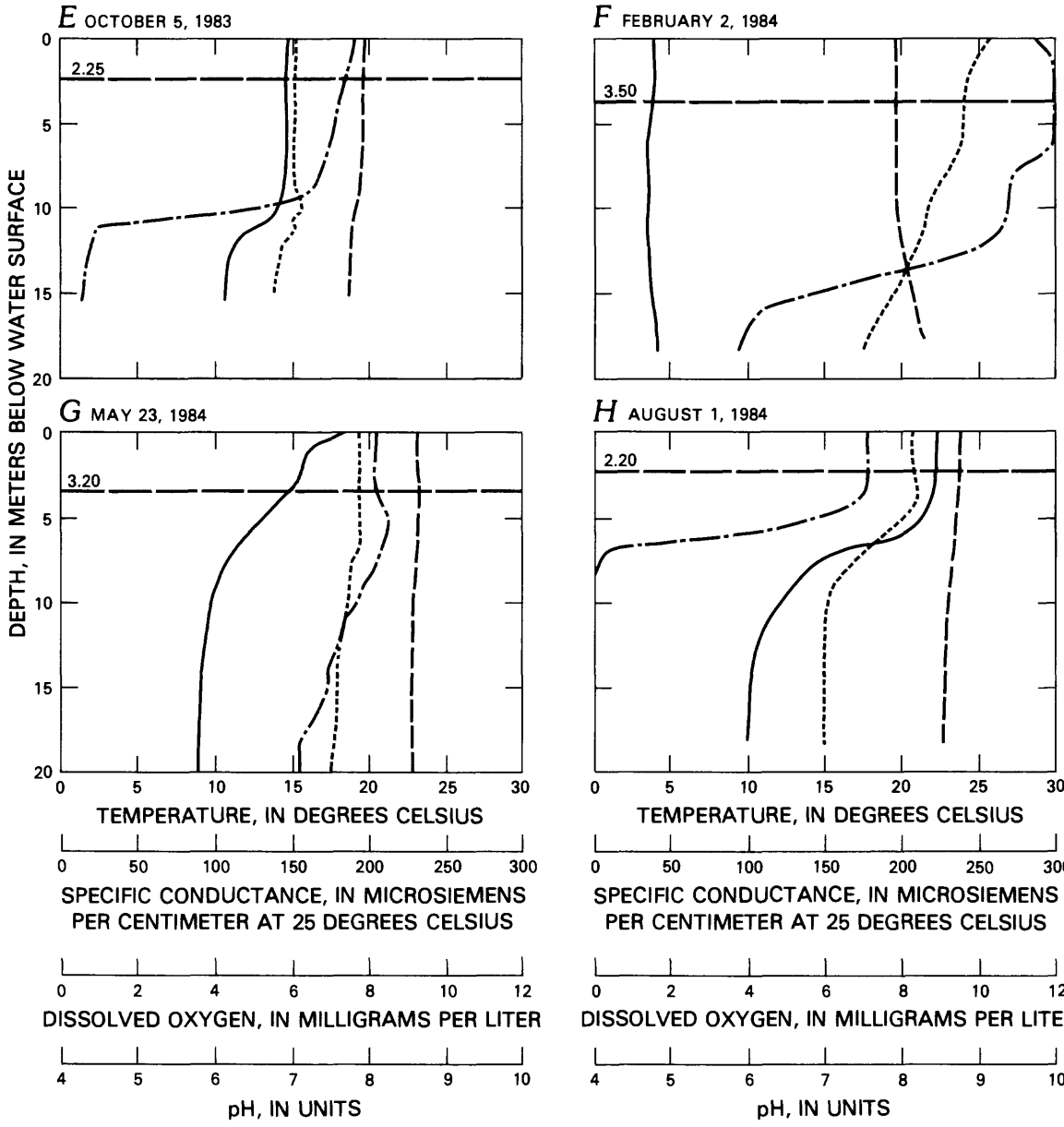

Figure 4.--Selected profile measurements for site A, 1983-85--Continued.

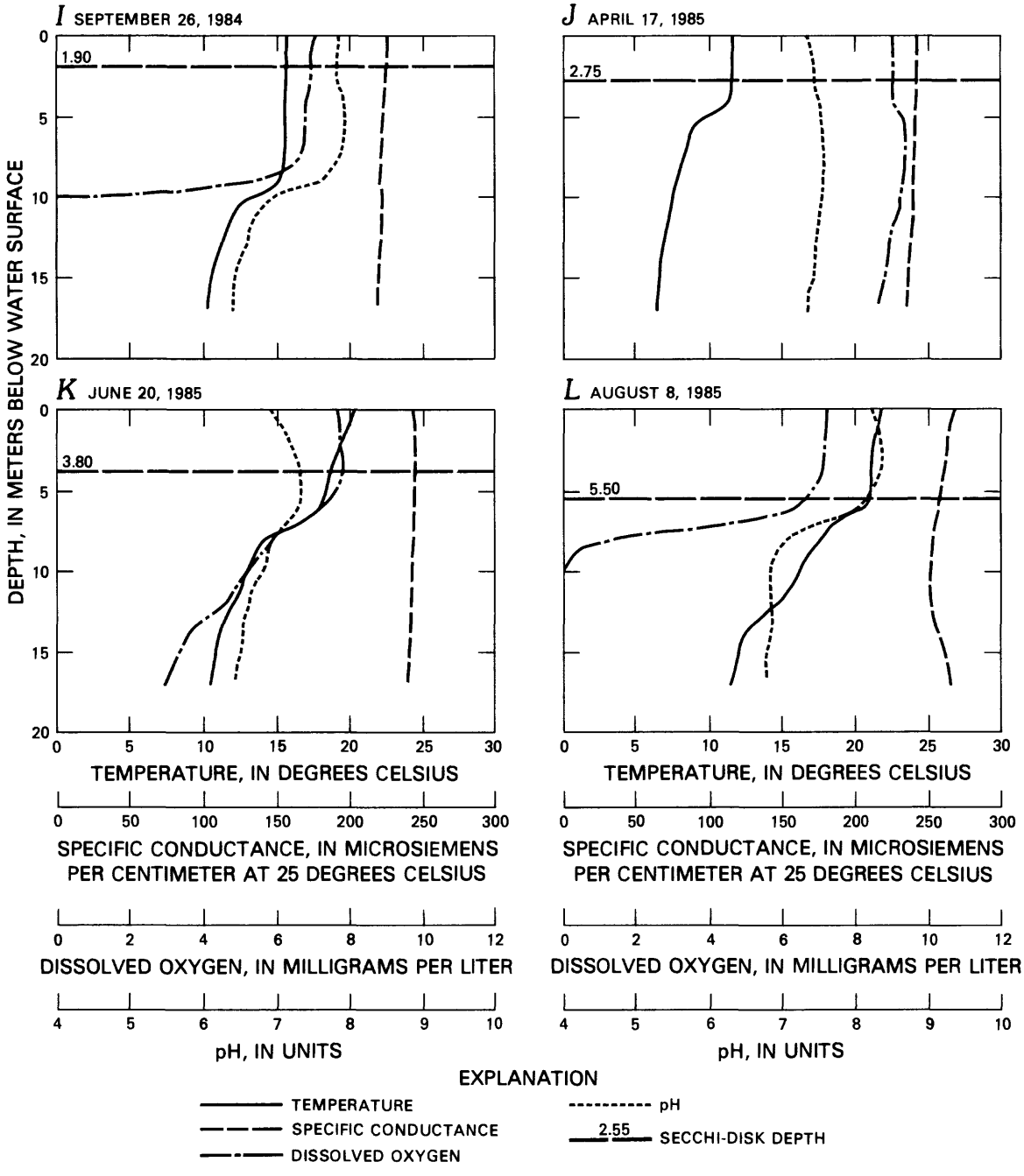

Figure 4.--Selected profile measurements for site A, 1983-85--Continued.

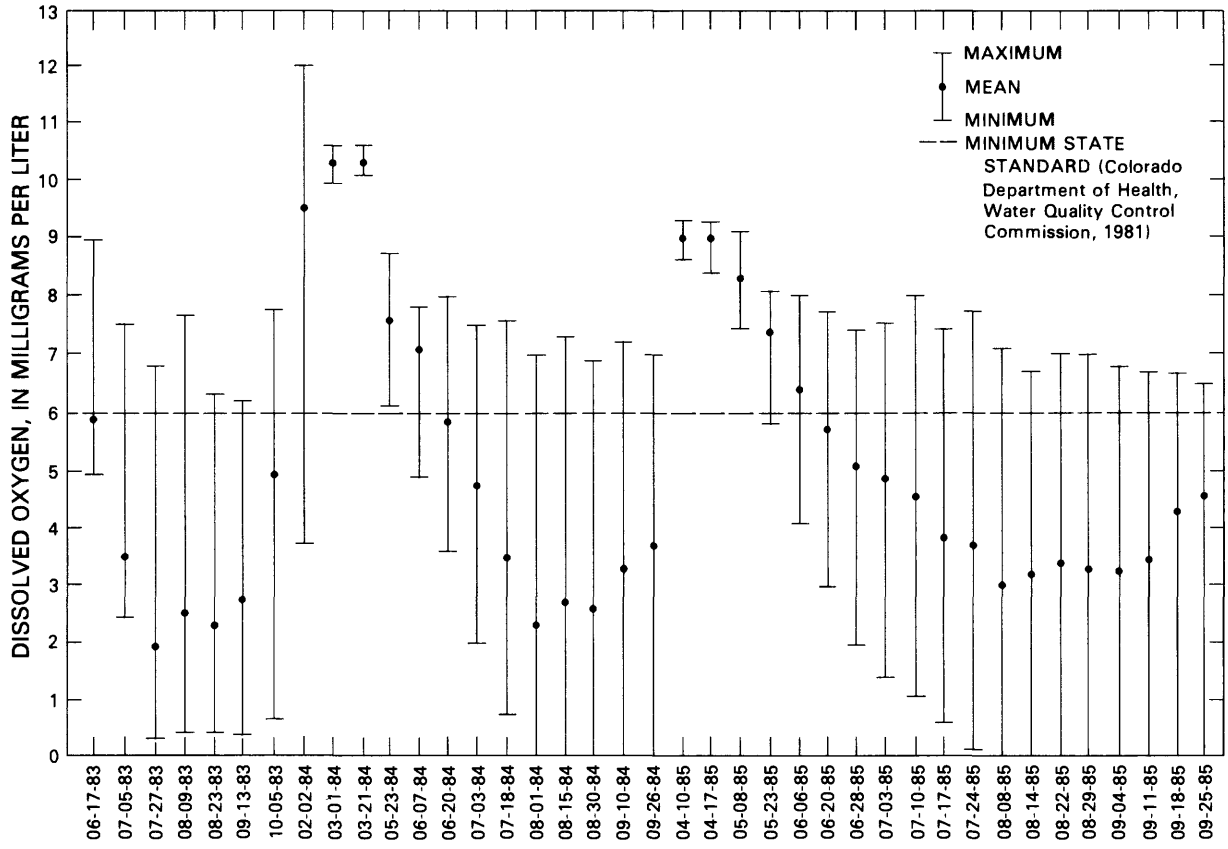

Figure 5.--Ranges and means of dissolved-oxygen concentrations for all sampling dates at site A.

Generally, specific conductance decreased or was unchanged from the surface to the bottom of the reservoir at site A. Because specific conductance of water is controlled by the concentrations of major ions, which are relatively conservative, only minor temporal and spatial fluctuations in the reservoir occur as a result of biotic use of the major ions or as a result of biotically caused environmental changes. The specific conductance at site A ranged from 177 µS/cm at a middepth on August 23, 1983 (fig. 4*D*), to 264 µS/cm at the water surface on August 8, 1985 (fig. 4*L*). An exception to the usual profile of specific conductance is shown for February 2, 1984, in figure 4*F*. On this date, specific conductance was larger near the bottom than at the water surface. As shown in figure 6, the near-surface measurements of specific conductance almost always are larger than the near-bottom measurements. There also is a trend of increasing specific conductance throughout the study, as shown for mean specific-conductance measurements (averaged for each sampling date throughout the sampling depths) in figure 7. Presently (1986) it is unknown why there is a trend of increasing specific conductance, but it may be due to a combination of the following: (1) Increased specific conductance from the variety of inflows to the reservoir, (2) increased dissolution of sediment particles after filling of the reservoir, and (3) concentration by direct evaporation.

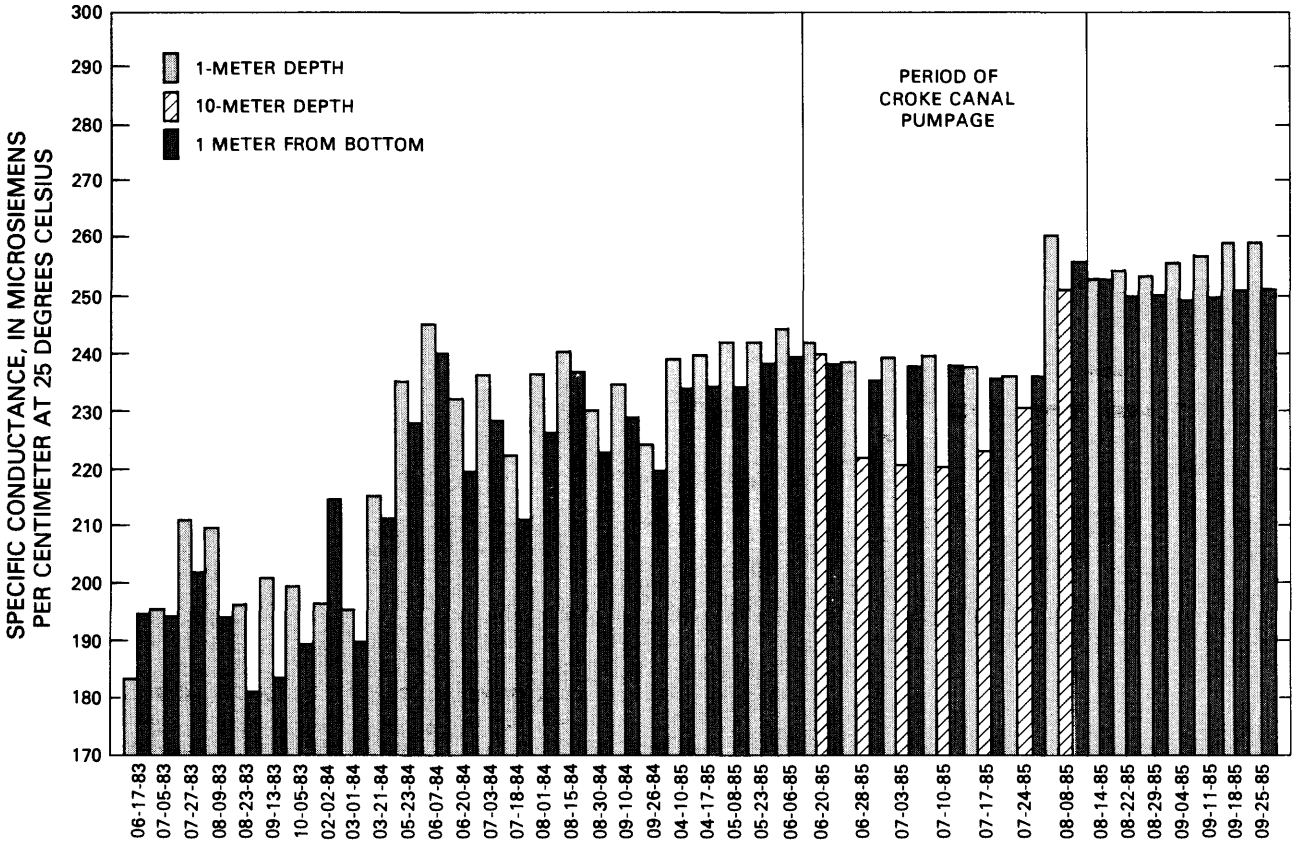

Figure 6.--Near-surface and near-bottom specific-conductance measurements for all sampling dates at site A.

The Croke Canal pipeline that pumps water into the reservoir, at the point shown in figure 2, affects the common specific-conductance curve as shown in figure 6. The result is a decrease in specific conductance beginning about the 9-m depth and extending to about the 13-m depth. Because specific conductance in Croke Canal ranged from 69 to 92 percent of the specific conductance near the water surface in the reservoir, a definite stratum of Croke Canal water is present throughout the metalimnion and upper hypolimnion The overall effect on mean specific-conductance measurements in the reservoir at site A also is shown in figure 7; mean specific conductance decreased between June 20 and July 24, 1985. A similar decrease in specific conductance occurred at site E (fig. 2) from about the 7-m depth to the bottom. Among the other sampling sites (sites B-E), specific conductance ranged from 174 µS/cm near the bottom at site B on October 5, 1983, to 266 µS/cm near the bottom at site E on September 18, 1985.

From the water surface to the bottom of the reservoir, pH generally decreased indicating carbon-dioxide uptake during photosynthesis near the water surface and carbon-dioxide release because of respiration near the bottom. The pH profile during late summer (fig. 4*L*), when the reservoir is thermally stratified, is typical because of respiratory generation of carbon dioxide throughout the trophogenic zone (stratum where photosynthetic

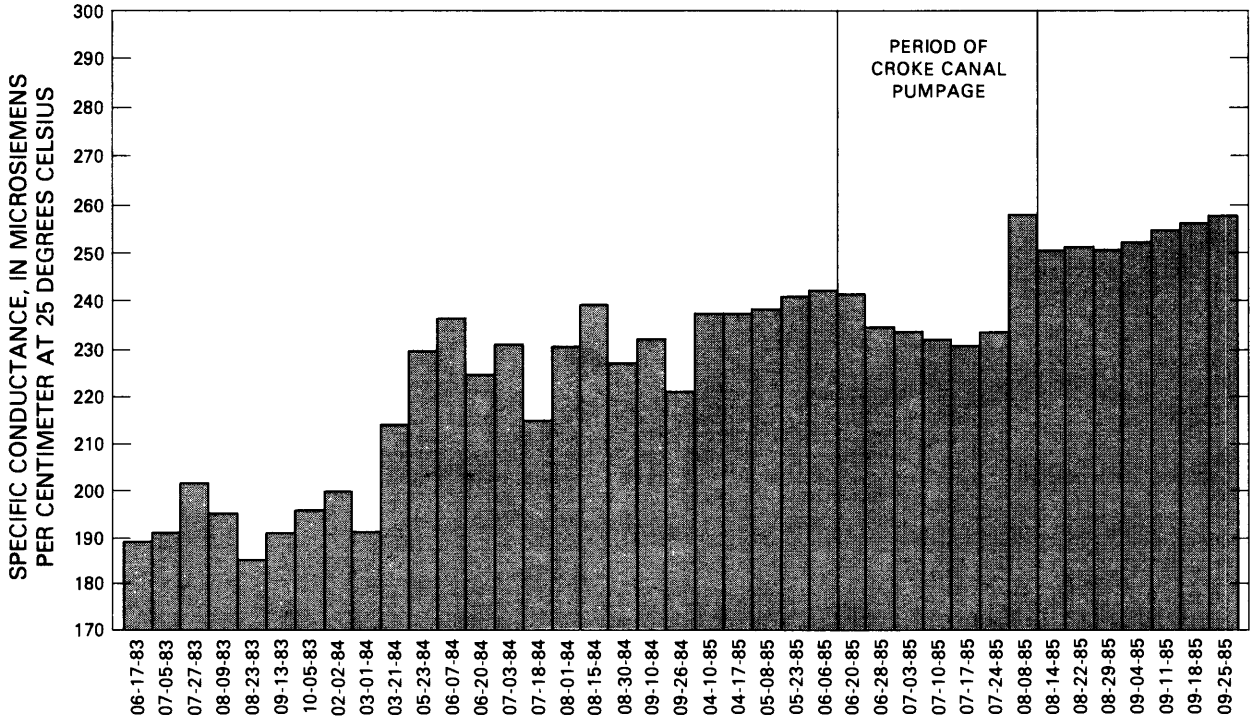

Figure 7.--Mean specific-conductance measurements averaged for all depths for all sampling dates at site A.

production predominates), which tends to decrease pH. The pH is characterized by a slightly stratified profile during the summer (see fig. 4*A*, for example); whereas, during the spring (fig. 4*J*) it is relatively unchanged from the water surface to the bottom. At site A, pH ranged from 6.4 near the bottom on September 26, 1984 (fig. 4*I*), and June 20, 1985 (fig. 4*K*), to 9.2 near the water surface on February 2, 1984 (fig. 4*F*). An allowable range of pH (6.5-9.0) has been determined for Arvada Reservoir and its drainage basin for maintenance of biological quality (Colorado Department of Health, Water Quality Control Commission, 1981).

Among the other sampling sites (sites B-E), pH values were similar to those at site A. The values ranged from 6.5 near the bottom at site E on July 24, 1985, to 8.7 at the 2-m depth at site B on June 17, 1983.

The Secchi-disk-depth measurements at site A ranged from 1.0 m on June 17, 1983, to 5.5 m on August 8, 1985. The mean for site A was 2.9 m, and the median was 2.7. The smaller Secchi-disk-depth measurements at the beginning of the study probably occurred because the reservoir was still filling, and suspension of particulate matter and an immediate increase in algal biomass due to nutrients entering the system from the newly inundated land surface contributed to turbidity in the reservoir. The Secchi-disk-depth measurements for all the sampling dates at site A are shown in figure 8. In addition, there seems to be a trend of increasing transparency throughout the

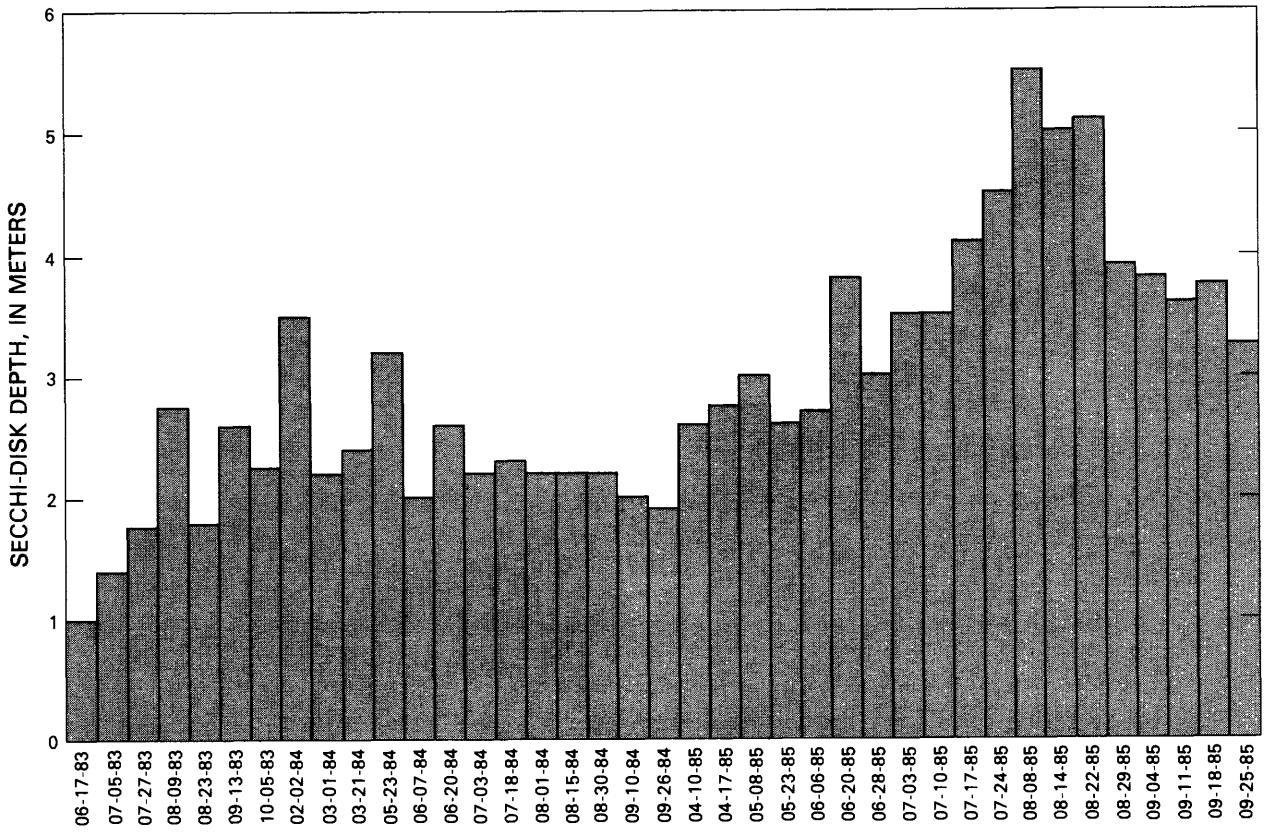

Figure 8.--Secchi-disk-depth measurements for all sampling dates at site A.

study period. Light transparency, as measured using the Secchi disk, occasionally indicated the effect of phytoplankton biomass. In some instances, light transparency improved as phytoplankton biomass decreased (indicated by chlorophyll a concentrations), probably as a result of zooplankton grazing (fig. 9). However, the results also indicate that there probably is some other factor, such as settling of particulate matter, that affects light transparency.

Among the other sites in the reservoir, Secchi-disk-depth measurements ranged from 0.9 m at site C on June 17, 1983, to 5.5 m at site E on August 8, 1985. The mean for all the sampling sites was 2.7 m. According to Wetzel (1983), the reservoir would be classified between mesotrophic and eutrophic based on the mean Secchi-disk-depth measurements. The mean of 2.7 m was within the range of Secchi-disk-depth measurements (0.8-8.1 m) used to define the different trophic states.

The dissolved-solids concentrations (as indicated by specific-conductance measurements) in Ralston Creek were less during higher flows because of dilution and possibly as a result of inflows from Upper Long Lake. Specific-conductance measurements ranged from 64 µS/cm (flow of about 2.8 ft^3/s) on

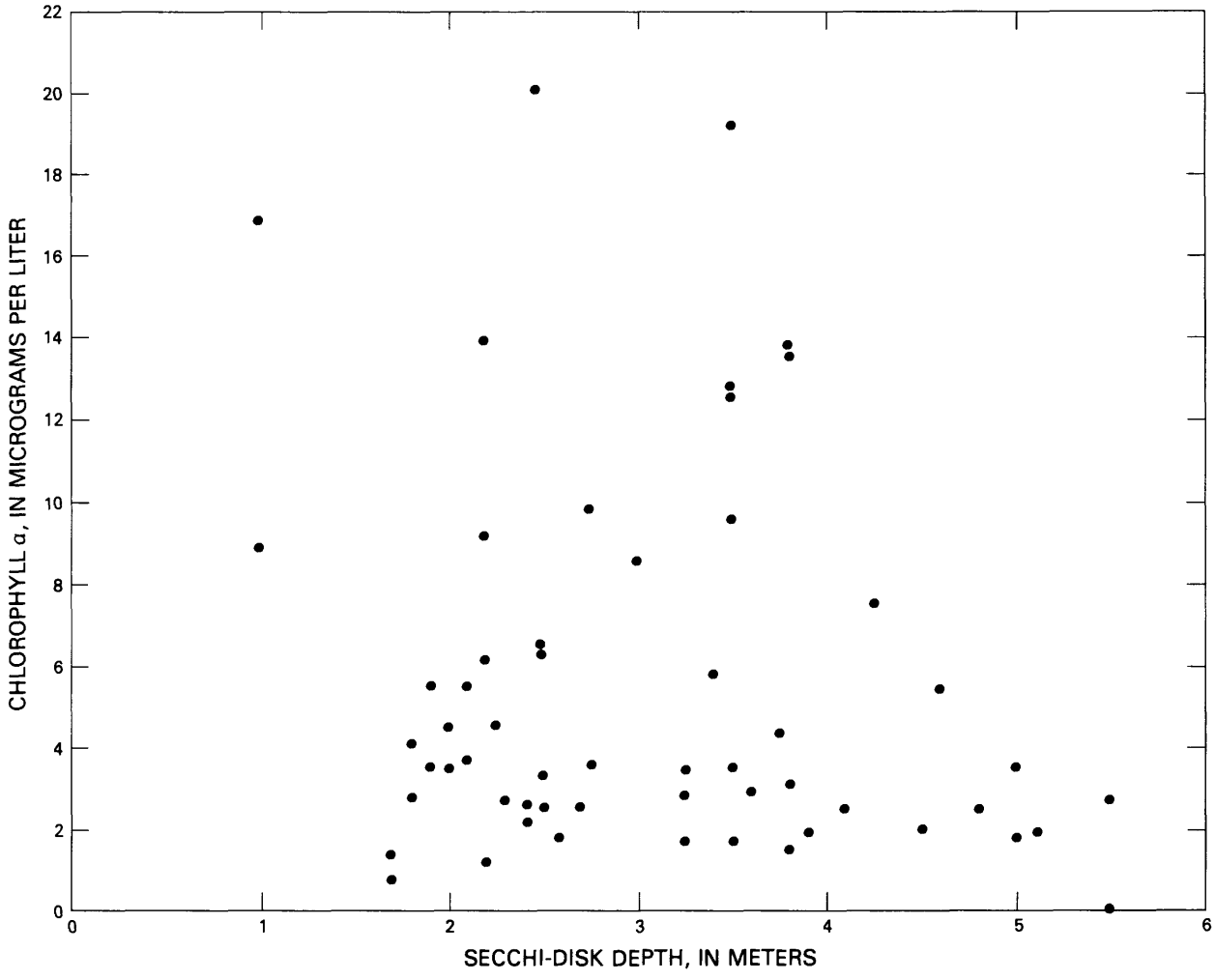

Figure 9.--Relation between Secchi-disk depth and chlorophyll *a* concentrations for all sampling dates at sites A, B, C, D, and E.

July 10, 1985, to 531 µS/cm (flow of <1 ft^3/s) on June 17, 1983. The quality of water in Ralston Creek was suitable for most uses, and the dissolved-oxygen concentrations were saturated on most sampling dates. Dissolved-oxygen concentrations ranged from 6.9 mg/L on October 5, 1983, to 9.6 mg/L on March 21, 1984. The dissolved-oxygen concentrations are dependent on water temperature, and this minimum dissolved-oxygen concentration of 6.9 mg/L occurred at the second highest water temperature (21.5 °C) recorded for Ralston Creek during the study. This minimum value is slightly less than the stream standard of at least 7.0 mg/L specified as a spawning requirement for Ralston Creek (Colorado Department of Health, Water Quality Control Commission, 1981). The maximum dissolved-oxygen concentration of 9.6 mg/L occurred at the second lowest water temperature (10.5 °C) recorded for Ralston Creek during the study. The pH measurements ranged from 6.7 on June 4, 1985, to 8.1 on several sampling dates. These measurements were within the ranges of those recorded at sampling sites in the reservoir. The pH values also were within the range of 6.5 to 9.0 specified as a stream standard for maintenance of

aquatic life and for recreational, water-supply, and agricultural uses designated for Ralston Creek by the Colorado Department of Health, Water Quality Control Commission (1981).

As in Ralston Creek, the dissolved-solids concentrations (as indicated by specific-conductance measurements) in Croke Canal (CC-1 and CC-2) were less during higher flows because of dilution. Specific conductance ranged from 89 µS/cm (flow of 369 ft^3/s) on July 5, 1983, to 313 µS/cm (flow of 24 ft^3/s) on July 27, 1983. In most instances, the specific-conductance measurements were less than those at the sampling sites in the reservoir for the same sampling date, as indicated by the plot of specific conductance (fig. 6) at the 10-m depth during pumping of Croke Canal water into the reservoir. The dissolved-oxygen concentration ranged from 6.4 mg/L (water temperature of 25.5 °C) on July 27, 1983, to 9.9 mg/L (water temperature of 25.5 °C) on July 10, 1985. The larger dissolved-oxygen concentration at the high temperature of 25.5 °C would seem to be suspect or could be due to extensive production of algae at this site. The dissolved-oxygen concentrations always were larger than the minimum of 5.0 mg/L specified as a stream standard for maintenance of aquatic life and for recreational and agricultural uses designated for Croke Canal by the Colorado Department of Health, Water Quality Control Commission (1981). The pH values ranged from 6.5 on June 28, 1985, to 8.0 on July 10, 1985, which were within the ranges of those measured in the reservoir. The pH values also were within the range of 6.5 to 9.0 specified as the stream standard for Croke Canal (Colorado Department of Health, Water Quality Control Commission, 1981).

Chemical Quality

The relation between specific conductance and dissolved-solids concentrations for sites A, B, C, and D in Arvada Reservoir is shown in figure 10. Specific-conductance measurements and dissolved-solids concentrations at these sites were within a fairly narrow range, and the data points are greatly scattered. The narrow range and scatter make it difficult to develop a statistical relation between these two variables. However, continued sampling could add additional points to this graph and improve the data set.

Water-quality data collected from the five sampling sites in Arvada Reservoir indicate that dissolved nitrate concentrations are less than the primary drinking-water standard (10 mg/L as nitrogen) specified by the U.S. Environmental Protection Agency (1986a) for nitrate, but dissolved manganese concentrations occasionally exceeded the secondary drinking-water standard (50 µg/L) specified by the U.S. Environmental Protection Agency (1986c). In addition, total-mercury and total-zinc concentrations occasionally exceeded the aquatic-life criteria (U.S. Environmental Protection Agency, 1986b), and total-uranium concentrations occasionally exceeded standards specified for the reservoir (within its water-supply classification) by the Colorado Department of Health, Water Quality Control Commission (1981). The dissolved-manganese concentrations that exceeded the secondary drinking-water standard, the total-mercury and total-zinc concentrations that exceeded aquatic-life criteria, and the total-uranium concentrations that exceeded water-supply standards in water samples collected from site A are listed in table 2. The standards and criteria, in micrograms per liter, also are listed. In most

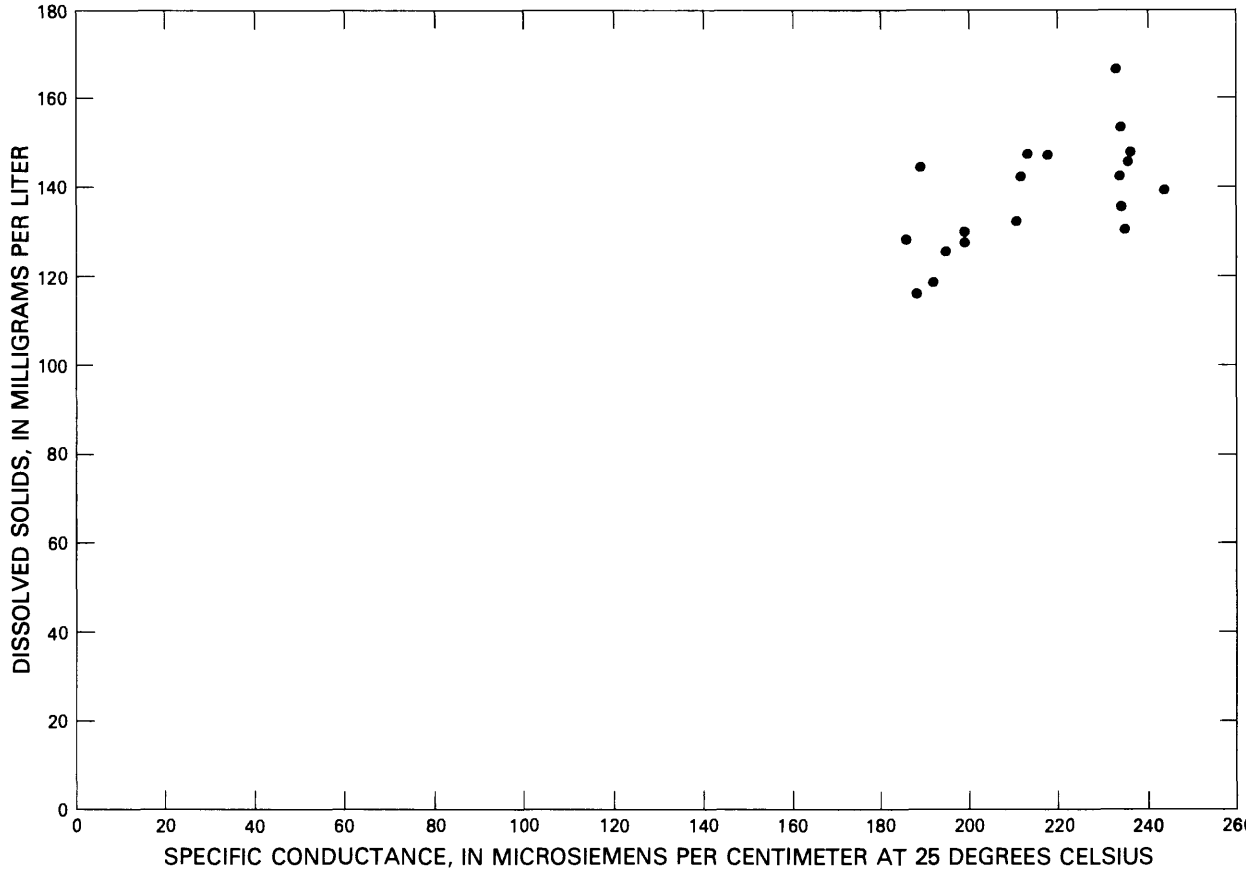

Figure 10.--Relation between specific conductance and dissolved-solids concentrations at sites A, B, C, and D.

instances, these standards or criteria coincide with those specified for the reservoir (within its water-supply classification) by the Colorado Department of Health, Water Quality Control Commission (1981).

In addition, dissolved-manganese concentrations exceeded secondary drinking-water standards in the hypolimnion at site C on August 9, 1983; total-mercury concentrations exceeded aquatic-life criteria at sites B and C on August 9, 1983, and at site D on August 1, 1984; and total-zinc concentrations exceeded aquatic-life criteria at site C on August 9, 1983.

The data also indicate that on several sampling dates there was a difference in concentrations of total nitrogen, total phosphorus, total iron, and total manganese between the epilimnion and the hypolimnion at site A, probably as a result of stratification. For example, on September 18, 1985, water samples collected at the 1- and 13-m depths had the following concentrations: total nitrogen at 1-m depth = <0.41 mg/L, at 13-m depth = <0.61 mg/L; total phosphorus at 1-m depth = 0.012 mg/L, at 13-m depth = 0.024 mg/L; total iron

Table 2.--*Selected water-quality constituents that exceeded standards or criteria at site A*

[All concentrations are in micrograms per liter; --, no analysis]

Date	Dissolved manganese	Total mercury	Total zinc	Total uranium
Standard or criteria	[1]50	[2]0.05	[2]150	[3]58
08-09-83	130	[4]0.1	--	--
08-01-84	[4]360	.1	--	--
03-15-85	--	.1	--	--
04-03-85	--	.3	--	--
07-24-85	--	7.5	110	89
08-22-85	--	[4].2	90	770
09-18-85	--	--	70	--

[1]Secondary drinking-water standard (U.S. Environmental Protection Agency, 1986c).

[2]Aquatic-life criteria (U.S. Environmental Protection Agency, 1986b).

[3]Water-supply standard (Colorado Department of Health, Water Quality Control Commission, 1981).

[4]Analyses from samples collected in the hypolimnion.

at 1-m depth = <10 µg/L, at 13-m depth = 390 µg/L; and total manganese at 1-m depth = 20 µg/L, at 13-m depth = 740 µg/L. Nutrient and trace-element concentration differences between the water surface and bottom of the reservoir occurred during the summers of 1983, 1984, and 1985. This is important when determining at what depth water will be withdrawn from the reservoir.

Concentrations of total nitrite plus total nitrate as nitrogen ranged from <0.01 to 0.16 mg/L and had a mean concentration of 0.027 mg/L at site A. Concentrations of total ammonia plus total-organic nitrogen as nitrogen ranged from 0.2 to 1.0 mg/L and had a mean concentration of 0.5 mg/L. Seasonal variations were apparent in the concentrations of total nitrite plus total nitrate and total ammonia plus total-organic nitrogen (except during early 1985). However, because the concentrations of total nitrite plus total nitrate as nitrogen generally were less than 0.05 mg/L, the effects of their seasonal change on the overall total-nitrogen concentration is slight (fig. 11). Total ammonia plus total-organic nitrogen is the major component of the total-nitrogen concentration of the reservoir as shown in figure 11. Concentrations of total nitrite plus total nitrate were largest during the summer of 1984; whereas, concentrations of total ammonia plus total-organic nitrogen were largest during the summers of 1983 and 1984 and had peak concentrations during the winter and spring of 1985.

Total-phosphorus concentrations ranged from 0.003 to 0.021 mg/L and had a mean of 0.009 mg/L and a median of 0.01 mg/L at site A. Seasonal variations in total-phosphorus concentrations are not apparent (fig. 12); however, there was more variation in total-phosphorus concentrations during 1985.

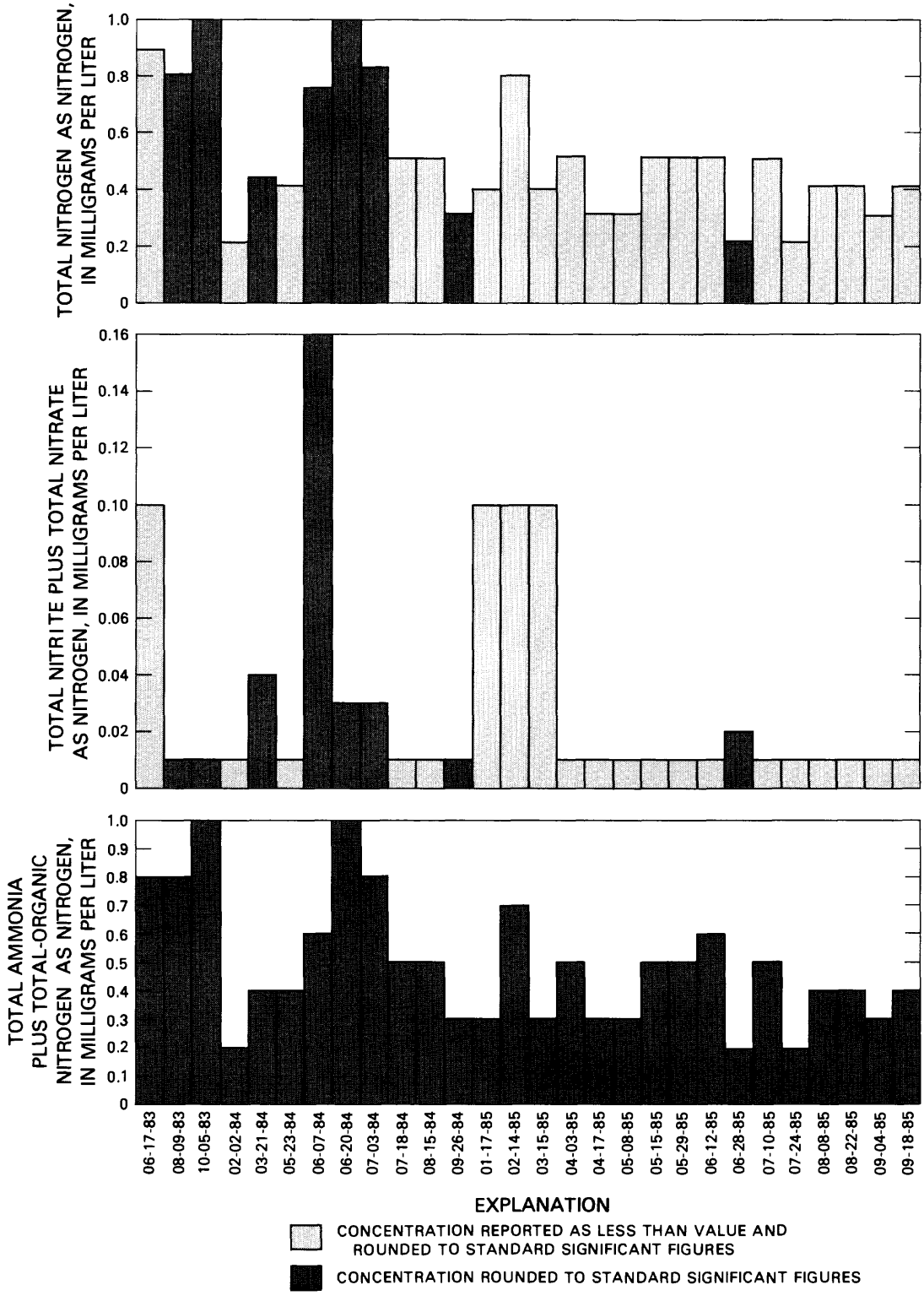

Figure 11.--Concentrations of total nitrogen, total nitrite plus total nitrate, and total ammonia plus total-organic nitrogen (all as nitrogen) for all sampling dates at site A.

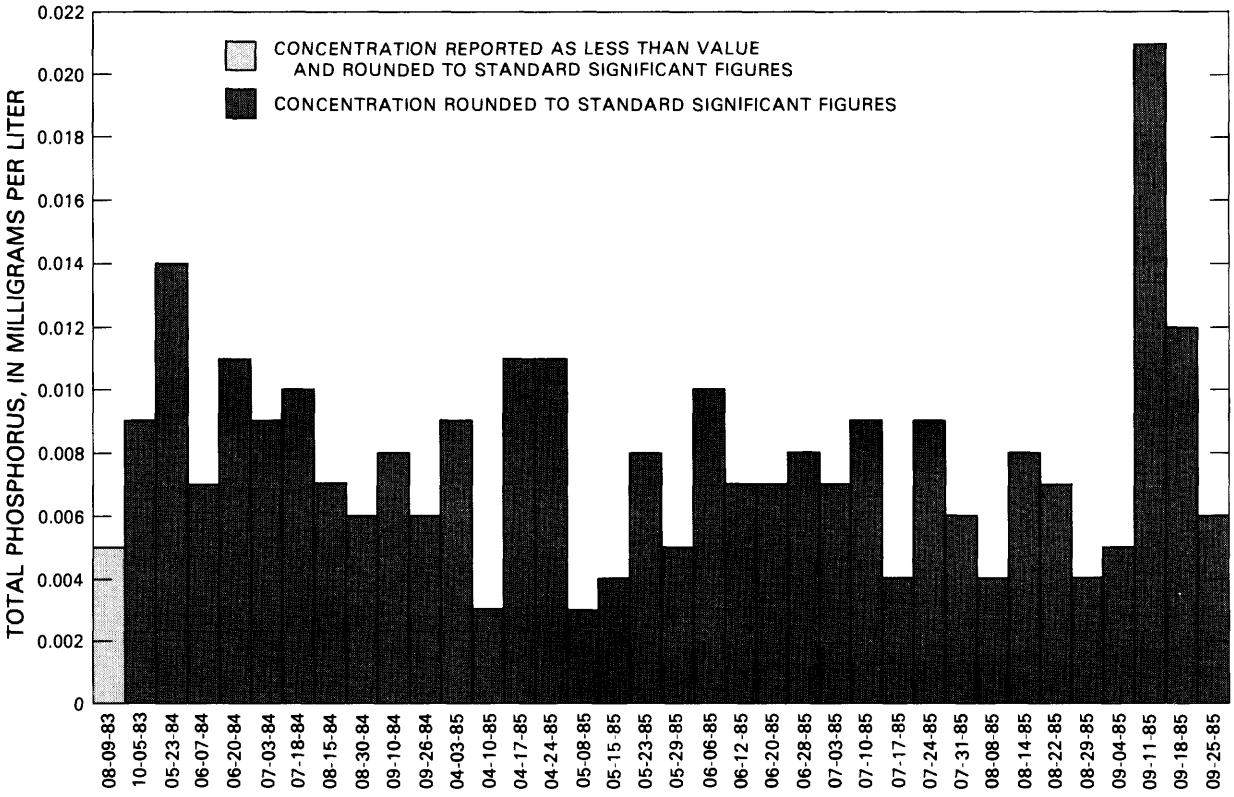

Figure 12.--Concentrations of total phosphorus for selected sampling dates at site A.

Mass ratios of total-nitrogen to total-phosphorus concentrations at site A ranged from less than 23:1 to greater than 162:1 (table 3). Some ratios are denoted as less than (<) or greater than (>) because the nutrient analyses commonly were denoted as such, even though low-detection levels were used. Lakes having a ratio of total-nitrogen to total-phosphorus concentration less than 26:1 are designated by Carlson (1977, 1981) as nitrogen limited. Results of laboratory bioassays (Chiaudani and Vighi, 1974, 1975) indicated that a ratio in weight larger than 10 is an index of phosphorus limitation; a ratio smaller than 5 indicates nitrogen limitation; a ratio between 5 and 10 indicates that the two nutrients are in an optimal assimilative proportion. In addition, blue-green algae tend to be rare when this ratio exceeds 29:1 (Smith, 1983). Based on the data in table 3 and the assumptions and limited data sets used by Chiaudani and Vighi (1974, 1975), Carlson (1977, 1981), and Smith (1983), Arvada Reservoir would be considered phosphorus limited on nearly all sampling dates, and blue-green algae would not be expected to dominate in the reservoir. However, blue-green algae occasionally are dominant in the reservoir (see "Phytoplankton" section), even though the total-nitrogen to total-phosphorus ratios in the reservoir usually are greater than 29:1. When using concentrations of total nitrogen and total

Table 3.--*Ratios of total-nitrogen to total-phosphorus concentrations at site A*

[TN, total nitrogen; TP, total phosphorus; <, less than; >, greater than]

Date	TN:TP ratio	Date	TN:TP ratio
06-17-83	<45:1	03-15-85	<80:1
08-09-83	>162:1	04-03-85	<57:1
10-05-83	112:1	04-17-85	<28:1
02-02-84	< or >42:1	05-08-85	<103:1
03-21-84	44:1	05-15-85	<128:1
05-23-84	<29:1	05-29-85	<102:1
06-07-84	108:1	06-12-85	<87:1
06-20-84	94:1	06-28-85	27:1
07-03-84	92:1	07-10-85	<57:1
07-18-84	<51:1	07-24-85	<23:1
08-15-84	<73:1	08-08-85	<103:1
09-26-84	52:1	08-22-85	<59:1
01-17-85	<50:1	09-04-85	<62:1
02-14-85	<100:1	09-18-85	<34:1

phosphorus to calculate ratios, the measured concentrations include the nutrients present in algal cells as well as those present in the water column. Another approach is to evaluate the ratio of the biologically available forms of nitrogen and phosphorus (sum of dissolved ammonia and dissolved nitrate as nitrogen and dissolved orthophosphate as phosphorus). The water-column concentrations of biologically available nitrogen and phosphorus measured during an algal bloom represent the unused nutrients in the water column (nutrients not needed by algae for growth) during a period when algal nutrient requirements should be maximum. In addition, analyses of nutrient limitation in lakes studied by Chiaudani and Vighi (1975) indicated that dissolved orthosphosphate as phosphorus was the major factor for phytoplankton growth (accounting for 72.8 percent of total variability). Comparison of the ratios of these nutrients against a reference value of 7.2N:1P (Redfield, 1934, 1958) can provide information on the potential limiting nutrient, provided the nutrient load to the waterbody does not change markedly from year to year. The ratios for Arvada Reservoir ranged from 1.3N:1P to 26N:1P (22 percent of ratios less than 7.2:1), indicating that nitrogen occasionally is limiting in the reservoir. However, these analyses were made prior to pumping of Croke Canal water into the reservoir. Finally, the absolute concentrations of the biologically available nutrients also can provide information about which nutrient (nitrogen or phosphorus) is limiting (W.R. Rast, U.S. Geological Survey, written commun., 1986). Accordingly, if the sum of the concentrations of dissolved ammonia and dissolved nitrate as nitrogen decreases to less than about 20 µg/L, or if the concentration of dissolved orthophosphate as phosphorus decreases to less than about 5-10 µg/L, then that nutrient(s) is likely to

be the limiting nutrient (W.R. Rast, U.S. Geological Survey, written commun., 1986). For Arvada Reservoir, the combined concentrations of dissolved ammonia and dissolved nitrate as nitrogen rarely were less than 20 µg/L (about 13 percent of the analyses), whereas the concentration of dissolved orthophosphate as phosphorus often was less than 5 µg/L (67 percent of the analyses). However, this information by itself is not conclusive because algal-growth-potential determinations (see "Algal-Growth Potential" section) and taxonomic composition of phytoplankton (see "Phytoplankton" section) provide additional information about some of the interactive relations that occur in the reservoir.

For the sampling dates (Britton and Gaggiani, 1986), the sediment at the bottom of the reservoir at site A contained 1,800 mg/kg of total nitrogen; 990 mg/kg of total phosphorus; 6,100 to 9,800 µg/g of total iron; and 180 to 200 µg/g of total manganese. When the reservoir is stratified and the hypolimnion is anaerobic, these nutrients and trace elements could become mobilized and available for biotic use when the reservoir becomes well mixed during the autumn-circulation period.

A compilation of all the water-quality data collected in Ralston Creek and Croke Canal are presented in a report by Britton and Gaggiani (1986). The water entering Arvada Reservoir from Ralston Creek and Croke Canal during the study period is similar in chemical type (sodium calcium sulfate) to the water in the reservoir. Selected chemical analyses from Ralston Creek and Croke Canal are listed in table 4. Concentrations of total nitrite plus total nitrate as nitrogen and dissolved solids were a little larger in both inflows compared to water in the reservoir.

The chemical water quality of Ralston Creek is more variable (table 4) than the chemical water quality at the sampling sites in Arvada Reservoir, because the water flowing through Ralston Creek into the reservoir comes from a variety of sources, such as the reaches of Ralston Creek upstream from Ralston Reservoir (fig. 1) and a series of canals, pipelines, tunnels, and reservoirs that transport and store water from the western slope of the Rocky Mountains. Concentrations of total nitrite plus total nitrate as nitrogen ranged from 0.025 to 1.20 mg/L and had a mean of 0.169 mg/L and a median of 0.056 mg/L. The large difference between the mean and the median was caused by one large concentration of total nitrite plus total nitrate in the water sample collected on June 17, 1983. Concentrations of total ammonia plus total-organic nitrogen as nitrogen ranged from <0.2 to 1.0 mg/L and had a mean of 0.6 mg/L and a median of 0.6 mg/L. Total-nitrogen concentrations ranged from 0.24 to 2.0 mg/L and had a mean of 0.74 mg/L and a median of 0.67 mg/L. Total-phosphorus concentrations ranged from <0.005 to 0.10 mg/L and had a mean of 0.020 mg/L and a median of 0.015 mg/L. Ratios of total-nitrogen to total-phosphorus concentrations for Ralston Creek are listed in table 5. The ratios are similar to those in the reservoir and range from 12:1 to greater than 316:1. In addition, limited data indicate ratios of biologically available nitrogen (dissolved ammonia plus dissolved nitrate as nitrogen) and dissolved orthophosphate as phosphorus in the reservoir of 30:1 and 140:1, concentrations of dissolved ammonia plus dissolved nitrate as nitrogen much larger than 20 µg/L, and concentrations of dissolved orthophosphate as phosphorus less than 5-10 µg/L. Therefore, in most instances, algal biomass in the creek is

Table 4.--*Selected chemical analyses of water samples collected from Ralston Creek and Croke Canal*

[ft^3/s, cubic feet per second; µS/cm, microsiemens per centimeter at 25 degrees Celsius; °C, degrees Celsius; mg/L, milligrams per liter; µg/L, micrograms per liter; <, less than; --, no data; E, estimated]

Date	Streamflow, instantaneous (ft^3/s)	Specific conductance (µS/cm)	Solids, residue at 105 °C, total (mg/L)	Solids, residue at 180 °C, dissolved (mg/L)	Total nitrogen (mg/L)	Total phosphorus (mg/L)	Dissolved iron (µg/L)	Total iron (µg/L)	Dissolved manganese (µg/L)	Total manganese (µg/L)
				RALSTON CREEK						
06-17-83	<1.0	531	--	--	2.0	0.10	20	210	10	<10
08-09-83	E7.7	--	--	--	.77	<.005	110	430	10	30
10-05-83	E9.1	155	--	104	.63	.007	90	340	10	20
05-23-84	E4.7	272	--	162	.74	.014	--	--	--	--
06-07-84	E10.0	236	--	137	.92	.017	50	740	10	50
06-20-84	E1.1	254	--	152	.70	.014	--	--	--	--
06-04-85	E9.0	78	61	--	.36	.03	--	--	--	--
07-10-85	E2.8	64	60	--	.24	.005	--	--	--	--
08-08-85	E3.0	--	384	--	1.04	.020	--	--	--	--
08-22-85	E3.8	305	199	--	.44	.015	--	--	--	--
09-04-85	E3.0	307	203	--	.63	.013	--	--	--	--
09-18-85	E2.0	310	201	--	.45	.020	--	--	--	--
				CROKE CANAL						
07-18-84	84	94	--	--	.31	<.005	--	--	--	--
08-03-84	12	135	--	87	.36	.073	80	2,200	190	280
08-15-84	130	137	--	--	.64	.048	--	--	--	--
05-08-85	206	166	104	--	.67	.123	--	--	--	--
05-15-85	93	205	123	--	.46	.052	--	--	--	--
06-04-85	42	137	81	--	.61	.066	--	--	--	--
06-20-85	107	--	85	--	.44	.027	--	--	--	--
06-28-85	15	162	123	--	.49	.026	--	--	--	--
07-10-85	10	192	122	--	.64	.049	--	--	--	--
07-17-85	18	240	167	--	--	.036	--	--	--	--

Table 5.--*Ratios of total-nitrogen to total-phosphorus concentrations for each sampling date, Ralston Creek and Croke Canal*

[TN, total nitrogen; TP, total phosphorus; <, less than; >, greater than]

Ralston Creek		Croke Canal	
Date	TN:TP ratio	Date	TN:TP ratio
06-17-83	(1)		
08-09-83	>155:1	07-18-84	< or >61:1
10-05-83	>316:1	08-03-84	4.9:1
05-23-84	53:1	08-15-84	13:1
06-07-84	54:1	05-08-85	5.4:1
06-20-84	50:1	05-15-85	8.8:1
06-04-85	12:1	06-04-85	9.2:1
07-10-85	<48:1	06-20-85	16:1
08-08-85	52:1	06-28-85	19:1
08-22-85	29:1	07-10-85	13:1
09-04-85	48:1	07-17-85	(2)
09-18-85	22:1		

[1]Ratio not computed because low-level analysis was not used.
[2]Ratio not computed because total-nitrogen concentration not determined.

likely to be limited by phosphorus, based on ratios presented by Chiaudani and Vighi (1975), Carlson (1977, 1981), and information by W.R. Rast (U.S. Geological Survey, written commun., 1986).

The only trace elements analyzed in samples from Ralston Creek were iron and manganese (table 4). Neither of these trace elements had concentrations that exceeded drinking-water standards or aquatic-life criteria. No water samples for analysis of uranium were collected from Ralston Creek.

During the study period, Croke Canal water was pumped into Arvada Reservoir for only about 1 month during the late summer of 1985. Therefore, evaluation of the effects of the canal water on the reservoir is limited. The water samples were collected from the canal (sites CC-1 and CC-2) upstream from where the water is diverted to the reservoir. The water was analyzed for concentrations of dissolved and total nitrogen and phosphorus during 1983 through 1985 and for concentrations of major cations and anions, dissolved and total trace elements, and organic carbon only once in August 1984. No water samples for analysis of uranium were collected from Croke Canal.

In Croke Canal, dissolved-solids residue at 105 °C ranged from 81 to 167 mg/L and had a mean of 115 mg/L and a median of 119 mg/L. Concentrations of total nitrite plus total nitrate as nitrogen ranged from 0.046 to 0.206 mg/L and had a mean of 0.130 mg/L and a median of 0.140 mg/L. Concentrations of total ammonia plus total-organic nitrogen as nitrogen ranged from

<0.2 to 0.6 mg/L and had a mean of 0.4 mg/L and a median of 0.4 mg/L. Total-nitrogen concentrations ranged from 0.31 to 0.67 mg/L and had a mean of 0.51 mg/L and a median of 0.61 mg/L. Total-phosphorus concentrations ranged from <0.005 to 0.98 mg/L (not in table 4) and had a mean of 0.135 mg/L and a median of 0.049 mg/L. Compared to Ralston Creek, Croke Canal actually had smaller total-nitrogen concentrations than Ralston Creek (a mean of 0.51 mg/L in Croke Canal and a mean of 0.74 mg/L in Ralston Creek). However, the total-phosphorus concentrations are considerably larger in Croke Canal (a mean of 0.020 mg/L in Ralston Creek and a mean of 0.135 mg/L in Croke Canal). The ratios of total-nitrogen to total-phosphorus concentrations for Croke Canal (table 5) are smaller than those for Ralston Creek or for Arvada Reservoir (table 3). The ratios generally are less than 20:1 (89 percent of ratios), occasionally about 5:1 (22 percent of ratios), and about 10:1 (22 percent of ratios), indicating that algal biomass in the canal likely is nitrogen limited, or at least colimiting (Chiaudani and Vighi, 1975; Carlson, 1977, 1981). Of the biologically available nutrients (dissolved ammonia plus dissolved nitrate as nitrogen and dissolved orthophosphate as phosphorus), one analysis indicated a ratio of 3.5:1 and one analysis indicated a ratio of 35.2:1. The available nitrogen (dissolved ammonia plus dissolved nitrate as nitrogen) concentrations were less than 20 µg/L, and the smallest concentration of dissolved orthophosphate as phosphorus was 5 µg/L. Therefore, an exclusive use of water from Croke Canal ultimately could affect nutrient limitation and production of algal biomass in the reservoir. A single analysis of dissolved manganese in Croke Canal exceeded the secondary drinking-water standard on August 3, 1984, having a concentration of 190 µg/L (table 4). In addition, total-iron (2,200 µg/L, table 4), total-mercury (0.2 µg/L), and total-zinc (210 µg/L) concentrations exceeded the criteria for maintenance of aquatic life on the same sampling date.

Biological Quality

Description and quantification of the densities, biomass, and kinds of organisms inhabiting a lake is important for determining the overall quality and trophic state of the lake. Several species of phytoplankton have been identified in the literature as indicators of excessive organic pollution, taste and odor, or other nuisances in water. The densities and kinds of zooplankton may be important as controls on phytoplankton densities. Chlorophyll a concentrations are used extensively in models to determine relations of phytoplankton densities with nutrient inputs and to estimate primary production. Algal-growth potential also is useful to assess limiting-growth factors. Each of these organisms, properties, and determinations will be discussed separately.

Phytoplankton

During the study, 193 taxa of phytoplankton were identified at site A in Arvada Reservoir. Fewer taxa of phytoplankton were identified at the other sites in the reservoir because of less extensive sampling. However, there were 72 taxa identified at site B (18 different from those at site A), 52 at site C (11 different from those at site A), and 113 at site D (18 different from those at site A). No samples for phytoplankton analyses were collected

at site E. A total of 233 taxa of phytoplankton were identified at all the sampling sites in the reservoir. These phytoplankton were represented by 51 taxa of green algae, 117 taxa of diatoms, 14 taxa of golden-brown algae, 24 taxa of blue-green algae, 13 taxa of cryptomonads, 6 taxa of euglenoids, 7 taxa of dinoflagellates, and 1 taxa of yellow-green algae.

Among the composite samples collected, the largest density of phytoplankton at site A was 23,000 cells/mL collected on April 17, 1985 (fig. 13). This moderately large density was due to a diatom bloom of *Rhizosolenia eriensis*. The smallest density of phytoplankton at site A was 2,200 cells/mL collected on February 2, 1984. The large density of green algae collected on August 15, 1984, was due to a bloom of *Chlorella ellipsodea*, which had a

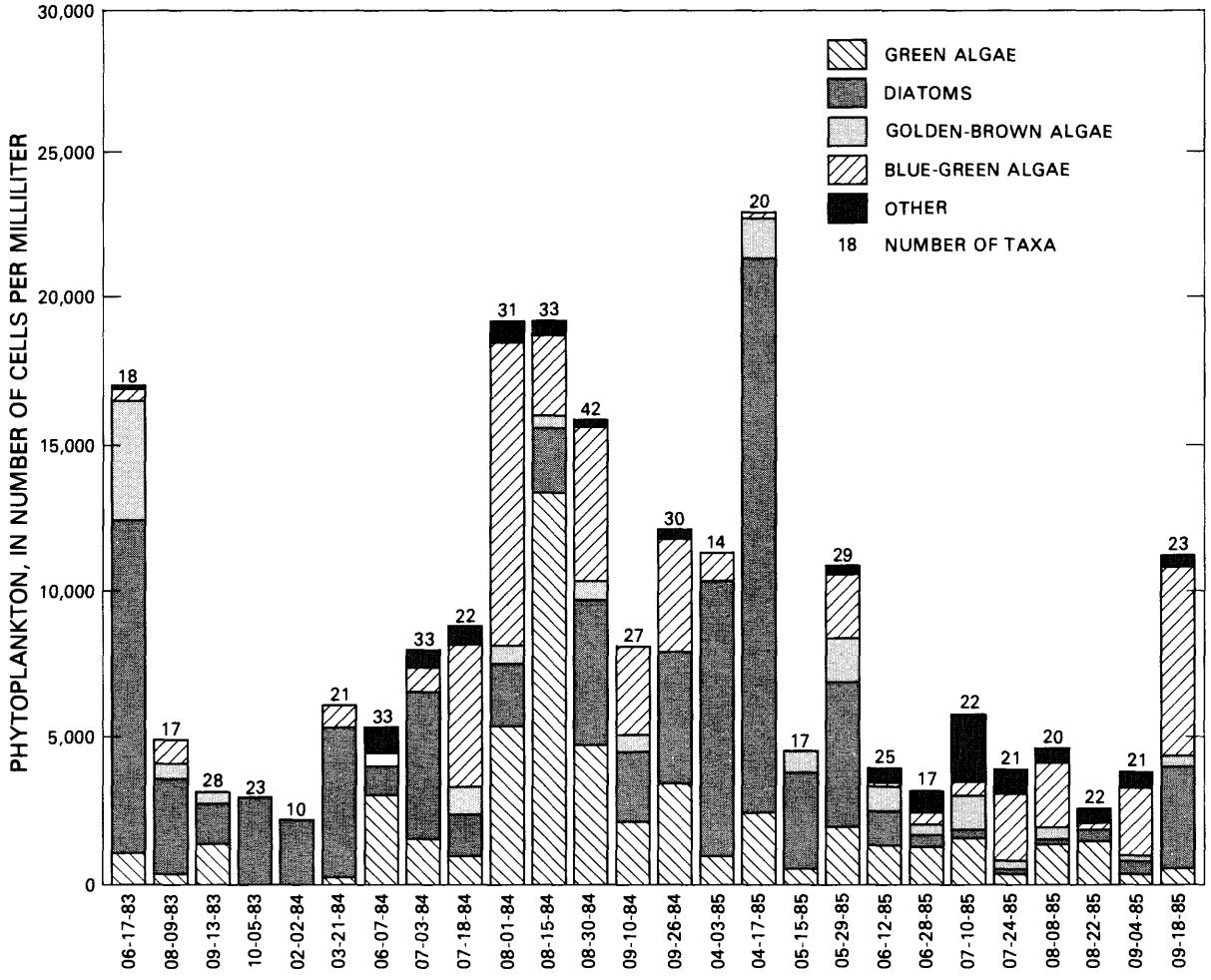

Figure 13.--Taxonomic composition and densities of phytoplankton collected from composite samples to average densities of zooplankton collected from photic (total if only available) tows for all sampling dates at sites A, B, C, and D.

density of 12,000 cells/mL. The large density of blue-green algae on August 1, 1984, was due to a bloom of *Microcystis firma*, which had a density of 9,500 cells/mL. The green algae and the blue-green algae are abundant only during the warmer sampling periods as shown in figure 13. Among the other sampling sites in the reservoir, the largest density of phytoplankton (29,000 cells/mL) was collected from site D on August 15, 1984. The smallest density of phytoplankton (1,400 cells/mL) was collected from site B on September 13, 1983.

The number of taxa identified from composite samples at site A ranged from 10 on February 2, 1984, to 42 on August 30, 1984 (fig. 13). The phytoplankton diversity-index values for composite samples collected at site A ranged from 1.74 on April 3, 1985, to 4.18 on August 30, 1984 (fig. 14). In the reservoir, the smaller diversity-index values were associated with dates when diatoms predominated in the sample, but had few taxa, and when few or no taxa from other classes were collected in the sample. These dates would be considered transition dates (winter and spring) when environmental conditions were not favorable for production of many types of phytoplankton. The larger

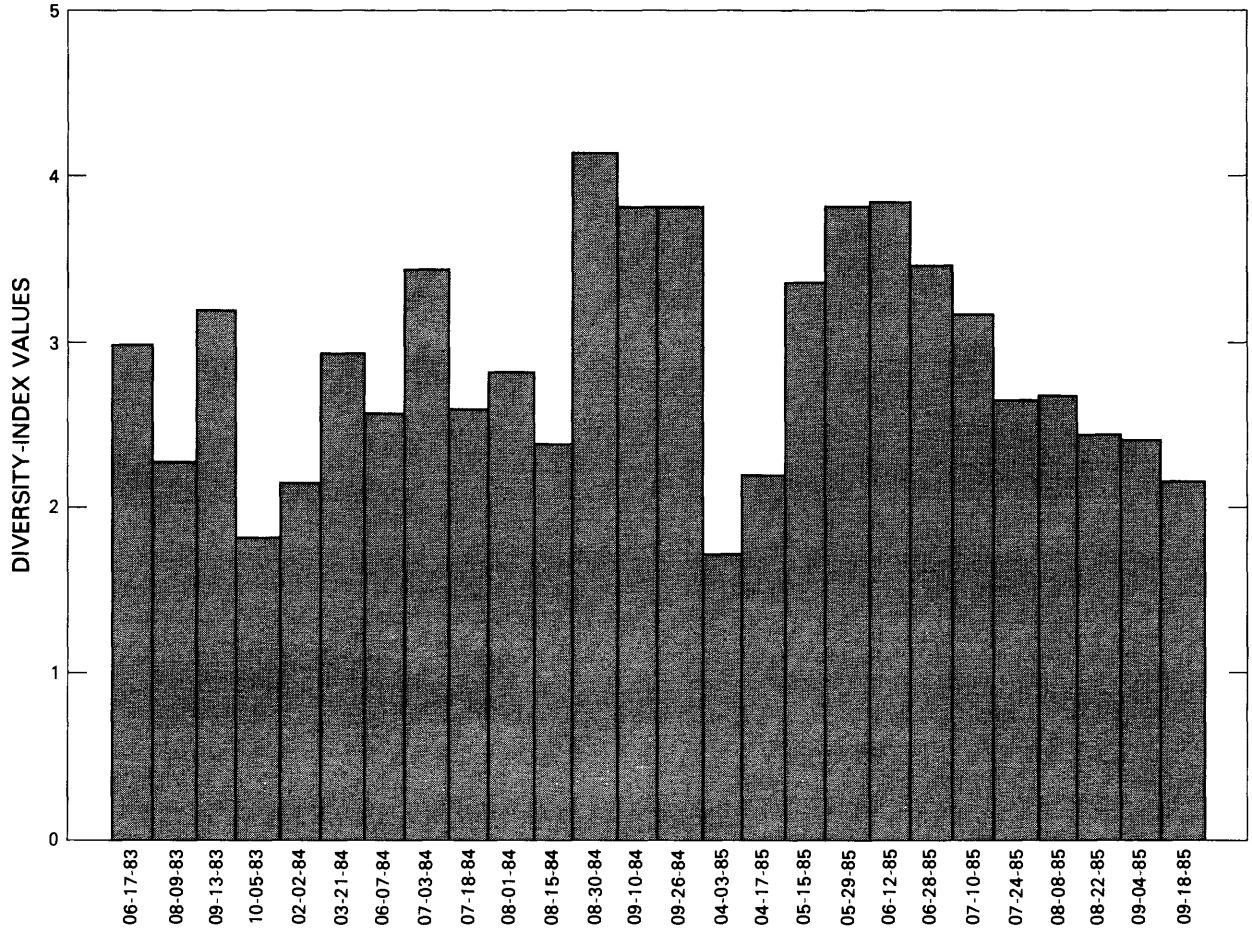

Figure 14.--Diversity-index values for phytoplankton collected from composite samples for all sampling dates at site A.

diversity-index values occurred during the late spring, after the spring-circulation period, and during midsummer when environmental conditions were favorable for production of many types of phytoplankton (warm water and available nutrients). The resultant diversity-index changes probably are more or less cyclic and, during a single year, would seem to be related to ecological succession. This succession is a gradual change in community composition from loosely organized systems of phytoplankton that have large ratios of productivity to biomass to stabler communities that have smaller ratios of productivity to biomass.

The average diversity-index value for site A was 2.8, which is comparable to the value of 3.0 specified as a value characteristic of unpolluted water. Among the other sites, diversity-index values ranged from 1.85 at site C on October 5, 1983, to 4.65 at site B on September 13, 1983.

The three numerically dominant phytoplankton taxa collected from composite samples for each sampling date at site A are listed in table 6. For some dates, codominant taxa existed, which increased the number of dominant taxa. Diatoms dominated (indicated by largest density) on 10 sampling dates; blue-green algae dominated on 7 sampling dates; green algae dominated on 5 sampling dates; cryptomonads dominated on 2 sampling dates; golden-brown algae dominated on 2 sampling dates; and a dinoflagellate dominated on 2 sampling dates.

Of the diatoms, *Rhizosolenia eriensis* dominated most often (five sampling dates) (table 6). This alga has been associated with water having large nutrient concentrations (Lowe, 1974). The largest density was 14,000 cells/mL, or 60 percent of the total density, on April 17, 1985. This also was the largest single density of an alga recorded during the study. Of the other dominant diatoms, *Asterionella formosa* has a fishy taste and is a filter-clogging alga; *Cyclotella kutzingiana* is not specifically associated with pollution but is capable of producing blooms; *Synedra radians* is associated with eutrophic water (Lowe, 1974); and *Diatoma tenue* var *elongatum* is not associated with pollution.

Of the blue-green algae, *Chroococcus dispersus* and *Aphanocapsa delicatissima* each dominated on two sampling dates. Neither of these species is identified among those by Palmer (1969) as being tolerant of organic pollution or identified by Palmer (1977) as being associated with water pollution. However, the genus *Aphanocapsa* has been characterized by Palmer (1977) as a slime-producing alga. The largest single density of a blue-green alga was for *Microcystis firma* (9,500 cells/mL) on August 1, 1984. It also is not identified by Palmer (1969) as a pollution-tolerant alga, but the genus *Microcystis* is considered a toxic freshwater alga that can cause hay-fever symptoms and is capable of producing blooms. *Microcystis* does not fix nitrogen, which is important when discussing nutrient ratios and limitations in subsequent sections of this report.

Of the green algae, *Chlorella ellipsodea* dominated most often (twice) and had the single largest density of a green alga (12,000 cells/mL) on August 15, 1984. The genus *Chlorella* has been identified by Palmer (1977) as having a musty odor, causing coloration of water, and having a capability of producing

blooms. Of the other green algae that were dominant at site A, *Mesotaenium* sp., *Sphaerocystis schroeteri*, and *Scenedesmus serratus* are not associated with pollution, although *Scenedesmus* is fourth on a list of 60 genera that are considered to be tolerant of organic pollution (Palmer, 1969). *Chlamydomonas* is capable of producing blooms, has a musty odor and a sweet taste, and is third on a list of 60 genera that are considered tolerant of organic pollution.

Of the cryptomonads, *Cryptomonas erosa* dominated on two sampling dates (maximum density of 2,000 cells/mL on July 10, 1985). *Cryptomonas erosa* has been identified by Palmer (1977) as a taste-and-odor alga that has a sweet taste and as number 28 on a list of 80 species tolerant of organic pollution (Palmer, 1969).

Of the golden-brown algae, *Uroglenopsis americana* codominated on June 17, 1983, and had a density of 4,000 cells/mL. This alga has been identified by Palmer (1977) as causing a cucumber or fishy odor. An unidentified flagellate dominated on June 12, 1985, with a density of 800 cells/mL.

Of the dinoflagellates, *Ceratium hirundinella* dominated on two sampling dates and had a maximum density of 3,600 cells/mL on September 4, 1985. *Ceratium hirundinella* has been identified by Palmer (1977) as a taste-and-odor alga that has a fishy to pronounced septic odor and a bitter taste. This alga was collected with the zooplankton samples, but occasionally it was simultaneously collected with the phytoplankton using the water-sampling bottle. This alga has been classified by different taxonomists as a protozoan (zooplankton) (Pennak, 1953; Needham and Needham, 1962) and as a dinoflagellate (phytoplankton) (Smith, 1950; Edmondson, 1959; Palmer, 1977). For this study, the organism is classified and tabulated as part of the phytoplankton community. The number of *Ceratium hirundinella* collected in the photic tows was converted to cells per milliliter and averaged for the replicate photic tows. Because the organism is clumped in the reservoir, more organisms were collected using a net, which samples a larger volume of water than does the water-sampling bottle.

The most dominant taxa collected at the other sampling sites in the reservoir are listed in table 7. On most sampling dates, the dominant taxa were the same as those among the three dominant taxa at site A. On at least one sampling date, sites B and D had a dominant taxon that was not among the three dominant taxa at site A. This was expected because sites B and D are the farthest distance from site A and are the shallowest sites, and because phytoplankton normally have a patchy distribution. Therefore, these sites may have an environment that occasionally supports different types of algae.

A comparison of phytoplankton densities collected from composite samples and average densities of zooplankton collected from photic or total-depth tows for all sampling dates at site A is shown in figure 15. The phytoplankton densities varied somewhat randomly throughout the sampling period, although the largest densities measured were during the spring when the zooplankton densities generally were small. Zooplankton densities increased through the late spring and summer and decreased during fall, winter, and early spring. However, when combined with other limiting factors, zooplankton probably affect the phytoplankton population. Experiments done by Berquist and others

Table 6.--*Dominant taxa of phytoplankton collected from composite samples for each sampling date at site A*

[cells/mL, cells per milliliter; D, diatoms; G, green algae; Crypt, cryptomonads; BG, blue-green algae; GB, golden-brown algae; Dino, dinoflagellates; --, not applicable]

Date	Taxa	Density (cells/mL)	Taxa	Density (cells/mL)	Taxa	Density (cells/mL)
06-17-83	*Asterionella formosa* (D)	4,000	*Synedra radians* (D)	2,900	*Diatoma tenue* var *elongatum* (D)	1,600
	Uroglenopsis americana (GB)	4,000	--	--	*Fragilaria crotonensis* (D)	1,600
08-09-83	*Rhizosolenia eriensis* (D)	2,700	*Aphanothece* sp. (BG)	560	*Dinobryon divergens* (GB)	490
09-13-83	*Sphaerocystis schroeteri* (G)	1,200	*Dinobryon divergens* (GB)	390	*Cyclotella kutzingiana* (D)	330
	--	--	--	--	*Navicula miniscula* (D)	330
10-05-83	*Cyclotella kutzingiana* (D)	1,800	*Melosira italica* (D)	630	*Cyclotella meneghiniana* (D)	380
02-02-84	*Asterionella formosa* (D)	1,000	*Diatoma tenue* var *elongatum* (D)	700	*Synedra ulna* var *longissima* (D)	180
03-21-84	*Diatoma tenue* var *elongatum* (D)	2,200	*Asterionella formosa* (D)	1,700	*Aphanocapsa* sp. (BG)	400
06-07-84	*Chlamydomonas* sp. 1 (G)	2,900	*Chroomonas* sp. (Crypt)	590	*Rhizosolenia eriensis* (D)	530
07-03-84	*Rhizosolenia eriensis* (D)	1,900	*Cyclotella kutzingiana* (D)	1,500	*Chlamydomonas* sp. 1 (G)	1,200
07-18-84	*Microcystis* sp. (BG)	4,800	*Mallomonas* sp. (GB)	980	*Cyclotella kutzingiana* (D)	750
08-01-84	*Microcystis firma* (BG)	9,500	*Chlorella ellipsodea* (G)	3,400	*Cyclotella kutzingiana* (D)	810
08-15-84	*Chlorella ellipsodea* (G)	12,000	*Aphanothece* sp. (BG)	2,000	*Cyclotella kutzingiana* (D)	880
08-30-84	*Chlorella ellipsodea* (G)	2,600	*Chroococcus limneticus* (BG)	2,400	*Aphanocapsa delicatissima* (BG)	2,100
09-10-84	*Chroococcus dispersus* (BG)	1,500	*Cyclotella kutzingiana* (D)	1,100	*Pteromonas limneticus* (G)	850
	--	--	*Aphanothece* sp. (BG)	1,100	--	--

Table 6.--*Dominant taxa of phytoplankton collected from composite samples for each sampling date at site A*--Continued

Date	Taxa	Density (cells/mL)	Taxa	Density (cells/mL)	Taxa	Density (cells/mL)
09-26-84	*Chroococcus dispersus* (BG)	2,400	*Pteromonas limneticus* (G)	1,900	*Chroococcus limneticus* (BG)	1,400
04-03-85	*Rhizosolenia eriensis* (D)	8,100	*Ankistrodesmus falcatus* (G)	800	*Synedra radians* (D)	460
04-17-85	*Rhizosolenia eriensis* (D)	14,000	*Cyclotella kutzingiana* (D)	1,800	*Dinobryon divergens* (GB)	1,400
	--	--	*Ankistrodesmus falcatus* (G)	1,800	--	--
05-15-85	*Rhizosolenia eriensis* (D)	940	*Synedra radians* (D)	800	*Diatoma tenue* var *elongatum* (D)	650
05-29-85	*Synedra radians* (D)	2,200	*Aphanocapsa delicatissima* (BG)	1,800	*Cyclotella kutzingiana* (D)	1,200
	--	--	--	--	*Rhizosolenia eriensis* (D)	1,200
06-12-85	Flagellate (GB)	800	*Chlorococcum* sp. (G)	620	*Cyclotella kutzingiana* (D)	600
06-28-85	*Scenedesmus serratus* (G)	570	*Cyclotella kutzingiana* (D)	400	*Lyngbya nana* (BG)	340
	Cryptomonas erosa (Crypt)	570	--	--	--	--
07-10-85	*Cryptomonas erosa* (Crypt)	2,000	*Scenedesmus* sp. (G)	1,100	*Chrysochromulina parva* (GB)	710
07-24-85	*Aphanocapsa delicatissima* (BG)	2,300	*Ceratium hirundinella* (Dino)	310	*Cryptomonas erosa* (Crypt)	310
08-08-85	*Aphanocapsa delicatissima* (BG)	2,200	*Scenedesmus* serratus (G)	1,100	*Dinobryon divergens* (GB)	360
	--	--	*Ceratium hirundinella* (Dino)	1,100	--	--
08-22-85	*Ceratium hirundinella* (Dino)	2,900	*Mesotaenium* sp. (G)	1,200	*Melosira italica* (D)	280
09-04-85	*Ceratium hirundinella* (Dino)	3,600	*Aphanothece* sp. (BG)	1,600	*Aphanocapsa delicatissima* (BG)	770
09-18-85	*Aphanothece* sp. (BG)	6,400	*Cyclotella kutzingiana* (D)	3,200	*Ceratium hirundinella* (Dino)	990

Table 7.--*Dominant taxa of phytoplankton collected from composite samples for each sampling date at sites B, C, and D*

[cells/mL, cells per milliliter; GB, golden-brown algae; D, diatoms; G, green algae; BG, blue-green algae; --, no data]

Date	Site B		Site C		Site D	
	Taxa	Density (cells/mL)	Taxa	Density (cells/mL)	Taxa	Density (cells/mL)
06-17-83	*Uroglenopsis americana* (GB)	5,400	--	--	--	--
08-09-83	*Rhizosolenia eriensis* (D)	1,900	*Rhizosolenia eriensis* (D)	1,700	--	--
09-13-83	*Melosira italica* (D)	180	*Dinobryon divergens* (GB)	690	--	--
10-05-83	*Cyclotella kutzingiana* (D)	1,800	*Cyclotella kutzingiana* (D)	1,500	--	--
03-21-84	*Diatoma tenue* var *elongatum* (D)	2,100	*Asterionella formosa* (D)	2,800	--	--
06-07-84	--	--	--	--	*Chlamydomonas* sp. 1 (G)	1,800
07-03-84	--	--	--	--	*Rhizosolenia eriensis* (D)	1,700
	--	--	--	--	*Synedra radians* (D)	1,700
07-18-84	--	--	--	--	*Microcystis* sp. (BG)	1,100
08-15-84	--	--	--	--	*Chlorella ellipsodea* (G)	15,000
08-30-84	--	--	--	--	*Microcystis firma* (BG)	12,000
09-10-84	--	--	--	--	*Chroococcus dispersus* (BG)	4,000
09-26-84	--	--	--	--	*Chroococcus dispersus* (BG)	3,800

(1985) have indicated that different taxa and sizes of phytoplankton respond differently to zooplankton communities. Shifts in size can alter light transparency (Edmondson, 1980) and productivity (Carpenter and Kitchell, 1984), and size-selective predation probably accounts for some of the variability in these limnological conditions in Arvada Reservoir.

A comparison of phytoplankton and zooplankton densities at sites A, B, C, and D (fig. 16) is shown in a scatter plot for the study period. In general, when phytoplankton densities are large, the zooplankton densities are small. Zooplankton seem to be a control of phytoplankton densities. A complete discussion of the zooplankton follows the "Algal-Growth Potential" section.

Chlorophyll a

Concentrations of chlorophyll a ranged from 1.1 to 20.4 µg/L in all samples from site A, and the mean concentration for the samples was 6.2 µg/L. The large value of 20.4 µg/L was analyzed using methanol extraction (1 of 3

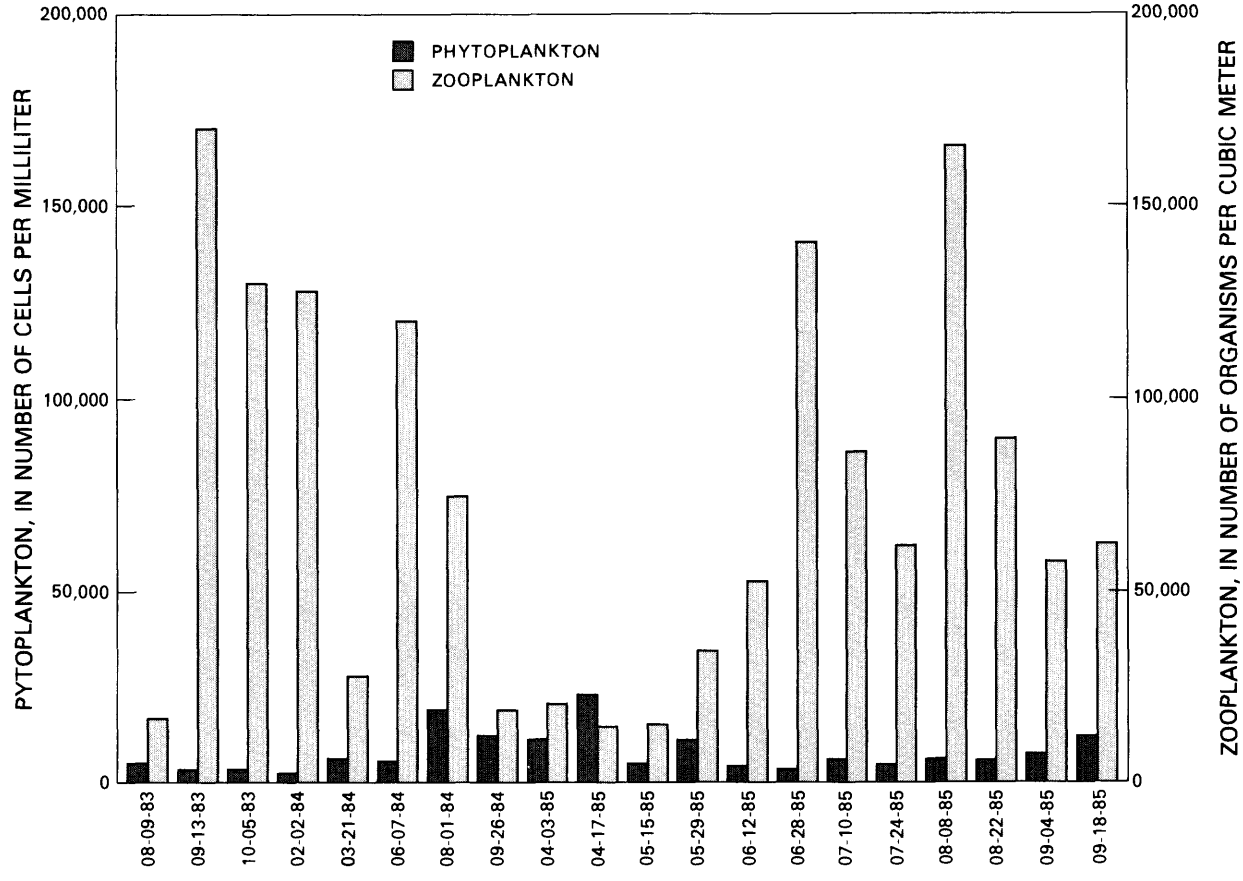

Figure 15.--Comparison of densities of phytoplankton collected from composite samples to average densities of zooplankton collected from photic (total if only available) tows for all sampling dates at site A.

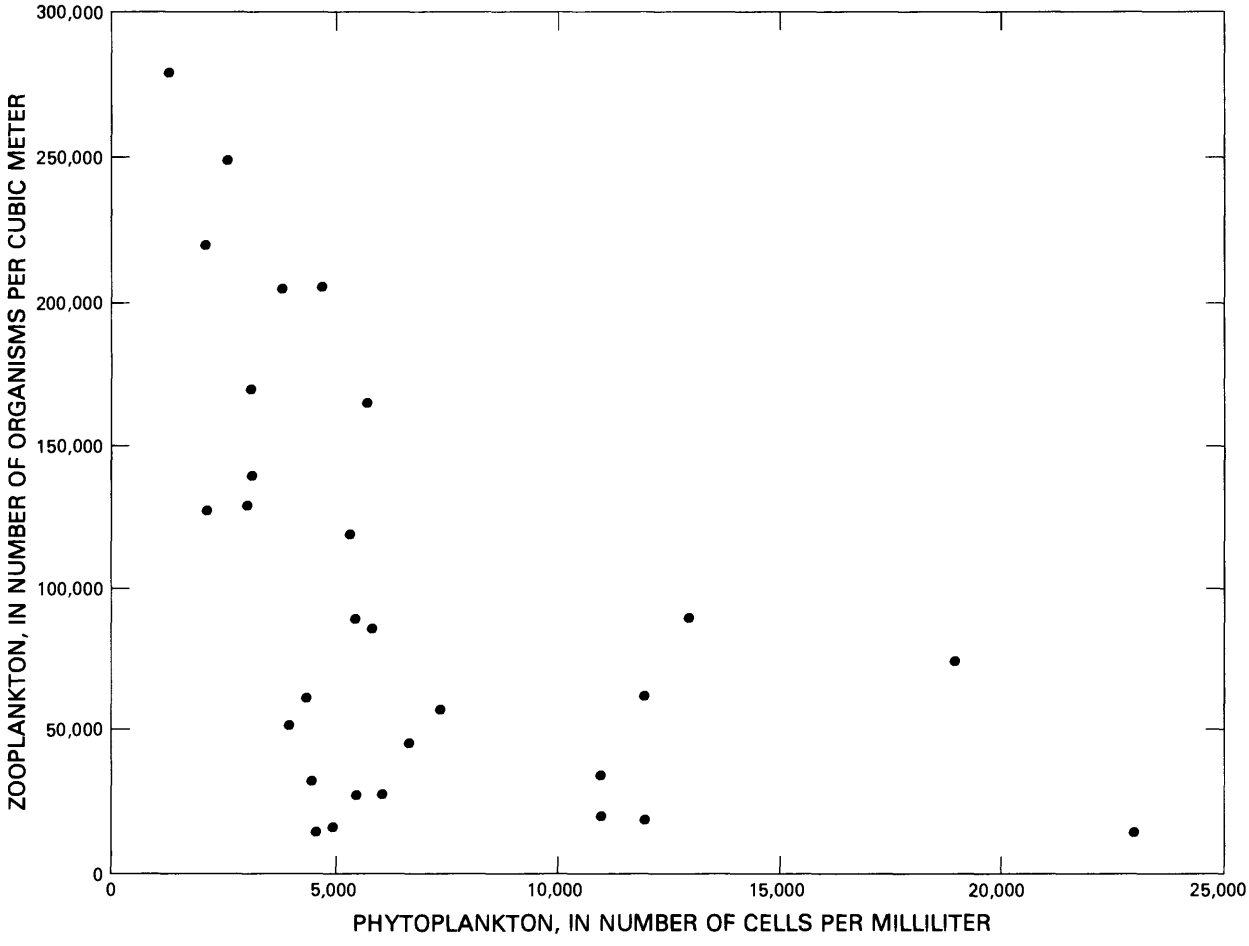

Figure 16.--Relation between densities of phytoplankton collected from composite samples and average densities of zooplankton collected from photic (total if only available) tows for all sampling dates at sites A, B, C, and D.

out of 35 sampling dates), instead of acetone extraction, because of complications with digestion of a new type of filter. Analyses using methanol extraction usually yields greater concentrations for chlorophyll a than do other extraction methods (Holm-Hansen and Riemann, 1978). The mean for only the composite samples collected at site A was 5.7 µg/L. During 1985, samples for chlorophyll a were collected biweekly and analyzed separately for the surface, middle, and bottom parts of the euphotic zone and for a composite of the euphotic zone. The mean values for these samples were: surface, 6.5 µg/L; middle, 7.2 µg/L; bottom, 6.1 µg/L; and composite, 6.6 µg/L. Based on comparison with the 3-year mean concentrations from composite samples, no significant difference ($p > 0.05$) between the mean concentrations from the four euphotic-zone samples and the 3-year mean composite samples was indicated. Therefore, concentrations of chlorophyll a from composite samples seem to be sufficient for analyzing and interpreting chlorophyll a concentrations in Arvada Reservoir.

Chlorophyll a concentrations for each sampling date at site A are shown in figure 17. More than 60 percent of the chlorophyll a concentrations were 5 µg/L or less, and 85 percent were 10 µg/L or less. Among the other sampling sites in the reservoir, chlorophyll a concentrations ranged from 0.0 to 20.0 µg/L, and the mean was 5.4 µg/L.

During recent years, predictive techniques have been used to assess limnological conditions and changes in water quality. The greatest emphasis has been on the relation between nutrients and phytoplankton variables. Phosphorus generally is considered to be the major limiting factor for phytoplankton biomass in most lakes. In Arvada Reservoir, phosphorus seems to be the major limiting nutrient for algal biomass (see "Algal-Growth Potential" section); therefore, an attempt was made to learn how phosphorus and chlorophyll a data collected in Arvada Reservoir compared to the predictive relations developed by Dillon and Rigler (1974).

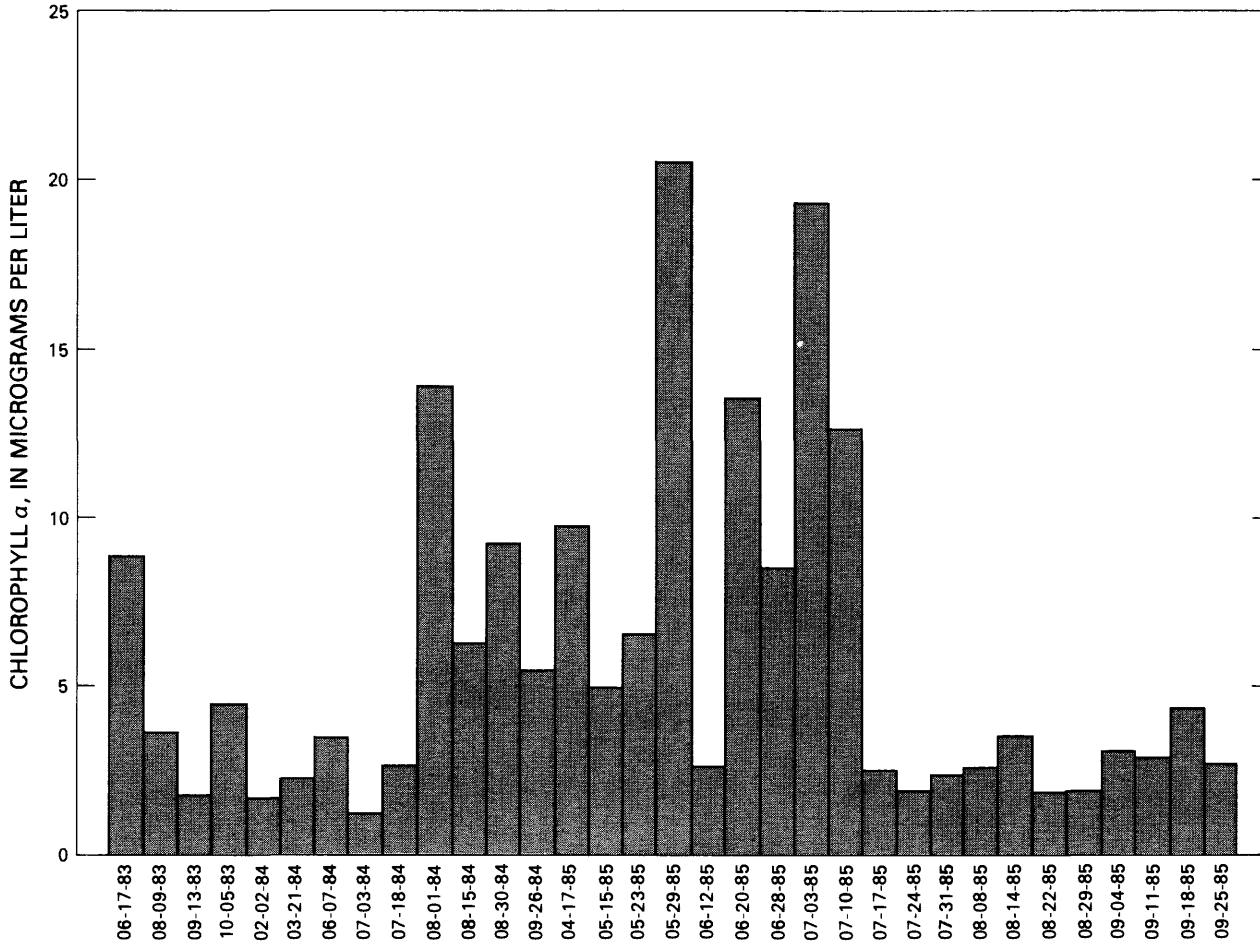

Figure 17.--Chlorophyll a concentrations from composite samples for all sampling dates at site A.

According to Dillon and others (1978), a literature search yielded 16 different phosphorus-chlorophyll a relations. However, there are two basic types of models for predicting chlorophyll a: (1) Those based on nutrient-concentration data only, and (2) those based on phosphorus-budget data modified by hydrologic characteristics (LaBaugh and Winter, 1984). The confidence interval of the chlorophyll a prediction based on total-phosphorus concentrations are described by Dillon and Rigler (1974). The measurements for total phosphorus and chlorophyll a used to predict chlorophyll a concentrations were within the range of those used to develop the model. Based on the model (Dillon and Rigler, 1974), the measured mean concentrations of chlorophyll a generally were not within the 95-percent confidence interval of the predicted mean chlorophyll a concentrations (fig. 18). The data indicate that there is more chlorophyll a in the reservoir than can be attributed to phosphorus concentrations alone. Because measured chlorophyll a concentrations from Ralston Creek generally were less than 2 µg/L (1983 data only, Britton and Gaggiani,

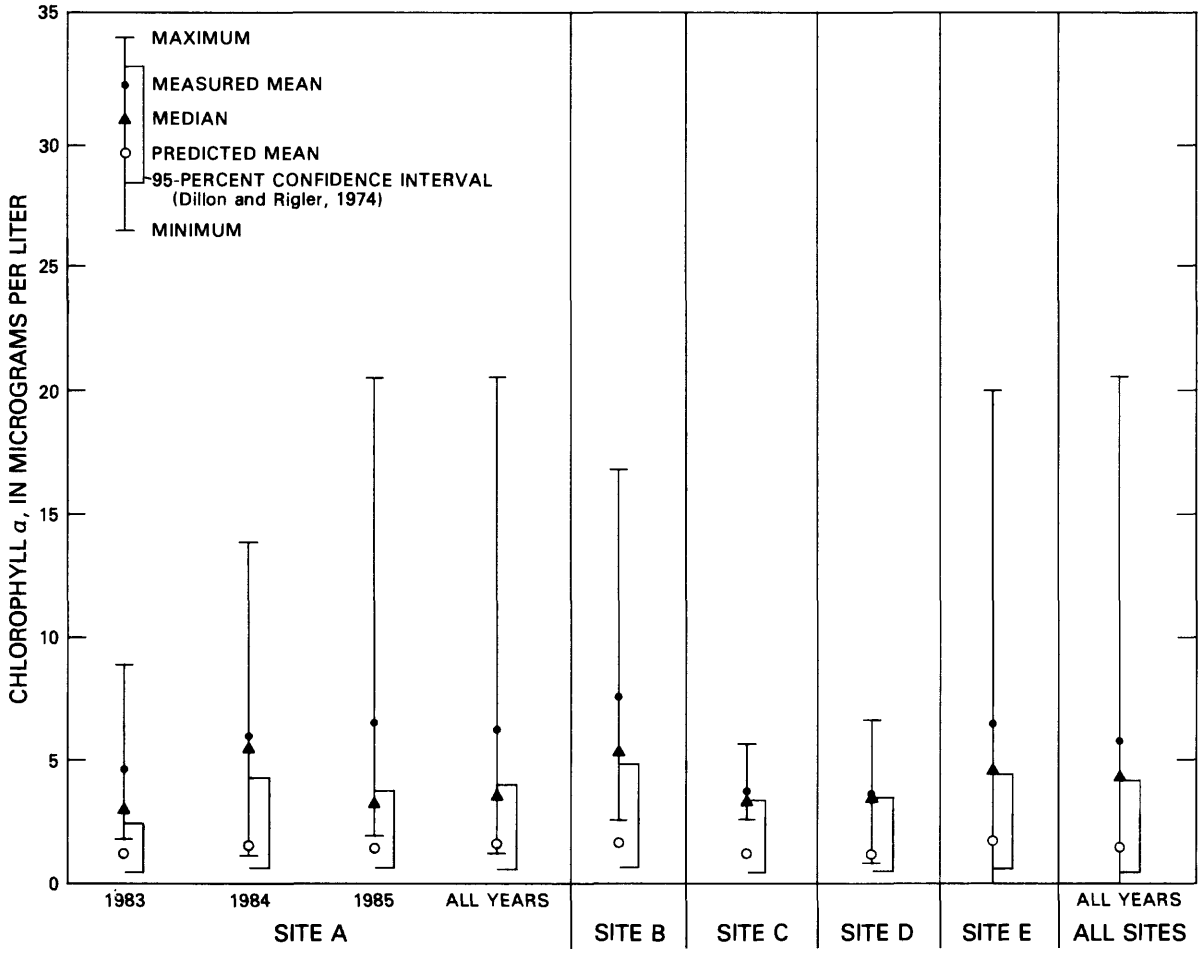

Figure 18.--Measured and predicted chlorophyll a concentrations from composite samples for all sampling sites in the reservoir.

1986), the data indicate that other factors affect chlorophyll *a* concentrations in the reservoir. However, the median values, which may be better indicators of the chlorophyll *a* distribution because they are unaffected by the extremes, sometimes were within the 95-percent confidence interval of the predicted mean concentration of chlorophyll *a*. However, the uncertainty of using chemical data to predict biological changes and the sources of variability in measurements of phosphorus and chlorophyll *a* have been well documented in the literature.

As Dillon and others (1978) indicate, some of the uncertainty in the various phosphorus-chlorophyll *a* relations are based on data sets where no two of the studies used the same sampling periods, laboratory procedures, or sampling zones in the lakes evaluated. Furthermore, most of the data used to determine phosphorus-chlorophyll *a* relations are based on one sample from lakes that are predominantly located on the Precambrian shield in North America or in Japan. Substantial variability in a single phosphorus-chlorophyll *a* relation using ideal sampling and analytical techniques can occur. For example, total phosphorus may not be universally the best measure of available phosphorus, and chlorophyll *a* may not properly represent algal biomass (Nicholls and Dillon, 1978). Some forms of phosphorus in lake water may not be available, yet are recorded as a fraction of the total phosphorus. In addition, several factors that affect the chlorophyll *a* content of algal cells seem to be: (1) Incidental radiation, (2) species composition, (3) nutrient supply, and (4) innate physiology (Nicholls and Dillon, 1978). As a result of these factors, chlorophyll *a* content of algal cells can vary by two orders of magnitude (0.1-9.7 percent of fresh weight).

If light were the major controlling factor for chlorophyll *a*, similar chlorophyll *a* concentrations would be expected in Arvada Reservoir for dates when Secchi-disk measurements were similar, although nonalgal turbidity also is a factor. However, this similarity did not occur, probably because nutrient limitation and species composition affected chlorophyll *a* concentrations. Because the reservoir possibly is limited or colimited by nitrogen on occasion, nitrogen would be a controlling factor for chlorophyll *a* concentrations, but it is not a part of the typical predictive models described in the literature. In fact, Talling (1966) indicated that nitrogen availability can cause a tenfold-to-twelvefold change in chlorophyll *a* content of diatoms (the predominant class of algae present in Arvada Reservoir) under constant light conditions. Similarly, with constant light and constant species composition, Antia and others (1963) reported a fourfold change in chlorophyll *a* content of algae that was dependent on inorganic nitrogen. Knowlton and others (1984) reported that limnological conditions in lakes routinely vary as a result of random or cyclic processes, which do not result in any long-term alteration of the lake ecosystem. Their analyses indicated that the magnitude of this short-term, within-lake variability can be large compared to the measured differences among or within lakes. This variability results in uncertainty when measuring the average condition of lakes over time and decreases detection of differences between lakes and the effects of long-term changes within lakes. In Arvada Reservoir, the nonsteady-state condition of the reservoir during filling and during periods of withdrawal could have produced atypical results in regard to in-lake water quality and trophic state.

Algal biomass, as measured by chlorophyll *a* concentrations, is smaller in Ralston Creek than in Croke Canal. The mean chlorophyll *a* concentration for Ralston Creek was 1.2 µg/L (only 1983 data were available) and was 6.4 µg/L for Croke Canal (1983-84). Because Croke Canal has larger nutrient concentrations and algal biomass, prolonged periods of pumping this water into Arvada Reservoir possibly could increase algal biomass in the reservoir, as opposed to an exclusive inflow from Ralston Creek.

Algal-Growth Potential

The results of the algal-growth-potential determinations for site A are shown in figure 19. The growth in samples spiked with nutrients were nearly always larger than the growth in control samples. The initial indication, based on ratios of total-nitrogen to total-phosphorus concentrations, was that Arvada Reservoir is phosphorus limited. The algal-growth-potential determinations indicate that on four out of seven sampling dates, when individual nutrient spikes were used (September 15, 1983; October 5, 1983; June 7, 1984; and August 3, 1984), phosphorus spikes produced marked increases in growth as compared to the control, and on two sampling dates (June 7, 1984, and August 3, 1984), nitrogen spikes produced marked increases in growth as compared to

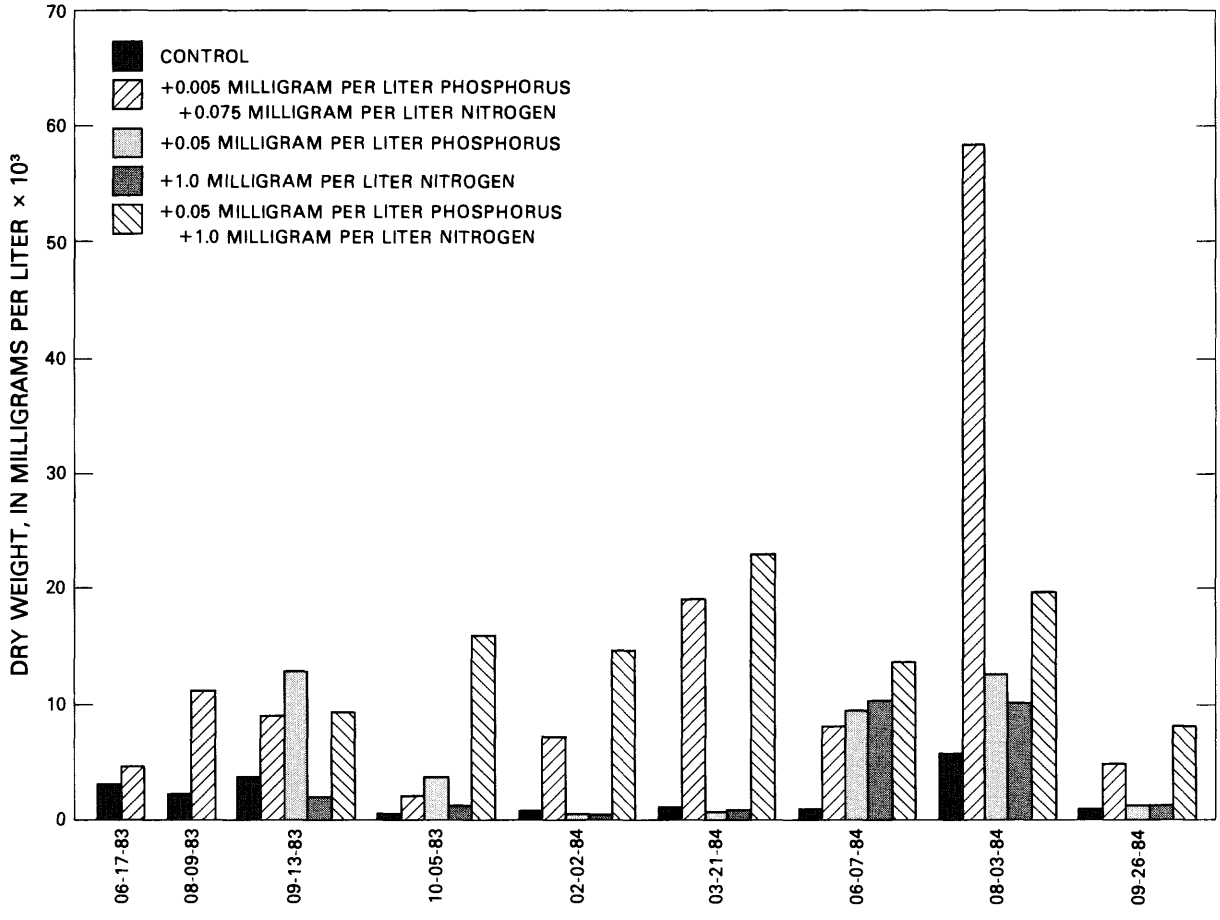

Figure 19.--Results of algal-growth-potential determinations for site A.

the control. However, addition of phosphorus produced the maximum growth among all spikes only in the September 15, 1983, sample. Otherwise, combined spikes of phosphorus and nitrogen produced the largest increases in growth on all other sampling dates, as compared to the growth produced by phosphorus spikes alone. For the dates sampled, except the June 7, 1984, sample, the data indicate that phosphorus initially seems to be limiting, but an addition of nitrogen markedly increases the growth in the samples. For the August 3, 1984, sample, largest growth occurred using the spike of 0.005 mg/L of phosphorus plus 0.075 mg/L of nitrogen. This result is suspect because such a substantial growth using these smaller concentrations of phosphorus and nitrogen would not be expected to occur.

The results of algal-growth-potential determinations from the other sampling sites (sites B, C, and D) (fig. 20) in the reservoir are similar to the results from site A. No algal-growth-potential determinations were made for site E. On all sampling dates, except on September 15, 1983, the combined spikes of phosphorus and nitrogen produced the most growth. In summary, on most sampling dates, phosphorus and nitrogen seem to be limiting. Only on the September 15, 1983, and October 5, 1983, sampling dates did the phosphorus spike produce substantially more growth as compared to the nitrogen spike. Therefore, concentrations of phosphorus and nitrogen seem to be small in Arvada Reservoir, and addition of both nutrients causes substantial growth of the test alga. However, different algae respond differently to nutrient stimulation (Lange, 1971). This growth of algae is used only as an indicator of limiting nutrients and is used with ratios of total-nitrogen to total-phosphorus concentrations to substantiate the results of the algal-growth-potential determinations.

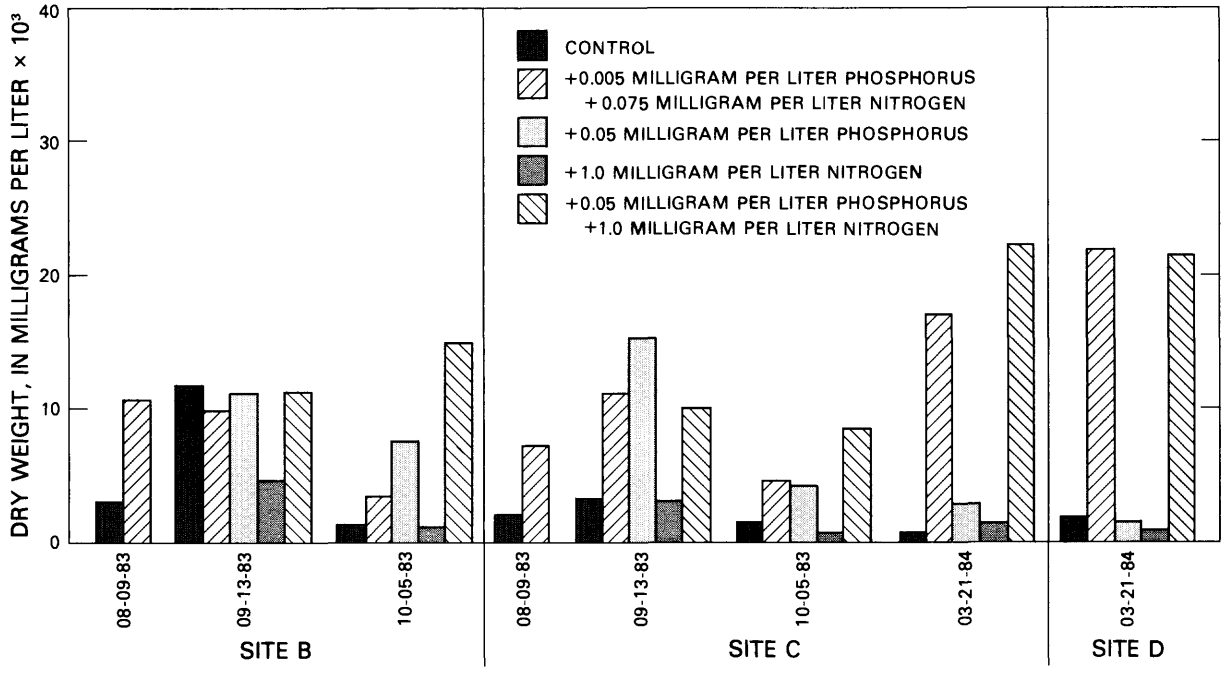

Figure 20.--Results of algal-growth-potential determinations for sites B, C, and D.

The results of algal-growth-potential determinations for Ralston Creek and Croke Canal are shown in figure 21. On all sampling dates, the combined spikes of phosphorus and nitrogen produced the most growth. As at site A, the results of the August 3, 1984, sample for Croke Canal indicating largest

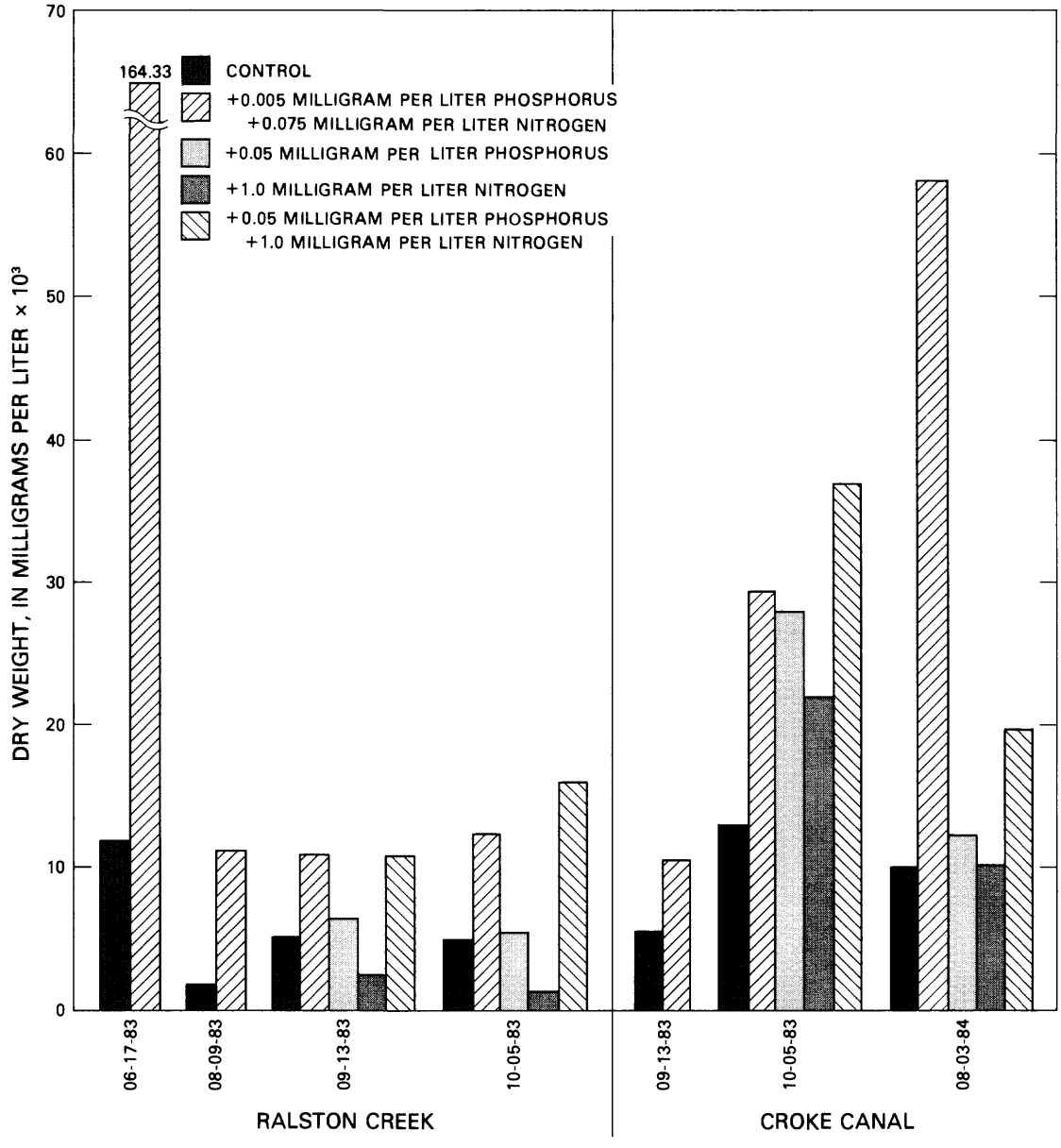

Figure 21.--Results of algal-growth-potential determinations for Ralston Creek and Croke Canal.

growth using the spike of 0.005 mg/L of phosphorus plus 0.075 mg/L of nitrogen is suspect. For Ralston Creek and Croke Canal, the spikes of phosphorus produced more growth than did the spikes of nitrogen, indicating that phosphorus initially is limiting, but nitrogen is in small enough concentrations that its addition markedly increases growth in the cultures. However, based on the dry-weight values (fig. 21), nitrogen spikes in samples from Croke Canal produced nearly the same increase in growth as did phosphorus spikes, and in combination with the results of ratios of total-nitrogen and total-phosphorus concentrations, Croke Canal seems to be more nitrogen limited than is Ralston Creek.

Zooplankton

During the study, 23 taxa of zooplankton were identified among the photic and total-depth tows at site A. Fewer taxa of zooplankton were identified at the other sampling sites in Arvada Reservoir because of less extensive sampling. No samples for zooplankton analyses were collected at site E. However, there were 16 taxa identified at site B, 17 at site C, and 12 at site D (1 different from those at site A). Therefore, a total of 24 taxa of zooplankton were identified at sampling sites in the reservoir. These zooplankton were represented by 6 taxa of cladocerans, 2 taxa of copepods, 1 taxon of protozoans, and 15 taxa of rotifers.

The largest concentration of zooplankton for a single tow collected at site A was 180,000 organisms/m^3 on August 8, 1985. The average of the two photic tows was 165,000 organisms/m^3 (fig. 22). This moderately large concentration was because of copepod concentrations, including an average of 90,500 organisms/m^3 of nauplii (immature copepods). The smallest concentration of zooplankton collected in a single tow was 10,000 organisms/m^3 on April 17, 1985. The average of the two photic tows was 14,500 organisms/m^3 (fig. 22).

Generally, rotifers and cladocerans were more dominant during the first sampling dates of the study period; rotifers comprised about 85 percent of the zooplankton sample on February 2, 1984 (fig. 22). Throughout the study, there was a shift toward a zooplankton population dominated by copepods. Protozoans were never an important contributor to the zooplankton population.

The rotifers had a diversity of morphological variations and adaptations. For example, *Keratella* is one of the most widespread rotifers, capable of adapting to a variety of habitats. *Keratella cochlearis*, which dominated on February 2, 1984 (table 8), is one of the most common rotifers of the plankton in lakes in temperate regions (Hutchinson, 1967). Because most rotifers feed mainly by sedimenting seston particles into the mouth, this rotifer probably was more abundant at the beginning of the study because the lake was filling, and detritus and suspended sediment were more available in the open water. The rotifers that dominate generally are cold stenotherms that develop greatest populations during the winter and early spring. *Keratella* and *Polyarthra* are cold-water forms that also can tolerate decreased dissolved-oxygen concentrations. These rotifers can feed on the cellular contents of large diatoms,

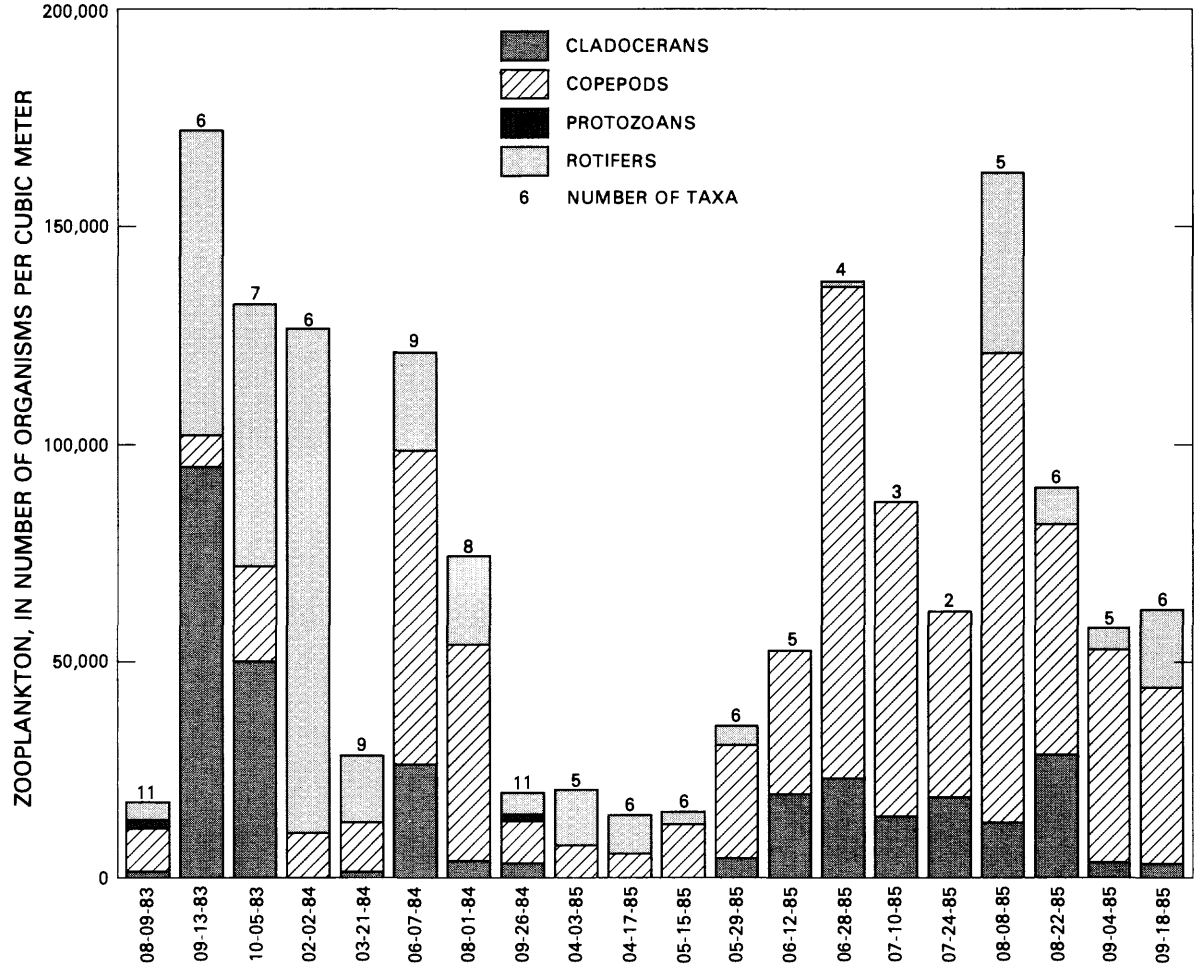

Figure 22.--Taxonomic composition and average densities of zooplankton collected from photic (total if only available) tows for all sampling dates at site A.

such as *Asterionella*, which dominated on February 2, 1984, and may increase in abundance during spring diatom blooms. A larger predatory rotifer, *Asplanchna* sp., dominated in early spring (April 3, 1985, and April 17, 1985, samples) (table 8) and thus, is abundant when diatoms are present. However, experiments have indicated that *Keratella cochlearis* feeds selectively on detritus when dead and living algae are present. The probable decrease of detrital material in the reservoir during the study possibly precluded this organism ever becoming dominant again after the March 21, 1984, sample (table 8). In contrast, *Polyarthra* sp. ingest only algal cells and was dominant during several sampling dates throughout the study.

Table 8.--*Dominant taxa of zooplankton collected from photic (total if only available) tows for each sampling date at site A*

[Organisms/m^3, organisms per cubic meter; Cop, copepods; Cla, cladocerans; Rot, rotifers; Prot, protozoans; --, not applicable]

Date	Taxa	Density (organisms/ m^3)	Taxa	Density (organisms/ m^3)	Taxa	Density (organisms/ m^3)
08-09-83	*Cyclops bicuspidatus thomasi* (Cop)	5,800	Nauplii (Cop)	4,100	*Difflugia* sp. (Prot)	1,900
09-13-83	*Bosmina longirostris* (Cla)	94,000	*Polyarthra* sp. (Rot)	56,000	*Asplanchna* sp. (Rot)	10,000
10-05-83	*Bosmina longirostris* (Cla)	49,500	*Polyarthra* sp. (Rot)	41,000	Nauplii (Cop)	15,000
02-02-84	*Keratella cochlearis* (Rot)	104,500	*Polyarthra* sp. (Rot)	9,400	Nauplii (Cop)	6,350
03-21-84	*Cyclops bicuspidatus thomasi* (Cop)	7,850	*Polyarthra* sp. (Rot)	6,650	*Keratella cochlearis* (Rot)	6,550
06-07-84	Nauplii (Cop)	57,500	*Cyclops bicuspidatus thomasi* (Cop)	14,500	*Bosmina longirostris* (Cla)	11,500
	--	--	*Daphnia rosea* (Cla)	14,500	--	--
08-01-84	*Cyclops bicuspidatus thomasi* (Cop)	30,500	Nauplii (Cop)	17,500	*Polyarthra* sp. (Rot)	17,000
09-26-84	*Cyclops bicuspidatus thomasi* (Cop)	5,800	Nauplii (Cop)	3,200	*Bosmina longirostris* (Cla)	2,600
04-03-85	*Asplanchna* sp. (Rot)	9,400	Nauplii (Cop)	5,200	*Polyarthra* sp. (Rot)	2,600
04-17-85	*Asplanchna* sp. (Rot)	5,250	*Cyclops bicuspidatus thomasi* (Cop)	4,000	*Filinia longiseta* (Rot)	1,800
05-15-85	Nauplii (Cop)	10,100	*Cyclops bicuspidatus thomasi* (Cop)	1,700	*Kellicotta longispina* (Rot)	950
05-29-85	Nauplii (Cop)	15,500	*Cyclops bicuspidatus thomasi* (Cop)	10,650	*Bosmina longirostris* (Cla)	3,600
06-12-85	*Cyclops bicuspidatus thomasi* (Cop)	18,500	*Bosmina longirostris* (Cla)	14,500	*Daphnia rosea* (Cla)	4,450
	--	--	Nauplii (Cop)	14,500	--	--
06-28-85	Nauplii (Cop)	81,000	*Cyclops bicuspidatus thomasi* (Cop)	32,000	*Daphnia rosea* (Cla)	12,000
07-10-85	*Cyclops bicuspidatus thomasi* (Cop)	62,000	*Daphnia rosea* (Cla)	13,500	Nauplii (Cop)	10,500
07-24-85	*Cyclops bicuspidatus thomasi* (Cop)	36,000	*Daphnia rosea* (Cla)	18,400	Nauplii (Cop)	6,900
08-08-85	Nauplii (Cop)	90,500	*Polyarthra* sp. (Rot)	41,000	*Cyclops bicuspidatus thomasi* (Cop)	17,500
08-22-85	Nauplii (Cop)	29,000	*Cyclops bicuspidatus thomasi* (Cop)	24,000	*Daphnia rosea* (Cla)	19,000
09-04-85	Nauplii (Cop)	27,000	*Cyclops bicuspidatus thomasi* (Cop)	13,500	*Diaptomus ashlandi* (Cop)	8,850
09-18-85	*Polyarthra* sp. (Rot)	15,500	*Cyclops bicuspidatus thomasi* (Cop)	14,500	Nauplii (Cop)	13,500

The cladocerans also were an important component of the zooplankton community but dominated only twice during the beginning of the study. They generally are important during summer and autumn and are relatively nonexistent during the winter and spring. *Bosmina longirostris* was the dominant zooplankton collected in the September 13, 1983, and October 5, 1983, samples, but *Daphnia rosea* commonly was abundant during the summer months of the study (table 8). The abundance of *Polyarthra* and *Keratella* can be greatly affected by the presence and distribution of *Asplanchna* and predatory copepods. For example, the predatory copepod *Cyclops* feeds effectively on the rotifers *Asplanchna* and *Polyarthra* but not on the rotifer *Keratella*. *Asplanchna* regularly eats *Keratella* but cannot capture *Polyarthra* because of its effective escape behavior. In many lakes (Wetzel, 1983), *Daphnia rosea* is the most important grazer and accounts for 80 percent of the total annual grazing activity. *Bosmina longirostris*, although very small (0.4-0.6 mm), commonly forms a dominant percentage of the total grazing during the autumn because of its large population densities (table 8).

The copepods currently (1985) are the most dominant zooplankton in Arvada Reservoir. Copepods dominated on 14 sampling dates, and *Cyclops bicuspidatus thomasi* dominated on 7 sampling dates. A number of studies have indicated that most adult cyclopoid copepods are carnivorous and that their predatory activities can be substantial in the population dynamics of other copepod species. For example, *Cyclops* preys largely on nauplii of *Diaptomus* and its own species. Some 30 percent of the nauplii population has been determined to be lost by copepod predation and by cannibalism (Wetzel, 1983). For this reason, how much effect the zooplankton population, as dominated by copepods, has on the phytoplankton population of Arvada Reservoir is difficult to ascertain. The zooplankton in Arvada Reservoir seem to be a healthy, diverse population, which helps control the phytoplankton population, thus increasing light transparency. According to Wetzel (1983), a positive correlation exists between the rates of production of phytoplankton and zooplankton. However, this does not necessarily imply that the herbivorous zooplankton are consuming the algae directly in proportion to the biomass and growth. Size, quality, and other factors affect ingestion and assimilation of algae.

TROPHIC STATE OF ARVADA RESERVOIR

According to Reckhow and Chapra (1983), a number of attempts have been made to establish a trophic-state index as a function of commonly measured water-quality variables. Designation of the trophic state (degree of nutrient enrichment) of a lake can be based on a number of criteria. Several methods of assigning the terms "oligotrophic" (small nutrient concentrations and productivity), "mesotrophic" (medium nutrient concentrations and productivity), or "eutrophic" (large nutrient concentrations and productivity) to lakes have been proposed (Hutchinson, 1957; Sakamoto, 1966; Fruh, 1967; Vollenweider, 1968, 1976; National Academy of Sciences, National Academy of Engineering, 1973; Dobson and others, 1974; U.S. Environmental Protection Agency, 1974). Because changes between oligotrophic and eutrophic conditions do not occur at well-defined intervals, a lake or reservoir may be considered oligotrophic by one criterion and eutrophic by another criterion. More robust trophic-state criteria or indices have been developed using a multivariate approach.

A numerical trophic-classification method developed by Carlson (1977) can be used to quantitatively define a trophic state and to monitor changes in water quality in an individual lake with time. This trophic-state index categorizes most lakes using a scale of 0 to about 100 (0 indicating almost no biomass for chlorophyll a). Carlson's (1977) index may be estimated from summer values of Secchi-disk depth, summer total-phosphorus concentration, summer chlorophyll a concentration, or a weighted combination of all three. Carlson (1977) used regression analysis to relate Secchi-disk depth to total-phosphorus concentration and to chlorophyll a concentration. Using a combination of all three indicator variables serves as a check on sampling techniques and analysis methods, and on assumptions concerning relations among various components of the lake ecosystem (Carlson, 1977). For example, all indicator variables should have the same or nearly the same value when transformed to the trophic-state index. However, if the three variables do not exactly coincide when transformed to the index values, it does not necessarily mean that the index values are inaccurate. Instead, the deviation may indicate certain factors about the dynamics of the lake or reservoir. For example, if the trophic-state index based on total phosphorus is larger than the trophic-state index based on Secchi-disk depth or chlorophyll a, it could indicate that either the lake is not phosphorus limited, or that grazing by herbivorous zooplankton is important (Shapiro, 1979). Actually, the best indicator variable of trophic state may vary from lake to lake and also may vary seasonally; therefore, the best indicator variable (total-phosphorus concentration, chlorophyll a concentration, or Secchi-disk depth) needs to be chosen based on available knowledge of the lake ecosystem (Carlson, 1977).

For Arvada Reservoir, data for the three indicator variables were within the range of data used by Carlson (1977) to calculate the trophic-state index. Because it is unknown which of the three indicator variables will be continually monitored in Arvada Reservoir in the future, and because deviations among trophic-state-index values calculated using the three indicator variables can provide valuable information about the lake, all three indicator variables were used to calculate the trophic state of Arvada Reservoir.

The trophic-state index (TSI) is calculated for the three indicator variables using the following equations:

$$TSI\ (SD) = 10\ \ 6 - \frac{ln\ SD}{ln\ 2}\ ,$$

$$TSI\ (TP) = 10\ \ 6 - \frac{ln\ \frac{48}{TP}}{ln\ 2}\ ,$$

$$TSI\ (Chl) = 10\ \ 6 - \frac{2.04\text{-}0.68\ ln\ Chl}{ln\ 2}\ ,$$

where SD = Secchi-disk depth, in meters;
TP = total-phosphorus concentration, in milligrams per liter;
Chl = chlorophyll *a* concentration, in micrograms per liter; and
ln = the natural logarithm (logarithm to the base 2) of the indicator variable.

It is emphasized that the values obtained using these equations are only indices of the relative trophic state of the reservoir and do not define the actual or absolute trophic state. However, the index itself is considered absolute; that is, the index is determined independently for each lake or reservoir and is not dependent on values from other lakes or reservoirs used in the original data base. In calculating the index for Arvada Reservoir, data were used from the sampling dates when the reservoir was thermally stratified. The results of the calculations are listed in table 9.

Table 9.--*Trophic-state indices for all sampling sites in the reservoir*

[TSI, trophic-state index; SD, Secchi-disk depth; TP, total phosphorus; Chl, chlorophyll *a*; --, no data]

Site and sampling date	TSI (SD)	TSI (TP)	TSI (Chl)
Site A			
1983	49	32	46
1984	48	35	48
1985	41	33	49
All dates	44	34	48
Site B			
1983	50	36	45
Site C			
1983	49	32	44
1984	48	--	--
All dates	49	--	--
Site D			
1984	51	32	42
Site E			
1985	41	35	49
All sites			
1983	50	34	44
1984	49	34	46
1985	41	35	49
All dates	45	34	48

The TSI values predicted using Secchi-disk depth and chlorophyll *a* concentration are similar. The TSI values predicted using total-phosphorus concentration are less than those values predicted by Secchi-disk depth and chlorophyll *a* concentration, indicating that phosphorus is not the only major limiting factor for algal growth. The TSI values predicted using Secchi-disk depth improved during the study period but remained fairly constant as predicted using total-phosphorus and chlorophyll *a* concentrations. As indicated by Carlson (1979), if a TSI value of 41 was considered the upper limit of oligotrophy, and a TSI value of 51 was considered the lower limit for eutrophy, the reservoir would be classified as mesotrophic based on TSI values predicted using Secchi-disk depth and chlorophyll *a* concentration. The reservoir would be classified as oligotrophic based on TSI values predicted using total-phosphorus concentration.

The TSI values predicted using total-phosphorus concentration may be misleading because accurate index values from total-phosphorus concentration depend on the assumption that phosphorus is the major limiting nutrient for algal growth (Carlson, 1977). He determined a deviation of the chlorophyll *a* index from the total-phosphorus index when ratios of total-nitrogen to total-phosphorus concentrations were less than 26:1. Occasionally, the ratios of total-nitrogen to total-phosphorus concentrations were less than or similar to 26:1 in Arvada Reservoir.

For comparison, the variables used to calculate the TSI index were allocated to trophic categories according to the fixed-boundary system discussed by the Organisation for Economic Co-operation and Development (OECD) (1982). Proposed boundary values for trophic categories are established as a link between the quantitative characteristics (Secchi-disk depth, total-phosphorus, and mean chlorophyll *a* concentrations) to the qualitative terminology (oligotrophic, mesotrophic, and eutrophic). The fixed-boundary system is based on best judgment by a large group of limnologists who participated in the elaborate OECD program. Application of numerical values of the trophic variables measured for Arvada Reservoir to the fixed-boundary system are similar to those calculations for the TSI developed by Carlson (1977). Proposed boundary values for trophic categories and the values for Arvada Reservoir are listed in table 10. The reservoir is eutrophic with respect to Secchi-disk depth, oligotrophic with respect to total-phosphorus concentrations, and mesotrophic with respect to chlorophyll *a* concentrations. The TSI values and assignment of variables to trophic categories provide a base from which deviations can be monitored as long as Secchi-disk depth, total phosphorus, and chlorophyll *a* are determined in the future monitoring program.

FUTURE MONITORING NEEDS

1. Weekly vertical-profile measurements with depth of water temperature, dissolved oxygen, specific conductance, and pH during thermal stratification would enable the reservoir manager(s) to document possible abrupt changes and cyclic variability of water quality in Arvada Reservoir. These measurements are relatively inexpensive, although manpower intensive, but can be valuable diagnostic aids for understanding the ecological functioning of the reservoir and can be valuable indicators of potential or ongoing biological conditions. With additional collection of dissolved-solids and specific-conductance data,

Table 10.--*Comparison of trophic variables for Arvada Reservoir to proposed boundaries for trophic categories*

[m, meters, mg/L, milligrams per liter; µg/L, micrograms per liter]

Trophic category	Trophic variables		
	Mean annual Secchi-disk depth (m)	Mean annual total-phosphorus concentration (mg/L)	Mean annual chlorophyll a concentration (µg/L)
Ultra-oligotrophic	>12	<0.004	<1.0
Oligotrophic	>6	<.010	<2.5
Mesotrophic	6-3	.010-.035	2.5-8
Eutrophic	3-1.5	.035-.10	8-25
Hypereutrophic	>1.5	>.10	>25
Arvada Reservoir	2.7	.009	5.7

a relation possibly could be established to estimate dissolved-solids concentrations from specific-conductance measurements.

2. Continued monitoring of Secchi-disk depth, total-nitrogen, total-phosphorus, and chlorophyll *a* concentrations on a weekly basis during thermal stratification will provide information necessary to assess possible changes in trophic state. Additional data for concentrations of total nitrogen, total phosphorus, and the biologically available nutrients (dissolved ammonia and dissolved nitrate as nitrogen, and dissolved orthophosphate as phosphorus) would be useful in determining which nutrient is limiting and in determining temporal changes, if any, in nutrient limitation.

3. Some trace-element concentrations (dissolved manganese, total mercury, and total zinc) and a radiochemical concentration (total uranium) exceeded standards and, therefore, could be analyzed on at least a seasonal basis to ensure that these concentrations are monitored. Total-iron and total-manganese concentrations are larger in the hypolimnion when Arvada Reservoir is thermally stratified and the hypolimnion is anaerobic. Therefore, these constituents could be analyzed more often during the summer.

4. Based on ratios of total-nitrogen and total-phosphorus concentrations, Arvada Reservoir seems to be phosphorus limited. However, algal-growth-potential determinations, results of empirical relations between concentrations of total phosphorus and chlorophyll *a*, and trophic-state-index calculations indicate that some other variable, probably nitrogen, is a colimiting factor. Therefore, periodic determinations of algal-growth potential using phosphorus and nitrogen spikes in samples collected during the spring and summer or using in-situ analysis methods would be useful in further identifying the limiting nutrient(s) in Arvada Reservoir.

5. Continued monitoring on a monthly basis of phytoplankton (composite samples) and zooplankton (photic tows) at one site in Arvada Reservoir should be sufficient to characterize taxonomic compositions and enumerate densities. Nicholls and Dillon (1978) have suggested that phytoplankton-volume determinations may be a better indicator of algal biomass than chlorophyll *a*. In addition, knowledge of what taxa of phytoplankton dominate is important because some taxa can be noxious and affect treatment of the reservoir water. Also, chlorophyll *a* content of algal cells varies among different taxa of algae, which is useful information when attempting to analyze deviations in total phosphorus and chlorophyll *a* relations. In addition, correlations between volume determinations and chlorophyll *a* concentrations would be useful in identifying the most appropriate indicator of algal biomass for Arvada Reservoir.

6. Because there was only about 1 month of pumping of Croke Canal water into Arvada Reservoir, continued monitoring of water in Croke Canal would be beneficial to more clearly identify possible water-quality changes in the reservoir after addition of water from Croke Canal. Data needs would include weekly in-situ measurements similar to those identified in number 1 of this section, and analyses of total-nitrogen, total-phosphorus, and chlorophyll *a* concentrations on a schedule similar to that identified in number 2 of this section. In addition, because concentrations of dissolved manganese, total iron, total mercury, and total zinc exceeded standards, these constituents could be analyzed on a periodic basis to ensure that these concentrations are monitored. Because Ralston Creek is the major inflow to the reservoir and represents the "natural" source, the same monitoring needs would apply for both Ralston Creek and Croke Canal. Not only would such monitoring provide useful management information on the effects of impounding this water system, but it might provide a measure by which a simple model could be developed for predicting future water quality in the reservoir. In addition, since water sometimes is released directly from Ralston Reservoir to Arvada Reservoir, the quality of Ralston Reservoir water could affect the future water quality of Arvada Reservoir. This relation also could be investigated during future monitoring and studies of Arvada Reservoir.

SUMMARY AND CONCLUSIONS

Arvada Reservoir was thermally stratified on most sampling dates, generally from April through September during the study period. Dissolved-oxygen concentrations ranged from 0.0 to 12.0 mg/L, and the reservoir was anaerobic below the 10-m depth during most of the summer. Measurements of specific conductance with depth indicated that the reservoir contains small dissolved-solids concentrations. In addition, effects of Croke Canal water on the reservoir during June and July 1985 were apparent, as indicated by decreased specific conductance within a 4-m zone where the Croke Canal pipeline extends into the reservoir. Values of pH in the reservoir ranged from 6.4 to 9.2; therefore, the values were near the range of 6.5 to 9.0 specified as standards for the reservoir. Secchi-disk depth measurements in the reservoir ranged from 0.9 to 5.5 m, and generally the values increased slightly during the study period. This increase probably was because of a decrease in nonalgal turbidity after the reservoir completed filling.

Ralston Creek, the major inflow to Arvada Reservoir, had water quality suitable for most uses; dissolved-solids concentrations were small (indicated by specific-conductance measurements generally less than 400 µS/cm) and dissolved-oxygen concentrations were saturated on most sampling dates. As in Ralston Creek, specific-conductance measurements in Croke Canal indicated small dissolved-solids concentrations, and dissolved-oxygen concentrations in the canal always were larger than the minimum of 7.0 mg/L specified as a spawning requirement for Ralston Creek.

The results of chemical analyses indicated that water from Arvada Reservoir generally is of suitable quality for a raw-water-supply source and for maintenance of aquatic life. However, small dissolved-oxygen concentrations at lower depths in the reservoir during the summer may restrict vertical distribution of fish. Total-nitrogen and total-phosphorus concentrations were small, and ratios of total-nitrogen to total-phosphorus concentrations generally were larger than 26:1 in the reservoir. The exceptions for the suitability of water from Arvada Reservoir were that dissolved-manganese concentrations occasionally exceeded secondary drinking-water standards, and total-mercury and total-zinc concentrations occasionally exceeded the criteria for maintenance of aquatic life. In addition, total-uranium concentrations occasionally exceeded standards specified for the reservoir. In Ralston Creek, the total-nitrogen and total-phosphorus concentrations were similar to those in the reservoir; whereas, Croke Canal had larger total-nitrogen concentrations. As a consequence, the ratios of total-nitrogen to total-phosphorus concentrations generally were less than 26:1 in Croke Canal. No trace-element concentrations exceeded drinking-water standards or aquatic-life criteria in Ralston Creek. However, in Croke Canal, dissolved-manganese concentrations exceeded the secondary drinking-water standard, and total-iron, total-mercury, and total-zinc concentrations exceeded the aquatic-life criteria.

The phytoplankton community was diverse in Arvada Reservoir; densities ranged from 1,400 to 29,000 cells/mL, and diversity-index values were as much as 4.65. Diatoms were most numerically abundant. Chlorophyll a concentrations ranged from 0.0 to 20.4 µg/L, and overall mean values in the reservoir averaged 5.4 µg/L. The measured mean concentrations of chlorophyll a generally were not within the 95-percent confidence interval of the predicted mean chlorophyll a concentrations. Sources of variability in this relation as applied to Arvada Reservoir possibly occur because of: (1) Inherent assumptions used in the development of the relation, (2) sampling or analytical error, (3) temporal and spatial patchiness of phytoplankton and nutrient distribution, and (4) differences in chlorophyll a content of algal cells. In addition, algal-growth-potential determinations made using water samples collected from the reservoir indicated that spikes of phosphorus and nitrogen increased algal growth. Therefore, the assumption of phosphorus as the limiting nutrient used in developing predictions for chlorophyll a concentrations in Arvada Reservoir may be inconclusive. Algal biomass, as measured by chlorophyll a concentration, was larger in Croke Canal (mean of 6.4 µg/L) than in Ralston Creek (mean of 1.2 µg/L). Because Croke Canal has larger nutrient concentrations and algal biomass, prolonged periods of pumping this water into Arvada Reservoir possibly could increase algal biomass in the reservoir, as opposed to an exclusive inflow from Ralston Creek.

The zooplankton densities collected from single tows in Arvada Reservoir ranged from 10,000 to 180,000 organisms/m^3. A total of 24 taxa of zooplankton were identified from samples collected at sites in the reservoir. The zooplankton seem to help control the phytoplankton population, thus increasing light transparency.

Calculations of trophic state based on Secchi-disk-depth measurements, total-phosphorus concentrations, and chlorophyll a concentrations indicated similarity between the trophic-state index predicted using Secchi-disk depth and chlorophyll a. Based on these index values, Arvada Reservoir would be classified as mesotrophic. The index value predicted using total-phosphorus concentrations indicates that the reservoir would be classified as oligotrophic. The deviation between the indices indicates that other factors, possibly nitrogen limitation, affect transparency and algal biomass.

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